Applied and Environmental Soil Science

Soil Management for Sustainable Agriculture

Guest Editors: Philip J. White, John W. Crawford, María Cruz Díaz Álvarez, and Rosario García Moreno

Soil Management for Sustainable Agriculture

Applied and Environmental Soil Science

Soil Management for Sustainable Agriculture

Guest Editors: Philip J. White, John W. Crawford, María Cruz Díaz Álvarez, and Rosario García Moreno

Editorial Board

Contents

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 850739, 3 pages
doi:10.1155/2012/850739

Editorial

Soil Management for Sustainable Agriculture

Philip J. White,[1] John W. Crawford,[2] María Cruz Díaz Álvarez,[3] and Rosario García Moreno[3]

[1] *Ecological Sciences, The James Hutton Institute, Invergowrie, Dundee DD2 5DA, UK*
[2] *Faculty of Agriculture, Food and Natural Resources, The University of Sydney, Sydney, NSW 2006, Australia*
[3] *Centre for Studies and Research on Agricultural and Environmental Risk Management (CEIGRAM), Universidad Politécnica de Madrid, 28040 Madrid, Spain*

Correspondence should be addressed to Philip J. White, philip.white@hutton.ac.uk

Received 19 July 2012; Accepted 19 July 2012

The soil sustains most living organisms, being the ultimate source of their mineral nutrients. Good management of soils ensures that mineral elements do not become deficient or toxic to plants, and that appropriate mineral elements enter the food chain. Soil management is important, both directly and indirectly, to crop productivity, environmental sustainability, and human health. Because of the projected increase in world population and the consequent necessity for the intensification of food production, the management of soils will become increasingly important in the coming years. To achieve future food security, the management of soils in a sustainable manner will be the challenge, through proper nutrient management and appropriate soil conservation practices. Research will be required to avoid further degradation of soils, through erosion or contamination, and to produce sufficient safe and nutritious food for healthy diets.

The aim of this special issue is to present current research to assure food security whilst preserving natural resources. It comprises 16 papers arising from the Soil Management for Sustainable Agro-Food Systems Session at the European Geosciences Union General Assembly in April 2011. These range from reviews of the effects of different soil management practices on the sustainability of agricultural systems to papers reporting the influence of specific organic and inorganic amendments on the productivity and quality of particular crops.

The Special issue begins with an overview by P. J. White et al. of the role of plant mineral nutrition in food production, the delivery of essential mineral elements to the human diet, and the prevention of harmful mineral elements entering the food chain. The authors describe our progress towards global food security through the development of improved agronomic practices and novel crop genotypes for the sustainable intensification of agriculture. This paper is complemented by articles by R. Saha et al., who review the consequences of deforestation coupled with shifting cultivation practices on soil degradation in Northeast India, and S. E. Obalum et al., who review the problem of soil degradation in Sub-Saharan Africa. R. Saha et al. report massive losses of soil, soil carbon (C), nitrogen (N), phosphorus (P), potassium (K), calcium, magnesium, manganese, and zinc (Zn) following deforestation in the northeastern hill region of India with shifting cultivation practices. The consequent reduction in soil fertility prevents sustained agricultural production. However, they note that the adoption of appropriate agroforestry systems can reduce soil losses, increase soil organic matter (SOM), improve soil physical properties, and preserve water resources. In addition, techniques such as zero or minimum tillage, mulching, cultivating cover crops, and hedgerow intercropping can be used to increase SOM and sustain soil health. S. E. Obalum et al. report that land degradation, particularly soil erosion, also has a significant negative effect on soil quality and productivity in Sub-Saharan Africa. These authors propose the adoption of a lowland-based rice-production technology, termed the sawah ecotechnology, to meet demands for food security in this region. They argue that this farmer-oriented, low-cost system of managing soil, water, and nutrient resources could not only improve agricultural productivity but also alleviate the negative environmental impacts of land degradation in this region.

In many areas of the world, the loss of topsoil, either through mineral imbalance or erosion, is the single largest threat to agricultural productivity. Soil erosions by wind and

water are the main processes by which topsoil is lost. R. García-Moreno et al. report that soils with high soil surface roughness (SSR), such as those produced with conservation tillage, are less susceptible to erosion, and that there is an inverse relationship between SSR and soil porosity. They suggest that these soil properties might be used to predict the susceptibility of a soil to erosion by wind or water.

The influence of tillage on the physical, chemical, and microbiological properties of the soil is considered in several papers in this Special issue, with reference to specific agricultural systems. X. Gao and C. A. Grant report that durum wheat (*Triticum durum*) grown in the Canadian prairies tends to have greater grain yield, greater grain Zn concentrations, and lower grain cadmium (Cd) concentrations when cultivated with reduced tillage than with conventional tillage. The preceding crops in the rotation, whether spring wheat-flax, or canola-flax have little influence on grain yield, grain Cd concentration, or Zn concentration, but increasing P-fertilizer application tends to decrease grain Zn concentrations. This study suggests that tillage management can have beneficial effects on both grain yield and nutritional quality. R. P. Mathew et al. compared the long-term effects of conventional tillage and no-tillage practices on soil microbial communities in a silt loam soil under continuous maize (*Zea mays*) production in Alabama, USA. They observed that microbial biomass was greater in the topsoil from the untilled plots than the conventionally tilled plots, and also had greater phosphatase activity and higher carbon and nitrogen contents. The authors conclude that conservation tillage practices can, therefore, improve both the microbiological and physicochemical properties of soil. A. Munodawafa reports that grain yields of maize grown under semiarid conditions on the infertile, sandy soils of southern Zimbabwe can be predicted accurately from the amount and timing of rainfall. She observes that, for a given amount of rainfall, similar yields were achieved using mulch ripping (0.13 t ha^{-1} cm^{-1} rainfall) and conventional tillage (0.12 t ha^{-1} cm^{-1} rainfall), which were greater than those using tied ridging (0.09 t ha^{-1} cm^{-1} rainfall). However, much greater soil erosion occurred using conventional tillage than mulch ripping or tied ridging cultivation. She recommends that mulch ripping be practiced in this region, since the loss of topsoil under conventional tillage will ultimately result in a decline in productivity over time. M. Watkins et al. observe that in the well-managed dairy pastures of the Gippsland Region of south-eastern Australia, P and N are lost to the environment as dissolved rather than particulated forms. They report that the concentrations of P and N in soil solutions from ryegrass (*Lolium perenne*) or mixed ryegrass and clover (*Trifolium repens*) pastures are significantly lower in ploughed than in unploughed plots. Thus, they conclude that ploughing might reduce the amounts of P and N released to the environment from intensive dairy farms in this region.

Organic amendments often improve the productivity of soils and the nutritional value of crops grown thereon. In particular, crop residues can be used to increase the phytoavailability of essential mineral nutrients, reduce the phytoavailability of toxic mineral elements, improve soil physical properties, and promote a beneficial soil biota. In Ghana, cassava is an important staple crop, but it is also be used as a raw material for the production of industrial starch and ethanol. S. Adjei-Nsiah and O. Sakyi-Dawson demonstrate that cassava can contribute to mineral nutrient recycling, and to the maintenance of soil fertility, when integrated into crop rotations. Furthermore, they argue that the production of cassava for industrial purposes can contribute to poverty reduction without excessive depletion of soil mineral resources in the forest/savannah agroecological zone of Ghana. S. Adjei-Nsiah also reports that palm bunch ash, one of the major waste products generated from processing palm fruit, can be used as an effective, local, low-cost, K-fertilizer and liming material for maize production in Ghana. C. A. Abreu et al. report that the application of sugar cane filter cake at 40–80 Mg ha^{-1} organic-C can reduce barium (Ba) concentrations and increase shoot dry matter of sunflower (*Helianthus annuus*) and castor oil (*Ricinus communis*) plants, but not oilseed radish (*Raphanus sativus*), growing on a Brazilian soil (pH 7.5) contaminated with automobile scrap. However, neither sugar cane filter nor peat applications reduced soil Ba availability, which, they suggest, might be due to an effect of liming the soil. K. L. Rothlisberger et al. demonstrate that seed meal remaining after the extraction of oil for biodiesel production from white mustard (*Sinapis alba*), Indian mustard (*Brassica juncea*), camelina (*Camelina sativa*), or jatropha (*Jatropha curcas*) can act as a bioherbicide on johnsongrass (*Sorghum halepense*) and redroot pigweed (*Amaranthus retroflexus*), but that the efficacy and specificity of their bioherbicidal effects are related to plant species and affected by rate and timing of their application. M. M. Moreno et al. compared SOM, SOM mineralization, microbial biomass, and microbial activity in organic and conventional production systems for a rainfed crop rotation (durum wheat-fallow-barley-vetch) in the semiarid region of Castilla-La Mancha, Spain. Although it is often observed that management practices supplying more carbon to the system lead to the accumulation of more SOM, greater soil microbial biomass, and increased microbial activity, they observed that SOM was higher with chemical fertilization, which, they speculate, might be a consequence of either low compost inputs (2500 kg ha^{-1}) to the organic rotation or the arid conditions. Soil nitrate content was also higher when chemical fertilizers were applied, as were crop yields.

A. Korsaeth reports the N, P, and K budgets over a ten-year period of six crop rotations in a long-term experiment in southeast Norway. He observes that the conventional arable system and the three organic systems studied had negative N budgets, suggesting a reduction in soil N content. By contrast, a modified arable-farming practice with environmentally sound management appeared to be balanced with respect to N, and conventional practice for mixed dairy production generated an N surplus. Budgets for all conventional systems indicated P and K surpluses, whereas all organic systems appeared to mine the soil for P and K. Although these calculations corresponded well with the measured changes in topsoil P, only a common ranking of the systems for their N and K budgets and the measured N and K in topsoil was observed. He concludes, therefore, that crop production could be mining a soil of N and K over

many years before it is detected by traditional soil analyses, and that nutrient budgeting might be used to predict mineral imbalances of agricultural practices. In a similar study, P. Sharma et al. compared irrigation efficiencies, and water and nitrate balances, for onions (*Allium cepa*) grown with furrow or drip irrigation in an arid area of southern New Mexico where water is a limited resource for crop production. They observed that both the irrigation efficiency and the N-fertilizer use efficiency were slightly greater for drip systems than for furrow systems.

The paper of this special issue by J. A. Campos et al. reports the concentrations of 18 mineral elements in fruiting bodies from ectomycorrhizal, saprophytic, and epiphytic fungi from a mixed forest of pines and oaks on quartzite acidic soils in Ciudad Real, Spain. They report significantly higher copper (Cu) and rubidium (Rb) concentrations in fruiting bodies from ectomycorrhizal species and significantly higher Zn concentrations in fruiting bodies from saprophytic species. The species *Clitocybe maxima* and *Suillus bellini* appear to "hyperaccumulate" Cu and Rb, respectively.

Acknowledgments

We thank the European Geosciences Union for supporting the session on Soil Management for Sustainable Agro-Food Systems at the General Assembly in April 2011, all the reviewers for their timely reports and constructive comments, and the staff of Editorial Office of *Applied and Environmental Soil Science* for their assistance throughout this project.

Philip J. White
John W. Crawford
María Cruz Díaz Álvarez
Rosario García Moreno

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 539582, 17 pages
doi:10.1155/2012/539582

Research Article

N, P, and K Budgets and Changes in Selected Topsoil Nutrients over 10 Years in a Long-Term Experiment with Conventional and Organic Crop Rotations

Audun Korsaeth

Arable Crops Division, Norwegian Institute for Agricultural and Environmental Research, 2849 Kapp, Norway

Correspondence should be addressed to Audun Korsaeth, audun.korsaeth@bioforsk.no

Received 2 December 2011; Accepted 30 April 2012

Academic Editor: Philip White

This study presents soil system budgets of N, P and K in six contrasting cropping systems during 10 years of a long-term experiment in southeast Norway. The experiment included systems with arable cash-cropping and with mixed arable-dairy cropping (cash- and fodder crops), with organic and conventional management represented in both groups. All major nutrient inputs and outputs were measured or estimated. State of the art conventional cash-cropping appeared to be balanced in terms of N, whereas conventional mixed cropping had an N surplus. By contrast, less up to date conventional arable cash-cropping and all the organic systems showed indications of soil organic N depletion (negative N budgets). All the organic systems showed that mining of the soil P and K content occurs, whereas the conventional systems all had P and K surpluses. The results corresponded well with measured differences between systems in terms of ignition loss, P-AL, K-AL and K-HNO_3 measured in 2009. This study shows that a fertile soil may be exposed to substantial mining of N, P and K over many years before it is detectable by traditional analyses, and that field nutrient budgeting is a feasible, but data-demanding, approach to detect such misbalances at an early stage.

1. Introduction

In 1989, a large cropping system experiment, facilitated for measurements of runoff and leaching, was established at Apelsvoll in southeast Norway. Over the years, this experiment has provided data for many studies covering a range of different topics, including yields and yield quality (e.g., [1]), nutrient leaching and runoff losses (e.g., [2]), economic aspects (e.g., [3]), soil microbiology (e.g., [4]), soil physical and chemical properties (e.g., [5]), and the relation between food production and N losses [6].

Some major adjustments of the experimental design were made in 2000 [6]. In this overview, a synthesis of the results obtained after these changes are given for the major nutrient flows of N, P, and K, with focus on changes in topsoil nutrient pools, as affected by misbalances between nutrient inputs and outputs at field level.

Numerous long-term experiments have shown that crop rotation and management affect soil fertility (e.g., [7–13]). However, considerable time is needed before identifiable changes in soil fertility emerge [14]. Nutrient budgets have been used widely in a range of farming systems to assess long-term sustainability (e.g., [15]), thus, supplementing soil measurements. In a discussion of uncertainties in nutrient budgets, Oenema et al. [16] distinguished between farm gate soil surface and soil system budgets. The latter accounts for nutrient inputs and outputs, recycling of nutrients within the system, nutrient loss pathways, and changes in soil nutrient pools. Soil system budgets were considered to possess the highest uncertainty of the three budgeting approaches, since nutrient losses via leaching, runoff, volatilization, and denitrification are classified as the most uncertain nutrient flows [17]. De Vries et al. [18], when estimating uncertainties in the soil system N budget of The Netherlands, reported

that leaching to ground water and leaching to surface water had the highest relative uncertainty (coefficient of variation). Acquiring quality data on nutrient drainage and runoff may, therefore, considerably reduce the uncertainties of the soil system budget approach.

The present study is aimed at comparing the effects of management (i.e., organic versus nonorganic) and type of production (i.e., arable cash cropping versus mixed dairy farming) on their long-term sustainability in terms of plant nutrition, by a combination of soil system nutrient budgeting and soil measurements. Results on drainage discharge and water-borne nutrient losses will be presented in more detail elsewhere [19].

2. Material and Methods

2.1. Experimental Site and Treatments. In 1989, a 3.2 ha large experiment with pipe-drained plots was established at Apelsvoll Research Centre in central southeast Norway (60°42′ N, 10°51′ E, altitude 250 m). The climate of the region is humid continental with a mean annual precipitation of 600 mm and a mean annual temperature of 3.6°, and 12.0°C in the growing season (May to September). On the experimental area, which slopes 2–8% northeast, deforestation was performed in 1935, whereafter it was used as pasture until 1975. During the following years, up to the establishment of the experiment in 1988, the field was cropped with a rotation including 10% root crops, 40% cereals, and 50% ley, using an average of 10 tonnes cattle slurry $ha^{-1}\,yr^{-1}$ plus regular amounts of inorganic fertilizer. The first year after draining the experimental site (1989), the area was cropped with barley (*Hordeum distichum* L.). The major soil group of the experimental area is classified as Endostagnic Cambisol [20], with dominantly loam and silty sand textures. More detailed soil characteristics have been presented by Riley and Eltun [21], partly shown in Table 1.

The experimental site comprises 12 blocks (30 × 60 m), separated by 7.5 m grass border zones (Figure 1). In each block, surface runoff is collected at the lower end and led to a sedimentation tank, and the blocks are separately drained with PVC pipes at a depth of 1 m with 7.5 m spacing. Surface runoff from the sedimentation tank and drainage water is transported in sealed plastic pipes to measuring stations equipped for discharge measurements (tipping buckets) and volume proportional sampling.

Using a randomised complete block design, six cropping systems, each with 2 replicates, were established on the twelve blocks in 1989. The first ten years (1989–1999) the experiment comprised three arable systems (conventional and integrated arable cropping without farmyard manure and organic cropping with some farmyard manure) and three mixed dairy systems (conventional, integrated, and organic production of both arable and forage crops, all with farmyard manure). Each block comprised eight 7.5 × 30 m plots, on which all of the arable and/or fodder crops in the rotation were grown each year.

Some major adjustments of the experimental design were made in 2000. The number of rotation plots was reduced

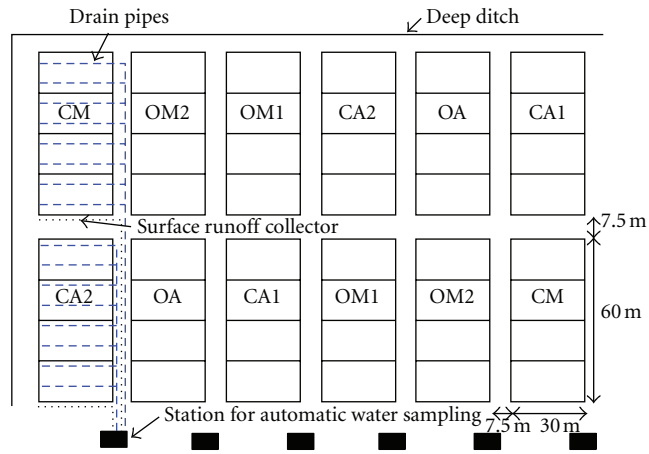


Figure 1: Layout of the cropping system experiment at Apelsvoll.

from eight to four by merging pairs of neighbouring plots, thus, reducing rotation length from eight to four years, but still with each crop present every year. A new organic mixed dairy system was introduced instead of the integrated mixed dairy system, and some smaller changes were made in the management of the other systems. The six adjusted cropping systems are described briefly below (see Table 2 for details).

CA1. Conventional arable cropping, managed as was common for the region in 1985 (tillage and fertilization as in 1985, but for practical reasons, present-day inputs of seeds and chemical plant protection). The year 1985 was selected, since the North-Sea Agreement (1987) used this year as the base for its planned 50% reduction in nutrient leaching to the North Sea within a 10-year period. Before this date, less attention was paid to nonpoint source losses of nutrients attributed to farming activities, and this cropping system is, thus, used as a reference.

CA2. Conventional arable cropping, using currently available knowledge in order to minimize the ratio of N lost by leaching and runoff to production. This optimisation involves the use of catch crops, split application of fertilizer, and reduced noninversion tillage.

OA. Organic arable cropping without cattle slurry, but with 25% of the area used for green manure (grass clover ley).

CM. Conventional mixed dairy farming, optimised similarly to CA2, but with spring ploughing, 50% of the area as grass clover ley and the use of slurry (amounts calculated from the theoretical number of cows sustained, see Section 2.3 for details).

OM1. Organic mixed dairy farming with 50% of the area as grass clover ley and slurry use (amounts calculated as in CM).

OM2. Organic mixed dairy farming with 75% of the area as grass clover ley and slurry use (amounts calculated as in CM).

Table 1: Mean values of selected soil physical parameters in topsoil (0–30 cm) measured in 1988, prior to the start of the experiment (from [21]).

System	Parameter number[1]									
	1	2	3	4	5	6	7	8	9	10
CA1	51	10	11	29	12	7	6	43	38	19
CA2	48	10	6	27	11	3	9	52	33	15
OA	49	7	7	30	12	4	6	46	35	19
CM	48	9	8	27	12	5	9	47	35	19
OM1	49	7	9	30	12	6	8	50	33	17
OM2	50	8	7	30	12	4	5	45	36	19
s.e.[2]	1.5	1.2	2.5	1.2	0.5	2.1	1.5	1.5	1.0	1.0
Mean	49	9	8	29	12	5	7	47	35	18

[1] Parameter number: (1) total porosity (%), (2) air capacity at pF2 (%), (3) air permeability at pF2 (μm^2), (4) total available water (pF 2–4.2) (%), (5) nonavailable water (pF > 4.2) (%), (6) hydraulic conductivity (cm/h), (7) gravel content (%), (8) sand content (%), (9) silt content (%), (10) clay content (%).
[2] Standard error of means.

The results in 2000 were partly influenced by the previous management. Therefore, this paper deals with the results from the decade May 2001–April 2011.

2.2. Measurements. Within each plot, dry matter (DM) yields of grass, grain, straw (when removed), and potato tubers were measured in quadruplicate (subplot size 1.5 × 6 m). Straw was removed from all the cereal plots of CA1 and from the plots with barley undersown with grass clover ley in OA, CM, OM1, and OM2. Crude protein contents of the cereals were measured by near infrared reflectometry (INFRA 250, Technicon, USA). Potato tuber size distribution and quality parameters were determined according to standard procedures. The proportion of legumes in the grass clover ley was determined visually before harvest.

From 2006 onwards, plant samples of all harvested crops (0.2 g DM) were digested in a mixture of sulfuric acid and hydrogen peroxide and analysed for N and P colorimetrically with an autoanalyser (Skalar 5100, Actlabs, Canada) and for K by flame photometry (Corning 400, Sherwood Scientific Ltd., UK).

Cattle slurry was sampled 1-2 weeks before application and analysed for total-N using the Kjeldahl method. Ammonium-N and nitrate-N were extracted with 2 M KCl and determined colorimetrically with an autoanalyser (Traacs, Bran and Luebbe, Germany).

The water samples (drainage water and surface runoff) were analysed on a monthly basis for total N, ammonium-N, nitrate-N, total P, phosphate P, and total K and determined spectrophotometrically (DR2800 spectrophotometer, Hach Lange, Germany). Potassium was first included in May 2009.

Soil samples have been taken every 3–5 years since 1989. Due to differences in sampling depths, sampling locations, and parameters analysed, only selected samples are comparable. In this study results are shown for topsoil samples (0–25 cm depth) taken in 1996, 1999, 2003, and 2009. The samples taken in 1999 and 2009 were analysed for ignition loss (at 550°C), whereas samples taken in 1996, 2003, and 2009 were used to quantify plant available P and K and acid soluble K. Plant available P (P-AL) and K (K-AL) was extracted by a mixture of acetic acid and ammonium lactate, according to Egnér et al. [22], whereas acid soluble K (K-HNO_3) was extracted by boiling in 1 M HNO_3. The P and K concentrations in the extracts were analysed by inductively coupled plasma (ICP) techniques (SPECTRO GENESIS, Analytical Instruments GmbH, Germany).

2.3. Calculations and Estimates. Dry and wet atmospheric N deposition was set to 2.7 and 7.2 kg N ha^{-1} yr^{-1}, respectively, and wet atmospheric K deposition was set to 2.1 kg ha^{-1} yr^{-1}, based on measurements at the nearest monitoring station, Hurdal [23], about 50 km south of Apelsvoll. Dry depositions of K and wet and dry P depositions were assumed to be negligible.

Symbiotic N fixation was estimated in accordance with Korsaeth and Eltun [2]. Nonsymbiotic N fixation was considered to be negligible. Nutrient inputs with seeds were estimated using measured N, P, and K contents of harvested grain and potatoes from the CON-A system. Literature values were selected for legumes and grasses.

The amounts of N in cattle slurry applied were calculated as the sum of N in harvested forage (grass clover ley) and feed concentrates (purchased and/or produced on the farm), minus the estimated N losses occurring from harvest until slurry application (forage losses, gaseous losses from the cow shed, and during slurry storage) and N exports via milk and livestock, as described in detail by Korsaeth [6]. The number of cows which each farming system could sustain was calculated from the average total available feed in the system during the previous three years (sliding mean) and the feed requirement for milk production, maintenance, activity and replacement. On the conventional farm, it was assumed that purchased cereal-based feed concentrates corresponded to 25% of the total fodder units available (feed concentrates plus forage grass). The organic systems were assumed to be completely self-sufficient, using on-farm produced barley as feed concentrates, for at most 20% of the

Table 2: Characteristics of the cropping systems at Apelsvoll, 2001–2010.

Crop rotation	Fertilizer, kg ha^{-1}			Slurry, kg ha^{-1}			Plant protection	Soil tillage
	N	P	K	N	P	K		
Conventional arable (CA1)[1]								
Potatoes	119	51	183				Chemical, mechanical	Autumn ploughing and spring harrowing
Spring wheat	141	25	66				Chemical	
Spring oats	120	22	57				Chemical	
Spring barley	120	22	57				Chemical	
Conventional arable (CA2)								
Potatoes	108[2]	47	164				Chemical, mech.	Spring harrowing only[7]
Wheat + catch crop[3]	134[2, 4]	29	80				Chemical, mech.[5]	
Oats + catch crop[6]	116[2]	21	55				Chemical, mech.[5]	
Barley + catch crop[3]	116[2]	21	55				Chemical, mech.[5]	
Organic arable (OA)								
Barley[8]							Manual weeding	Spring ploughing and harrowing
Grass clover[9]							Manual w.	
S. wheat + catch crop							Manual w., mech.[5, 10]	
Oats + peas							Manual w., mech.[10]	
Conventional mixed dairy farming (CM)								
Barley[11]	50[2]	9	24	70	10	90	Chemical	Spring ploughing and harrowing
1st ley year	99[2]	16	75	57	8	73	—	
2nd ley year	114[2]	19	86	80	11	104	—	
S. wheat + catch crop	86[2, 4]	16	43	69	10	90	Chemical, mech.[5]	
Organic mixed dairy farming (OM1)								
Barley[11]				75	11	98	Manual w.	Spring ploughing and harrowing
1st ley year				24	2	31	Manual w.	
2nd ley year				48	6	63	Manual w.	
S. wheat + catch crop				83	12	107	Manual w., mech.[5, 10]	
Organic mixed dairy farming (OM2)								
Barley[11]				93	13	112	Manual w.	Spring ploughing and harrowing
1st ley year				63	8	80	Manual w.	
2nd ley year				98	13	122	Manual w.	
3rd ley year				75	10	93	Manual w.	

[1] Managed as was common for the region in 1985 (tillage and fertilization as in 1985, but for practical reasons, present-day inputs of seeds and chemical plant protection).
[2] Fertilizer level in spring adjusted according to regional recommendations, based on measurements of mineral N in topsoil in early spring. Average values are shown.
[3] Perennial ryegrass (*Lolium perenne* L), sown about one week after the cereals.
[4] Split application of fertilizer with about 75% given at sowing, and 0–60 kg N ha^{-1} applied at growth stage (GS) 49, according to measurements with the N tester [6].
[5] Weed harrowing performed when the cereals are at GS 11-12.
[6] Italian ryegrass (*Lolium multiflorum* Lam), sown about one week after the oats.
[7] Performed twice with a horizontally rotating harrow.
[8] With undersown grass-clover mixture. Seed mix: 80% Timothy (*Phleum pratense* L.), 10% red clover (*Trifolium pratense* L.), and 10% white clover (*Trifolium repens* L).
[9] Green manure, not harvested but mulched 3-4 times per season.
[10] Harrowed in autumn after harvest some years to reduce the weed pressure.
[11] With undersown grass-clover ley. Seed mix: 60% Timothy (*Phleum pratense* L.), 30% Meadow fescue (*Festuca pratensis* L.), and 10% red clover (*Trifolium pratense* L.).

total feed requirement. Barley not used as concentrates was assumed to be sold.

The yearly average cereal yields of OA were reduced by 25% to correct for the area used for green manure production. In 2007, the mixture of oat and peas in OA was replaced in the rotation by *faba* bean (*Vicia faba* L.). These beans were totally damaged by chocolate spot (*Ascochyta* blight disease) and not harvested. All OA data from 2007 were, thus, excluded from further analyses.

Nutrient concentrations (N, P, and K) in harvested products (crops and straw) for the years 2001–2005 were set equal to the averages (separately for each cropping system) for the years 2006–2010. Removal of N, P, and K was calculated as the measured dry weight of products removed from the field multiplied by the estimated nutrient concentrations. Amounts of N, P and K in harvested grass were reduced by 10% to correct for likely losses under practical harvest.

Gaseous N-emissions (N_2O-N, NO_x-N, and NH_3-N) were estimated from the IPCC framework [24], which comprises estimates for both direct and indirect emissions. Nitrogen sources included in the direct estimates used here were mineral N fertilizer, applied N in slurry, and N in above-ground and below-ground crop residues. Net N mineralization associated with possible loss of SOM resulting from contrasting management was not considered. The volatilization of NH_3-N (and NO_x-N) is in the IPCC framework [24] related to the input of mineral fertilizer and organic N additions, not including crop residues. This implies that the mulched grass clover in OA would have zero emissions of NH_3-N using the IPCC-approach, which is very unrealistic (e.g., [25]). Hence, the volatilization of NH_3-N from this crop was calculated by means of a separate method [6].

Nutrient runoff occurring during each agrohydrological year, lasting from 1 May to 30 April, was attributed to the cropping season within that period. Calculations of N, P, and K transported via surface runoff and drainage water were based on measured nutrient concentrations and volumes of surface and drainage water. Organic N was calculated as the difference between total N and the sum of ammonium-N and nitrate-N. Potassium runoff occurring during the agrohydrological years 2001–2008 were set equal to the measured average K runoff for the agrohydrological years 2009/10 and 2010/11.

Nutrient soil system budgets were calculated separately for each system by considering all major flows of N, P, and K, respectively, with the above-ground crops representing the upper boundary and the drain pipes the lower boundary. A positive soil system budget, that is, where the inputs exceeded the outputs, was taken as an indication of nutrient accumulation, whereas a negative budget was taken as an indication of soil mining of the nutrient in question.

2.4. Statistics. Analyses of variance (ANOVA) were performed on yields and nutrient concentrations, using a split-plot model with cropping system as main plot and year as subplot. Grass-clover ley yields were analysed as the sum of two cuts, whereas differences in nutrient concentrations were analysed for each cut separately. Paired comparisons (LSD) were performed [26]. Comparisons of soil chemical properties measured on different occasions were conducted using the paired Student's t-test. In all tests, significance was assumed at P levels < 0.05. Mean data are presented with their standard errors (s.e.).

3. Results

3.1. Yields. There were significant yield differences between cereal crops within each group of cropping systems (Table 3). The conventional arable systems (CA1 and CA2) gave the largest overall cereal yields; the conventional mixed dairy system (CM) was intermediate, whereas the organic systems gave the lowest yields. The organic arable system had the lowest (area corrected) yields overall, achieving only 40 and 44%, respectively, of the barley and wheat harvested in the arable conventional systems. The mixture of oats and peas in OA compared slightly more favourably, but still with only 47% of the yield level obtained with monocropped oats in CA1 and CA2 (Table 3).

The total fresh weight yield of potatoes was $43 \pm 1.3\,\mathrm{Mg\,ha^{-1}}$, and there was no significant difference in yield (Table 3), size distribution, or selected quality parameters between the two cropping systems with potatoes in the rotation (CA1 and CA2, data not shown). The DM content was $0.224 \pm 0.005\,\mathrm{kg\,DM\,kg\,fresh\,weight^{-1}}$, and about 93% (weight basis) of the potato tubers were saleable.

The conventional mixed dairy system (CM) had significantly larger grass clover ley yields than the organic systems, both for the 1st and the 2nd ley years (Table 4). There was no significant yield difference between the organic systems. Their total production in the two first ley years was 86% of that obtained conventionally.

The annual DM yield of the systems averaged over crops appeared to follow the same pattern between years within each production group (Figure 2).

3.2. Nutrient Concentrations of the Harvested Crops. The only differences in nutrient concentrations among the cash crops were for barley N and K (Table 3). The barley N concentration was highest in CA1, followed by CA2 and OM2 whereas OA had the lowest N concentration. The differences were smaller for K, with highest concentration in barley from OM2, and lowest in barley from CA2 and CM.

The N concentration in herbage (grass clover ley) differed significantly between systems at the 2nd cut in both ley year 1 and 2 (ley year 3 was not comparable between systems), with lower concentration in CM than in the two organic systems, which had similar concentrations (Table 4). For the concentrations of K, the tendency was opposite, at least at the first cut. Organically cropped grass clover ley had significantly lower K concentration than that of CM at the first cut of ley year 2.

Table 3: Harvested yields of cereals and potatoes and their concentrations of N, P, and K.

Crop	System	Yield[1] (kg ha^{-1})			Nutrient concentrations (g 100 g DM^{-1})								
					N			P			K		
		Mean	s.e.[2]	c[3]	Mean	s.e.	c	Mean	s.e.	c	Mean	s.e.	c
Barley	CA1	5450	152	a	2.05	0.061	a	0.40	0.025		0.39	0.008	ab
	CA2	5181	148	ab	1.99	0.054	ab	0.40	0.027		0.37	0.006	b
	OA	2148	125	d	1.68	0.034	c	0.41	0.015		0.39	0.009	ab
	CM	4865	134	b	1.85	0.034	bc	0.39	0.019		0.37	0.006	b
	OM1	3881	130	c	1.77	0.035	bc	0.40	0.019		0.40	0.003	ab
	OM2	4017	153	c	1.94	0.042	ab	0.42	0.020		0.41	0.006	a
	LSD	347			0.18			n.s.			0.02		
Oats	CA1	5866	260	a	2.07	0.050		0.38	0.020		0.37	0.006	
	CA2	5264	214	a	1.99	0.059		0.39	0.012		0.39	0.007	
	OA[4]	2618	297	b	2.30	0.155		0.43	0.012		0.59	0.012	
	LSD	944			n.s.			n.s.			n.s.		
Wheat	CA1	5696	217	a	2.25	0.047		0.40	0.023		0.40	0.002	
	CA2	5606	252	a	2.30	0.037		0.41	0.018		0.39	0.008	
	OA	2494	185	d	2.23	0.070		0.45	0.012		0.39	0.007	
	CM	4900	260	b	2.34	0.044		0.42	0.014		0.39	0.005	
	OM1	3719	187	c	2.21	0.078		0.42	0.013		0.39	0.009	
	LSD	574			n.s.			n.s.			n.s.		
Potatoes	CA1	9474	328		1.20	0.016		0.30	0.004		1.45	0.047	
	CA2	9535	349		1.22	0.038		0.28	0.001		1.42	0.058	
	LSD	n.s.			n.s.			n.s.			n.s.		

[1] Yields of cereals were corrected to a water content of 15%, whereas yields of potatoes are presented as DM.
[2] Standard error of means. Yield data were averaged for the years 2001–2010, whereas data on nutrient concentrations were averaged for the years 2006–2010.
[3] Pairwise comparisons of yields/nutrient concentrations using LSD at the 5% level, where yields/nutrient concentration of the same crop differing significantly between systems are denoted different letters. Nonsignificant comparisons are denoted n.s.
[4] In the organic system OA, oat was grown in a mixture with peas. Yields are given as the sum of the two crops, whereas nutrient concentrations are given as a weighted average (based on the DM weights of the two crops).

3.3. Soil System Nutrient Budgets

3.3.1. Nitrogen. The N input was in the range of 60–112% of the N output (Table 5). The arable system CA1 and all the organic systems had negative soil system N budgets, indicating depletion of the soil organic N content. Over the 10 years, the reductions amounted to 280, 319, 225, and 114 kg N ha^{-1} for CA1, OA, OM1, and OM2, respectively. By contrast, the budget of CA2 appeared to be balanced, whereas CM had an N surplus amounting to 198 kg N ha^{-1}.

The annual soil system N budgets (and those for P and K) were rather consistent over years for the arable systems, whereas the annual budgets of the mixed dairy systems appeared to be more positive in 2001 and 2002 than during the rest of the decade (Figure 2).

The amount of N in harvested cereals and potatoes of CA1 and CA2 corresponded to 83 and 84% of that applied, respectively (Table 5). The proportion was somewhat lower in the mixed dairy system CM (73%), whereas in the organic systems OM1 and OM2 the N removal exceeded that applied by 54 and 86%, respectively.

When comparing the sum of N lost via drainage and runoff with that applied (in fertilizer and/or cattle slurry), CA1 had the largest quotient within the arable cropping system group, and, correspondingly, OM1 had the largest quotient within the group of mixed dairy systems (Table 5).

The arable systems had the largest losses of N (via drainage and runoff) per unit of harvested N (loss-to-harvest ratio), with OA having the overall largest loss-to-harvest ratio (Table 5). The water based N losses from this system corresponded to 67% of the N in harvested products. By contrast, CA2 lost only amounts corresponding to 27% of the harvested N. The differences were much smaller within the group of mixed dairy systems, with N losses ranging from 13–24% of the harvested N.

3.3.2. Phosphorus. The P input was in the range of 8–156% of the P output (Table 6). The P budgets differed markedly between organic and conventionally managed systems. All three organic systems showed depletions of the soil P content amounting to 82, 100, and 98 kg P ha^{-1} for OA, OM1, and OM2, respectively. The conventional systems all had a calculated P surplus, particularly CA1 and CA2, which appeared to accumulate P in the same order of magnitude as the P reductions in the organic systems.

Table 4: Harvested ley yields, clover contentand concentrations of N, P, and K.

Year	System Cut	Yield (kg DM ha^{-1})			Clover[1] (%)			Nutrient concentrations (g 100 g DM^{-1}) N			P			K		
		Mean	s.e.[2]	c[3]	Mean	s.e.	c[4]	Mean	s.e.	c[5]	Mean	s.e.	c	Mean	s.e.	c
	CM	9630	335	a												
	Cut 1	4765	171		7	1.4	b	1.50	0.06		0.22	0.01		1.84	0.09	
	Cut 2	4865	243		10	2.4	B	1.48	0.09	B	0.28	0.02		1.98	0.14	
	OM1	8203	319	b												
	Cut 1	4092	218		27	2.6	a	1.55	0.09		0.22	0.01		1.69	0.06	
Yr 1	Cut 2	4111	248		45	4.7	A	2.24	0.05	A	0.31	0.02		2.10	0.10	
	OM2	8342	278	b												
	Cut 1	4099	194		27	2.4	a	1.61	0.10		0.24	0.016		1.68	0.11	
	Cut 2	4243	228		40	5.0	A	2.16	0.16	A	0.32	0.01		2.02	0.08	
	LSD[6]	710			12			n.s.			n.s.			n.s.		
	LSD[7]				11			0.45			n.s.			n.s.		
	CM	11021	304	a												
	Cut 1	6269	180		11	2.1	b	1.57	0.04		0.24	0.01		1.99	0.08	a
	Cut 2	4752	233		11	2.1	A	1.46	0.06	B	0.27	0.02		1.82	0.09	
	OM1	9542	264	b												
Yr 2	Cut 1	5453	188		34	2.5	a	1.64	0.11		0.22	0.01		1.65	0.06	b
	Cut 2	4089	201		42	3.6	B	2.19	0.04	A	0.30	0.01		1.76	0.07	
	OM2	9586	312	b												
	Cut 1	5399	209		30	2.9	a	1.64	0.10		0.25	0.01		1.69	0.06	b
	Cut 2	4187	189		36	2.5	B	2.22	0.13	A	0.30	0.01		1.84	0.05	
	LSD[6]	892			10			n.s.			n.s.			0.25		
	LSD[7]				13			0.51			n.s.			n.s.		
	OM2	8651														
Yr 3	Cut 1	5499	169		19	2.3		1.52	0.05		0.24	0.01		1.60	0.08	
	Cut 2	3151	217		24	2.8		2.11	0.06		0.31	0.01		1.65	0.09	

[1]The proportion of legumes was determined visually before harvest.
[2]Standard error of means. Yield data were averaged for the years 2001–2010, whereas data on nutrient concentrations were averaged for the years 2006–2010.
[3]Pairwise comparisons of yields (sum of two cuts) using LSD at the 5% level, where yields of the same crop not differing significantly between systems are denoted the same letter.
[4]Pairwise comparisons of clover content separate for each cut, where clover contents of the first cut not differing significantly between systems are denoted the same letter, whereas clover contents of the second cut not differing significantly between systems are denoted the same capital letter.
[5]Pairwise comparisons of nutrient concentrations separate for each cut, where concentrations of the first cut not differing significantly between systems are denoted the same letter, whereas concentrations of the second cut not differing significantly between systems are denoted the same capital letter.
[6]LSD values at 5% level for the total yields (sum of two cuts) and that for clover content and nutrient concentrations of first cut.
[7]LSD values at 5% level for clover content and nutrient concentrations of the second cut.

The P removal at harvest amounted to 70 and 72% of that applied in CA1 and CA2, respectively (Table 6). This percentage was somewhat larger for CM (82%), and very high for the organic systems OM1 (217%) and OM2 (182%), indicating soil P mining in these systems.

As the system differences in terms of P losses in drainage and runoff were not statistically significant, quotients between these losses and applied or harvested P were not calculated.

3.3.3. Potassium. The K input was in the range of 23–200% of the K output (Table 7). The pattern of the K budgets was similar to that for P; all the organic systems had calculated K deficits, whereas all the conventional systems appeared to accumulate K.

The amount of harvested K corresponded to 53% of that applied in CA1 and CA2 (Table 7). This proportion was somewhat higher in the mixed dairy system CM (66%). In the organic systems OM1 and OM2 the K removal

Table 5: Measured and estimated nitrogen flows (kg N ha^{-1} $year^{-1}$) and selected quotients (kg N kg N^{-1}) in the cropping systems at Apelsvoll, mean 2001–2010[1].

N flow	Cropping system[2]					
	CA1	CA2	OA	CM	OM1	OM2
Fertilizer applied	124.9	118.6	0.0	87.2	0.0	0.0
Cattle slurry applied	0.0	0.0	0.0	68.8	57.5	82.4
Dry atmospheric depositions	2.7	2.7	2.7	2.7	2.7	2.7
Wet atmospheric depositions	7.2	7.2	7.2	7.2	7.2	7.2
N fixation by legumes	0.0	0.0	34.0	14.3	57.4	59.1
N in seeds	12.9	13.1	4.4	2.5	2.6	1.1
Sum field N input	147.7	141.6	48.3	182.6	127.5	152.6
Harvest[3]	117.4	99.7	43.5	118.4	111.7	130.5
NH_3 and NO_x from applied fertilizer[4]	12.5	11.9	0.0	8.7	0.0	0.0
NH_3 and NO_x from applied cattle slurry[4,5]	0.0	0.0	6.7	13.8	11.5	16.5
Direct N_2O losses[4]	1.6	1.7	0.7	2.1	1.0	1.1
Drainage and runoff N	44.2	26.7	29.3	19.9	25.6	15.9
Sum field N output	175.7	140.0	80.2	162.8	149.9	164.0
Field N budget	−28.0	1.6	−31.9	19.8	−22.5	−11.4
Harvested N[6]/applied N	0.83	0.84	—	0.73	1.86	1.54
Drainage + runoff N /applied N	0.35	0.23	—	0.13	0.45	0.19
Drainage + runoff N /harvested N[6]	0.43	0.27	0.67	0.17	0.24	0.13

[1] Timestep used is the agrohydrological year (May–April), thus, covering the period May 2001–April 2011.
[2] Each system covers 0.18 ha and consists of four rotation plots á 0.045 ha.
[3] Including the N removed with straw. Harvested grass-clover N was reduced by 10% to account for likely harvest-related losses under practical conditions (see Section 2).
[4] Calculated according to [24].
[5] Volatilization of NH_3-N from mulched clover grass is not accounted for in [24], although it may be substantial [25]. It was, therefore, calculated in accordance with Korsaeth [6].
[6] Not including N removed with straw.

Table 6: Measured and estimated phosphorus flows (kg P ha^{-1} $year^{-1}$) and selected quotients (kg P kg P^{-1}) in the cropping systems at Apelsvoll, mean 2001–2010[1].

P flow	Cropping system[2]					
	CA1	CA2	OA	CM	OM1	OM2
Fertilizer	29.8	29.5	0.0	15.0	0.0	0.0
Cattle slurry applied	0.0	0.0	0.0	9.6	7.9	10.9
P in seeds	2.6	2.7	0.7	0.5	0.5	0.2
Sum field P input	32.5	32.1	0.7	25.1	8.4	11.1
Harvested P[3]	24.1	20.5	8.8	21.0	18.2	20.7
Drainage and runoff P	0.2	0.1	0.2	0.3	0.2	0.2
Sum field P output	24.3	20.7	8.9	21.2	18.4	20.9
Field P budget	8.2	11.5	−8.2	3.9	−10.0	−9.8
Harvested P[4]/applied P	0.72	0.70		0.82	2.17	1.82

[1] Timestep used is the agrohydrological year (May–April), thus, covering the period May 2001–April 2011.
[2] Each system covers 0.18 ha, and consists of four rotation plots á 0.045 ha.
[3] Including the P removed with straw. Harvested grass-clover P was reduced by 10% to account for likely harvest-related losses under practical conditions (see Section 2).
[4] Not including P removed with straw.

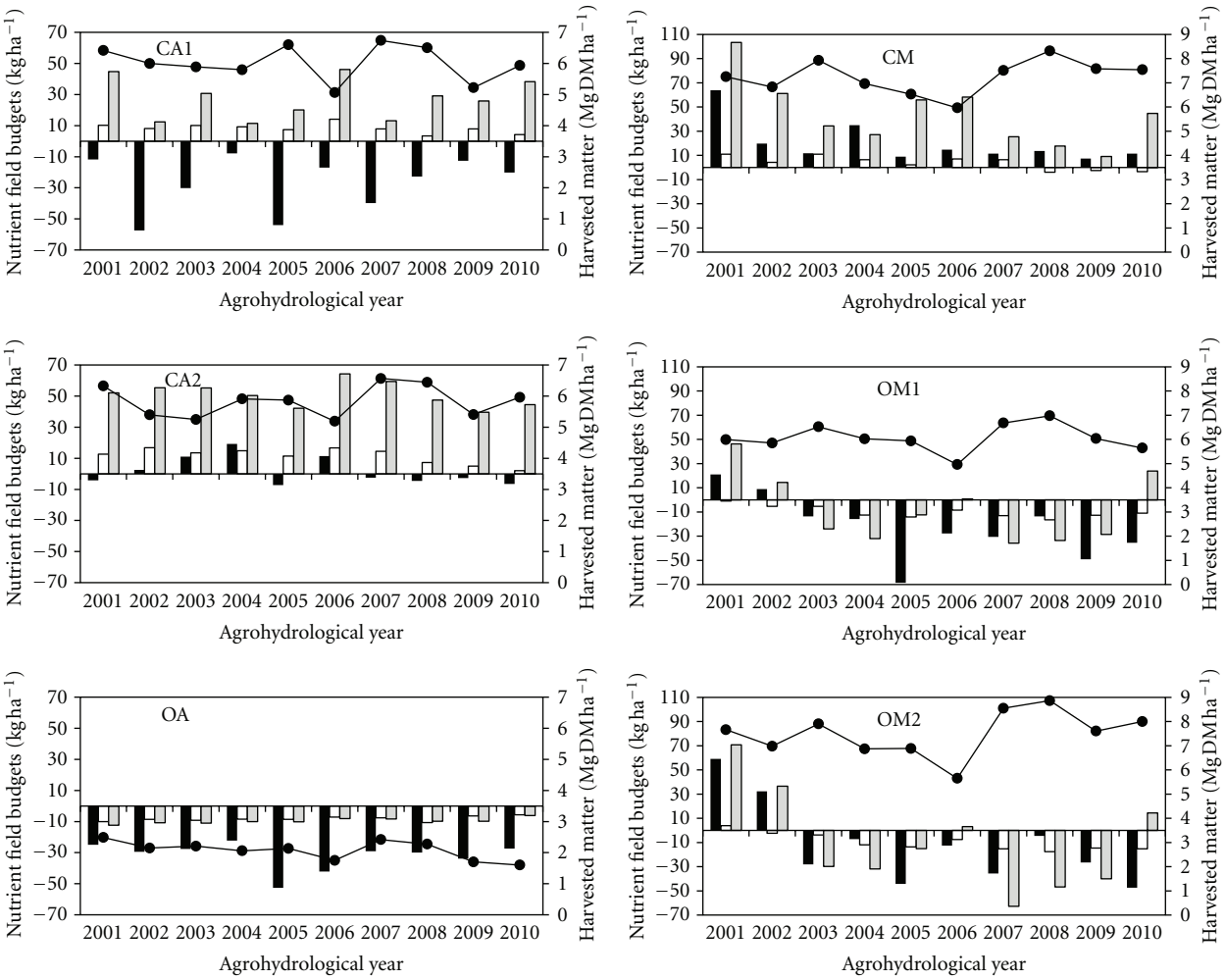


Figure 2: Annual (agrohydrological year) harvested dry matter yields averaged over crops (lines) (right y-axis) and soil system budgets for N (black bars), P (white bars), and K (grey bars) (left y-axis) for CA1 (conventional arable: upper left subplot), CA2 (conventional arable, environmentally sound: middle left subplot), OA (organic arable: lower left subplot), CM (conventional mixed dairy: upper right subplot), OM1 (organic mixed dairy: middle right subplot), and OM2 (organic mixed dairy with 75% clover ley: lower right subplot).

at harvest was only slightly larger than the amount of K applied.

The sum of K lost via drainage and runoff was 7% and 5% of that applied in CA1 and CA2, respectively (Table 7). The corresponding percentages for the mixed dairy systems were less, ranging from 2–4%.

The loss-to-harvest ratios for K followed a similar pattern as for N, but at a lower level (Table 7). The arable systems had the largest ratios, with the highest value calculated for OA. The differences were much smaller within the group of mixed dairy systems, with N losses ranging from 2–4% of the harvested K.

3.4. Changes in Topsoil Nutrient Content

3.4.1. Nitrogen. The ignition loss did not differ significantly between 1999 and 2009, although the measurements appeared to be at a lower level in 2009 (Table 8, Figure 3(a)). The tendency of reduction was particularly strong for CA1 ($P = 0.054$) and OM2 ($P = 0.088$). Although the measured changes in ignition loss over time did not support the calculated N budgets substantially, the differences between systems in terms of ignition loss measured in 2009 reflected the N budgets very well (Figure 4(a)). A ranking of the systems in terms of measured ignition loss levels in 2009 was almost identical with a ranking based on the calculated N budgets.

3.4.2. Phosphorus. P-AL changed significantly in four of the six systems during the period 1996–2009 (Table 8, Figure 3(b)). The measurements in 2009 showed the same pattern of differences between the systems as did those in 2003, but with a greater magnitude. The measured differences in P-AL reflected the calculated P budgets well, although the measured declines in OM1 and OM2 were not significant (Table 8). There was a strong linear relationship

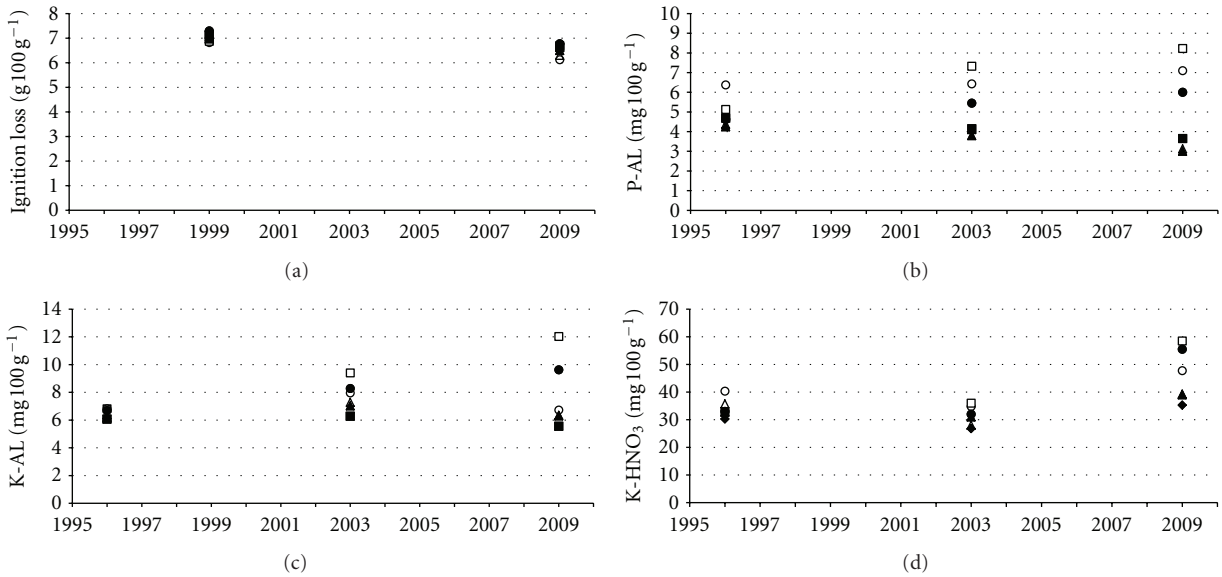


Figure 3: Topsoil (0–25 cm) content of ignition loss (subplot A), P-AL (subplot B), K-AL (subplot C), and K-HNO_3 (subplot D) for the six cropping systems CA1 (conventional arable: ◯), CA2 (conventional arable, environmentally sound: □), OA (organic arable: △), CM (conventional mixed dairy: •), OM1 (organic mixed dairy: ▲), and OM2 (organic mixed dairy with 75% clover ley: ■).

Table 7: Measured and estimated potassium flows (kg K ha^{-1} $year^{-1}$) and selected quotients (kg K kg K^{-1}) in the cropping systems at Apelsvoll, mean 2001–2010[1].

K flow	Cropping system[2]					
	CA1	CA2	OA	CM	OM1	OM2
Fertilizer	90.7	88.4	0.0	56.9	0.0	0.0
Cattle slurry applied	0.0	0.0	0.0	89.2	74.7	101.8
Wet atmospheric depositions	2.1	2.1	2.1	2.1	2.1	2.1
K in seeds	11.4	11.4	0.9	0.6	0.6	0.3
Sum field K input	104.2	101.9	3.0	148.8	77.4	104.2
Harvested K[3]	70.4	46.8	9.3	101.4	82.2	112.0
Drainage and runoff K	6.6	4.1	3.3	3.6	3.3	2.3
Sum field K output	77.0	50.9	12.6	105.1	85.4	114.3
Field K budget	27.2	51.0	−9.7	43.7	−8.0	−10.1
Harvested K[4]/applied K	0.53	0.53		0.66	1.04	1.05
Drainage + runoff K/applied K	0.07	0.05		0.03	0.04	0.02
Drainage + runoff K/harvested K[4]	0.14	0.09	0.36	0.04	0.04	0.02

[1] Timestep used is the agrohydrological year (May–April), thus covering the period May 2001–April 2011.
[2] Each system covers 0.18 ha and consists of four rotation plots á 0.045 ha.
[3] Including the K removed with straw. Harvested grass-clover K was reduced by 10% to account for likely harvest-related losses under practical conditions (see Section 2).
[4] Not including K removed with straw.

between P-AL measured in 2009 and the calculated P budgets of the systems (Figure 4(b)).

3.4.3. Potassium. K-AL followed a pattern similar to that of P-AL over the period 1996–2009, with increasing differences between systems over time (Figure 3(c)). The only significant differences between the measurements in 1996 and 2009 were found in CA2 and CM, which both showed increased levels of K-AL in 2009 (Table 8). The calculated K-deficits in the organic systems, and the K-surplus of CA1, could, thus, not be supported by the measured differences in K-AL over time. The differences between systems in terms of K-AL measured in 2009 corresponded, however, very well with the calculated K budgets (Figure 4(c)).

There was a significant increase in the topsoil content of K-HNO_3 from 1996 to 2009 in all systems (Table 8, Figure 3(d)). The rate of change decreased in the order CA2>CM> CA1>OM1>OM2>OA, a ranking which

Table 8: Measured changes in selected chemical properties in topsoil (0–25 cm) for the last decade.

Cropping system/year	CA1	CA2	OA	CM	OM1	OM2
			Ignition loss (g 100 g^{-1})			
1999	6.83	6.88	7.05	7.30	6.98	7.13
2009	6.13	6.70	6.30	6.78	6.50	6.63
Change (2009–1999)	−0.70	−0.18	−0.75	−0.53	−0.48	−0.50
P value[1]	0.054$^{n.s.}$	0.608$^{n.s.}$	0.204$^{n.s.}$	0.152$^{n.s.}$	0.265$^{n.s.}$	0.088$^{n.s.}$
			P-AL (mg 100 g^{-1})			
1996	5.38	4.00	4.25	4.00	3.75	3.88
2009	7.10	8.23	3.13	6.00	3.00	3.65
Change (2009–1996)	1.73	4.23	−1.13	2.00	−0.75	−0.23
P value	0.029	0.015	0.036	0.001	0.102$^{n.s.}$	0.409$^{n.s.}$
			K-AL (mg 100 g^{-1})			
1996	6.75	6.75	6.13	7.13	6.13	6.00
2009	6.73	12.0	6.35	9.63	6.30	5.55
Change (2009–1996)	−0.03	5.28	0.23	2.50	0.18	−0.45
P value	0.942$^{n.s.}$	0.043	0.362$^{n.s.}$	0.022	0.188$^{n.s.}$	0.517$^{n.s.}$
			K-HNO_3 (mg 100 g^{-1})			
1996	31.1	28.3	30.9	28.6	29.1	26.0
2009	47.8	58.5	39.3	55.5	39.0	35.3
Change (2009−1996)	16.6	30.3	8.38	26.9	9.88	9.25
P value	0.012	0.007	0.010	0.003	0.001	0.037

[1] Level of significance for the pairwise *t*-test of differences. Nonsignificance at the 5% level is denoted$^{n.s}$.

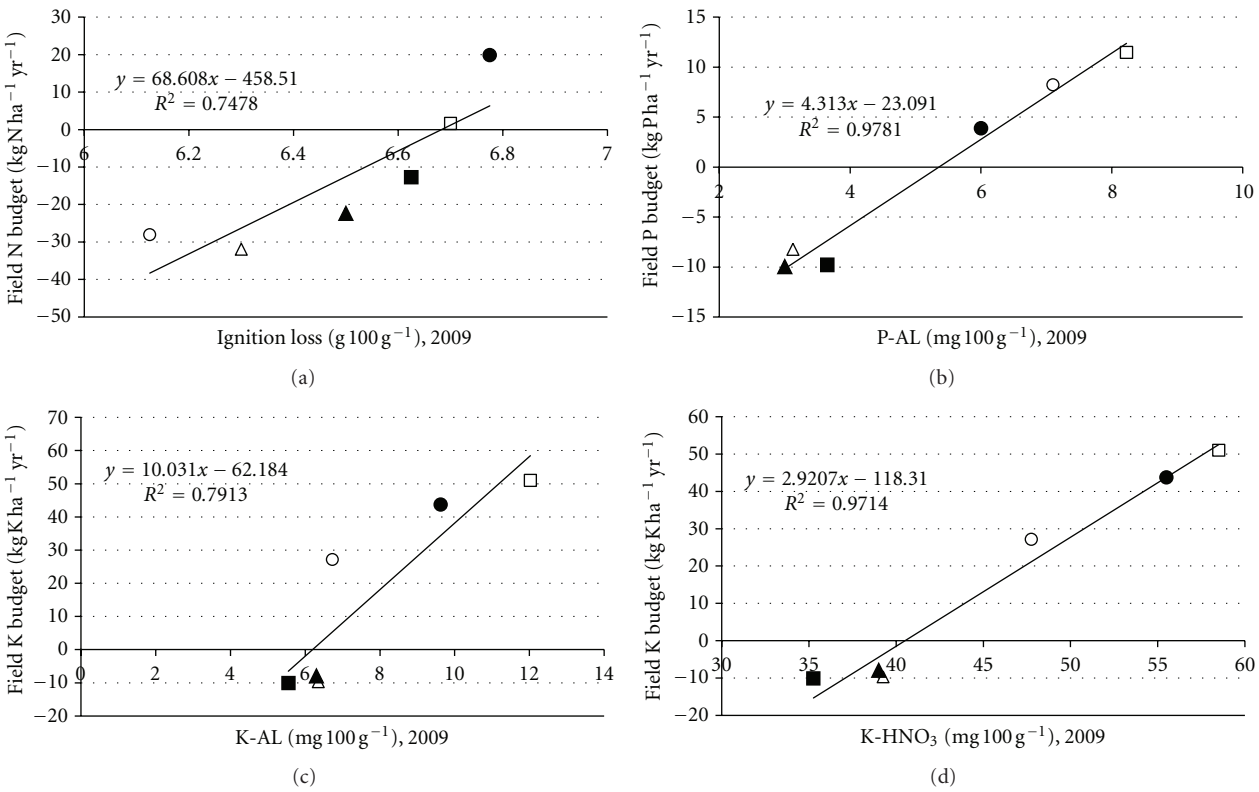


Figure 4: Calculated soil system budgets of N, P, and K for the years 2001–2010 plotted, respectively, against ignition loss (subplot A), P-AL (subplot B), K-AL (subplot C), and K-HNO_3 (subplot D) for the six cropping systems CA1 (conventional arable: ○), CA2 (conventional arable, environmentally sound: □), OA (organic arable: △), CM (conventional mixed dairy: •), OM1 (organic mixed dairy: ▲), and OM2 (organic mixed dairy with 75% clover ley: ■).

corresponded well with a ranking of the calculated K budgets. The relation between the soil measurements in 2009 and the calculated K budgets was even stronger for K-HNO_3 than it was for K-AL (Figure 4(d)).

4. Discussion

4.1. Yields. Both cereal yield levels and relative yield differences between systems followed the same pattern for the whole period 2001–2010 as for the years 2001–2004, which were discussed by Korsaeth [6]. Briefly, the yield differences were larger amongst the arable systems than amongst the mixed dairy systems. The low yields in OA may be explained by P and K deficits, as indicated by the negative P and K budgets and the significant reduction in plant available topsoil P, and due to the lack of chemical plant protection. More foliar diseases were generally observed on cereals in the organic systems than in the other systems (data not shown).

The lack of significant differences in measured potato parameters between the two cropping systems (CA1 and CA2) is partly in accordance with the findings of Riley and Ekeberg [27], who compared spring and autumn ploughing at different depths (10, 20, and 30 cm) with tine harrowing only in spring, on the same soil type at a nearby location. They found the same potato fresh weight yields in all treatments, but the tuber dry matter concentration was significantly lower in potatoes grown without ploughing.

Among the mixed dairy systems, the organically grown cereal yields were also lower than those grown conventionally. However, due to much smaller differences in nutrient regime, the differences between mixed dairy systems were less than those between arable systems. The most likely reasons for inferior yields in the organic mixed dairy systems relative to the conventional mixed dairy system were, as for the arable systems, suboptimal nutrition and a lack of plant protection.

The yield pattern of grass clover ley was also unchanged in 2005–10 compared with the first four years of the decade [6]. The high yields of the organic leys may be explained partly by their N fixation, which was estimated to be much greater in the organic leys than in the conventional leys. This was a result of the significantly higher proportions of clover in the organic leys and less suppression of N fixation by the use of inorganic N fertilizer. The reduced grass clover yields in the 3rd ley year of OM2 may be due to the reduced proportion of clover compared with the first two ley years.

The long period of active nutrient uptake by grass and clover may also partly explain the relatively high organic ley yields. A longer uptake period increases the utilization of less readily available nutrients (e.g., nutrients in organic form), since the mineralization of such nutrients occurs throughout the cropping season. Smaller yield differences between organic and conventional cropping for grass clover ley than for cereals have been reported previously for this experiment [28].

In a review of a number of Swedish field studies, Bergström et al. [29] reported that crop yields in organic rotations were reduced by 20 to 80%, compared with the same crops in conventional rotations. These authors explained this in terms of higher N deficiency, more weed competition, and greater infestation of crop diseases in the organic systems.

4.2. Soil System Nutrient Budgets and Nutrient Concentrations of Topsoil and Crops

4.2.1. Nitrogen. The large calculated deficits found for the arable systems CA1 and OA in the first part of the decade [6] were sustained. The suggested net soil N depletion corresponded to a relative decay rate of the topsoil (0–25 cm) N content of 0.4% yr^{-1}. This corresponds well with Riley and Bakkegard [30], who compared soil samples taken in 1991 and in 2001 from 291 arable fields located throughout southeast Norway. They found that the percentage relative decline rate of SOM was approximately one tenth of the initial percentage of organic matter in soil over the decade. In the present study, there was a strong tendency ($P = 0.054$) towards reduced ignition loss for CA1 in 2009 compared with 1999; but this was not the case for OA.

Comparing conventional and organic cropping systems in a pipe-drained plot experiment in Sweden, Torstensson et al. [31] also reported an N deficit ($-18\,kg\,N\,ha^{-1}\,yr^{-1}$, not including denitrification, N in seeds and atmospheric depositions) for a conventional arable rotation (CON, barley-oat-spring wheat-barley-oat-potato) comparable with CA1. They tested additionally an arable organic system with green manure as the only N source (OGM, oat-green manure-spring wheat-oat-green manure-potato), comparable with OA and found a positive soil system N budget ($13\,kg\,N\,ha^{-1}\,yr^{-1}$) in contrast to the present findings. In the experiment of Torstensson et al. [31], the proportion of green manure was, however, larger than in our case (33% versus 25%), which resulted in $37\,kg\,N\,ha^{-1}\,yr^{-1}$ more N fixation and $20\,kg\,N\,ha^{-1}\,yr^{-1}$ less harvested N, compared with OA.

The only arable system which appeared to have a balanced N budget was CA2. This may mainly be explained by its comparative low leaching and runoff N losses, which represent the main difference between CA1 and CA2 with regard to N flows. The findings indicate that reduced tillage counteracted soil N mining, which is a commonly reported result (e.g., [32]). Another factor which may have contributed to prevent soil N mining in CA2 is that straw was not removed. Straw incorporation has a well-known positive effect on the soil organic N content (e.g., [33]).

The conventional mixed dairy system CM had a calculated N surplus over the decade, which indicates that the system probably increased its soil organic matter content. Conservation of or an increase in soil N has also been reported for other rotations containing pasture or ley receiving organic N on relatively N-rich ($>2.0\,g\,kg^{-1}$) soils [32, 33]. The opposite was found in the organic system OM1, with the same crop rotation and tillage as CM, but with a calculated N deficit of $23\,kg\,N\,ha^{-1}\,yr^{-1}$, indicating that the relatively high production has been maintained at the cost of the soil organic N pool. Similarly, Steinshamn et al. [34] reported an annual N deficit of $16\,kg\,N\,ha^{-1}$, not including N leaching and denitrification, at the field level in an organic

crop rotation with 50% grassland (barley, forage rape + Italian ryegrass, oats + peas, 3-year grassland).

The slightly negative N budget of OM2 ($-16\,\mathrm{kg\,N\,ha^{-1}}$) shows that a high proportion of grass clover ley in the rotation does not guarantee a balanced N budget. By comparison, Syväsalo et al. [35] reported an even larger N deficit ($-31\,\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, not including ammonia emissions, deposition, or N in seeds) in an organic grass clover ley receiving $130\,\mathrm{kg\,N\,ha^{-1}}$ in cattle slurry. An extra year of grass clover ley instead of wheat in the OM2 rotation resulted in the largest calculated amounts of available cattle slurry for this system ($82\,\mathrm{kg\,N\,ha^{-1}}$), but the N fixation was apparently much less effective in the additional ley year than in the previous two ley years. The estimated N fixation of the third ley year was only $43\,\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$, compared with 82 and 88 kg in the 1st and 2nd ley year, respectively.

The annual nutrient budgets of the mixed dairy systems showed more positive figures for 2001 and 2002 compared with the following eight years. This was a result of too high yield expectations when calculating the initial amount of slurry available for these systems in 2001. The number of cows which each farming system could sustain, and, thus, the amount of slurry available for the crops, was calculated from the average total available feed in the system during the previous three years (sliding mean). The initialization problem was, thus, gradually levelled out.

The calculated changes in soil N, that is, misbalanced soil system N budgets, were in general poorly supported by the measured changes in topsoil ignition loss since 1999, which were all nonsignificant. Considering the large differences in the measured N flows between the systems, it is very unlikely that the SOM level of 1999 would have been sustained in all systems over the following decade. One explanation of the mismatch could be an over- or underestimation of the estimated gaseous N losses, which were the most uncertain N flows in the calculated budgets. If these losses were largely overestimated, the calculated deficits of CA1, OA, OM1, and OM2 would have been reduced, but the calculated surplus of OM would have increased, and *vice versa*. Another explanation could be that some organic matter has been transported from topsoil to subsoil. Such a translocation of organic matter may have taken place, but it seems unlikely that this process has diverged significantly between systems with more or less the same crop rotation.

The relative differences between the systems in 2009 matched the calculated soil system N budgets much better than the differences found between sampling times, indicating that the 1999 data may include some random variation. The relation between ignition losses in 2009 and the N budgets indicated that ignition loss would equilibrate at $67\,\mathrm{g\,kg^{-1}}$ with a balanced N budget. This corresponds to an SOM content of $47\,\mathrm{g\,kg^{-1}}$, calculated with a pedotransfer function developed solely for this site (SOM = 0.81 × ignition loss (%) − 0.038 × clay (%) − 0.70), [21]).

The significant differences between systems in terms of N concentrations of the harvested crops were few and reflected poorly the differences in fertilization regime and soil system N budgets. Greenwood et al. [36] developed a model linking N concentration in plant DM to growth rate and to plant mass per unit area. They found that subcritical values of N concentration during growth affected the growth rate. In order to define whether any of the crops in the present study were N limited, measurements of N concentration during crop growth and not at harvest would, thus, have been required.

4.2.2. Phosphorus. Conventional arable cropping appeared to give relatively large surpluses of P. In the past, more emphasis, in Norway, was placed on adjusting N fertilizer rates to crop requirements than to adjusting P rates. In 2007–2008 there was a change in the fertilizer recommendations for P in Norway, with a reduction of approx. 25% for cereals and grasses and 30% for potatoes. This prompted the fertilizer market leader in Norway (market share > 90%) to increase the N : P-ratio of their most used compound fertilizers, from 5.3 to 7.3 for cereals and from 2.2 to 3.0 for potatoes, thus reducing the amounts of P for a given amount of N.

The organic arable system did not receive any P, and the production was thus entirely dependent on P supply from the soil. The negative soil system budget of $8\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$ was the same as that found in a stockless organic farm (red clover-winter wheat-spring beans-spring cereal) in the UK [15]. The results show that in stockless organic farming systems, some form of external P addition becomes unavoidable sooner or later (depending on the size of the initial P pool and the ability of the soil to deliver plant available P). Berry et al. [15] showed that a system comparable with OA was almost in balance in terms of P when it received rock phosphate. From a resource economics point of view, it is questionable; however, whether the use of untreated rock phosphate is a good strategy, considering its low plant availability [37]. An alternative could be to use organic waste, such as biogas residue from household waste, which has been shown to be a valuable and inexpensive source of plant nutrients [38].

The conventional mixed dairy system had a field surplus of almost $4\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$, suggesting an unnecessary use of a limited resource. This appears to be no exception. In a comparable farming system in northern Sweden, Bengtsson et al. [39] reported a P surplus of $5\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$. The problem seems to be even worse on conventional dairy farms, that is, those with no or a very low proportion of arable crops. The P surplus on such farms is assumed to vary from 10 to $72\,\mathrm{kg\,ha^{-1}}$ in Europe (Pfimlin et al. 2006, cited by [40]).

The organic mixed dairy systems produced at the cost of their indigenous soil P pool, with a total deficit of about $10\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$. This was not surprising, considering that there was no P input to these systems, except for that in seeds. Even when some feed is purchased, P deficits are commonly reported. Berry et al. [15] reported soil surface budgets (i.e., maximum root depth as lower boundary) of $-3\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$ in a mixed dairy system in the UK (ley cropped in 3 out of 5 years, farm number 3). Steinshamn et al. [34] found a deficit between inputs and produce (losses not considered) in an organic dairy farming system in Norway of $6.3\,\mathrm{kg\,P\,ha^{-1}\,yr^{-1}}$.

All the significant changes in topsoil P-AL from 1996 to 2009 were in the same direction as the corresponding calculated soil system P budgets. The budget calculations were also supported by the very strong relation between the topsoil level of AL-extractable P in 2009 and the calculated soil system budget. The results clearly show the effect of both overfertilization and suboptimal P fertilization on the plant availability of P in soil.

In the present study, P losses to subsoil, that is, below root depth, were considered to be negligible. On a comparable soil in the long-term fertilizer trials at Møystad in S.E. Norway, Riley [41] found no effect of P treatment (no P addition, P in mineral fertilizer, or P in animal manure) below 40 cm depth. In contrast, Verloop et al. [40], studying intensive dairy farming systems, found that some topsoil P was transported to subsoil and that the P accumulation in the deeper layers practically equalled the P depletion in the upper topsoil. Their experiment was, however, run on a light sandy soil, characterized by a 0.3 m anthropogenic topsoil overlaying a layer of yellow sand hardly penetrable by roots.

With an assumed balanced P budget, P-AL appeared to equilibrate at 54 mg P kg^{-1}, a level which is normally considered adequate for optimum growth. In the long-term trials at Møystad, Ekeberg and Riley [8] also found a strong relationship between topsoil P-AL and P balance (P applied via fertilizer and/or farmyard manure minus P removed by harvest) for the period 1922–1983. They reported that P-AL equilibrated at 25–30 mg P kg^{-1} when the application of P equalled the removal of P by crops. This equilibrium point rose to about 40 mg P kg^{-1} in the period 1983–2003 (H. Riley, personal communication). Experiences from an intensive dairy farm in The Netherlands, where P-equilibrium fertilization (i.e., balancing P inputs via fertilizer and manure with P in crop products) is performed, has shown that the soil available P-status differs between crop rotations.

The P content of the crops did not differ between the systems, and the herbage concentrations of P were in the range of 0.2–0.3%, a level which is regarded as adequate [42]. In comparison, Mathews et al. [43] considered 0.2–0.34% P to be the critical concentration for cool-season grasses, that is, a concentration level below which a 10% yield drop is expected. It, thus, appears that the grass clover growth was not P limited in the organic mixed dairy systems, in spite of the continuous soil P depletion of these systems.

4.2.3. Potassium. The potassium soil system budgets showed that the conventional arable systems had unnecessarily high levels of K fertilization. As for P, the change in the compound fertilizers (from 2009) also altered the amount of K relative to N, with a N : K-ratio increasing from 2.1 to 2.2 for cereal fertilizer and from 0.65 to 0.67 for potato fertilizer. Although this change has some importance for practical farming, it did not influence the results presented here.

In contrast to the present findings, Torstensson et al. [31] reported a small K deficit (-3 kg K ha^{-1} yr^{-1}) in a 6 year conventional rotation with five years of spring cereals and one year with potatoes, comparable with CA1. Additionally, they found a large deficit (-28 kg K ha^{-1} yr^{-1}) in a similar crop rotation but with ryegrass grown as a catch crop after each main crop, comparable with CA2. The contrasting findings of Torstensson et al. [31] may largely be explained by the K leaching, which were 5–7 times larger in their experiment than in the present study. The literature appears inconclusive, however, when it comes to K-budgets for conventional arable cropping systems. This was well illustrated by Heming [44], who studied a large number of fields in southern England, reporting K budgets (K applied in fertilizer minus K in crop) ranging from -40 to $+70$ kg K ha^{-1} yr^{-1}.

The organic arable system OA had a calculated K-deficit. Stockless organic systems without any form of K application are bound to result in negative K budgets, as has been commonly reported (e.g., [15, 31, 45]). Interestingly, the calculated deficit in OA was of the same magnitude as the calculated deficits of the two organic mixed dairy systems (OM1 and OM2). The relatively large inputs of K in applied slurry in the mixed dairy systems, thus, appeared to be more than outweighed by large K export via harvested material. Reviewing a range of cropping systems in northern Europe, Öborn et al. [46] summarized that negative farm gate and soil surface K budgets are especially common in organic farming. This was supported by Øgaard and Hansen [47], who found negative K budgets in grassland fields on 23 out of 26 organic farms in Norway. In a study of three long-term field experiments with mixed cropping systems (six year rotations with 2/6 or 3/6 ley) on sandy loam soils over the 18 year period 1997–2004, Andrist-Rangel et al. [48] reported negative K budgets (input minus crop offtake) for organic systems in the range of -22 to -75 kg K ha^{-1} yr^{-1}. They also found, however, that a conventional mixed cropping system had negative field budgets during the same period, ranging from -21 to -60 K ha^{-1} yr^{-1}, in contrast to the conventional mixed cropping system of the present study (CM), which had a calculated surplus of 44 kg K ha^{-1} yr^{-1}. On the other hand, Bengtsson et al. [39] reported a K surplus of 39 kg K ha^{-1} yr^{-1} in a comparable conventional system in northern Sweden.

The only significant changes in measured topsoil K-AL were the increase in CA2 and CM, in correspondence with their large calculated field K surpluses. The relative differences between systems in terms of calculated K budgets were well reflected in the relative differences between changes of K-HNO_3 and there was a clear relation between K budgets and measured levels of both K-AL and K-HNO_3 in 2009. The relatively large, negative soil system K budgets for the organic systems were, however, not reflected by the soil measurements. Could it be that the soil system K budgets were largely underestimated (i.e., underestimated inputs and/or overestimated outputs)?

Errors in K mass budgets are not an uncommon phenomenon (e.g., [46]). It seems unlikely, however, that such errors alone could explain the lack of fit between the calculated K budgets and the soil K measurements. For example, the unfertilized system OA would need a budget correction of about 10 kg K ha^{-1} yr^{-1}, in order to achieve a balanced soil system K budget,

which would reflect the K-AL data (i.e., no significant difference between 1996 and 1999). Such a change would be of the same magnitude as the entire (current) calculated K off-take by harvest, and it would correspond to more than 330% of the total K input or 300% of the K measured in drainage and runoff. Moreover, the calculated depletions of K in the organic systems are well supported by relevant comparisons in the literature, as discussed above.

Another, and more likely, explanation of the poor relation between soil system K budgets and changes in soil K measurements over time, could be that the weathering of primary minerals has released K in amounts compensating for the calculated K depletion. Annual weathering rates for different Norwegian and Swedish soils have been estimated to range from 3 to 82 kg K ha^{-1} [49]. Øgaard and Hansen [47], looking at K uptake and requirement in organic grassland farming in Norway, found that K uptake from reserve K, that is, K located in the interlayers of the sheet silicates in the soil, was positively related to the acid soluble K content (i.e., K-HNO_3 minus K-AL) of the soil. In the present study, acid soluble K was in the range of 20–25 mg K 100 g^{-1} (1996 values), which corresponds to a potential uptake of reserve K between 30 and 110 kg K ha^{-1} yr^{-1}, based on the findings of Øgaard and Hansen [47].

The considerations above relate to the topsoil. Additionally, there may have been substantial K uptake from the subsoil, particularly in systems with negative soil system budgets. Experiments using a K/Rb isotope dilution method on loess-parabrown soils in N. Germany have showed that the subsoil (>30 cm depth) supplied between 9 and 70% of the total K uptake in spring wheat [50].

The data give no reason to assume that the availability of K was limiting for yield formation. In the future, separate analyses of grasses and legumes will nevertheless be performed, to enable a better assessment of the critical nutrient levels of the forage crops.

4.3. Implications for Future Cropping Systems. Agronomic practices affect the balance between utilization and removal of plant available nutrients in soil. Where the reserves of potentially available nutrients (e.g., organic N, organic/fixed P, and fixed K) are large, low input farming systems may maintain productivity at the cost of a gradual decline of the soil nutrient pools. It should be emphasised, however, that soil P mining is to be considered as an equally serious depletion of a limited resource as the mining of rock P for producing mineral fertilizer. In future agricultural systems, one main challenge is to close the P cycle. Today, a large share of harvested P ends up in persistent chemical compounds due to the widespread use of coagulants (e.g., aluminium and iron) to remove P from sewage [51]. One step in the right direction is to use biogas residue from household waste, as mentioned above, as a nutrient source. Such a practice has from 2011 been integrated into the management of the organic OA system of the current study, in order to improve its nutrient balance.

Nitrogen may be fixed biologically in sufficient amounts in clover ley dominated systems. Systems which have an N input based on green manure appear, however, to be very area-demanding, and the practice of leaving large amounts of N-rich material unharvested leads to severe risks for N losses to the environment [6]. The present study shows that a good alternative is to use moderate amounts of fertilizer, combined with steps to reduce the risk of losses, such as the use of spring tillage, catch crops, and split application of fertilizer. Such a system (i.e., CA2) has previously been shown to have the lowest ratio of N loss to food production [6].

Potassium is often given relatively little attention when studying agricultural systems, probably due to the absence of direct, negative environmental impacts associated with K losses. Moreover, many soils show a remarkable capacity for replacing removed plant available K through weathering and, thus, counteracting negative effects (i.e., plant K deficiency) when soil K mining occurs, as shown in this study. In the long term, however, all systems require a balanced supply of K, which implies that most organic farmers today need to consider means to enhance the K input to their farms. This study highlights also the need for balancing K, along with N and P, in conventional systems, as there is a risk for an oversupply of K in these systems. Oversupply means that the indirect environmental impacts, that is, those associated with the manufacturing and transport of K fertilizer, are unnecessarily increased.

To perform a thorough evaluation of agricultural systems in terms of environmental impact, such as their global warming potential, all processes governing the manufacturing and transport of input factors (not only those for fertilizer) should be considered. This points toward the need for life cycle assessment (LCA) studies, which use a holistic approach for the evaluation of a system's environmental impacts (e.g., [52]). An LCA study of the systems included in the present paper is currently in preparation.

5. Conclusions

Differences in cereal yields between organically and conventionally managed systems are larger in arable rotations than those in mixed cropping systems including ley and animal husbandry, mainly due to improved nutrition through N fixation in the ley and the availability of animal manure in the latter group.

The yield differences for grass clover leys are smaller between the two management types (i.e., organic versus conventional) than those of cereals, due to lower disease pressure in grasses and better nutrition, resulting from N fixation and better utilization of mineralised nutrients (i.e., long period of active uptake).

Arable cropping may result in soil N mining, even when fertilized with normal to high amounts of N, due to high potential for losses and poor utilization of mineralised N. The use of 25% of the production area for green manure production is insufficient as an N source alone to balance the N losses and the off-take by harvest, on a fertile soil with high yield potential. Arable cropping, which comprises the use of reduced tillage, catch crops, and moderate amounts of fertilizer appears, however, to balance the N flows at field

level. The use of P and K fertilizer in arable crop production may be used to balance the respective nutrient flows but it should be used with care, as over supply is a great risk.

Mixed dairy systems, producing both cereals and fodder crops, risk an undersupply of both N, P, and K, if there is no import of these nutrients to the system in the form of purchased fodder and/or other nutrient sources. The fixation of N by the legumes in the forage crops appears to be insufficient as the only N source. The use of mineral fertilizer may very well be used to balance the flows of N, P, and K in mixed dairy systems, but this leads to a risk of oversupply.

A relatively fertile soil may be exposed to substantial mining of N, P, and K over many years without causing detectible nutrient deficits in plants cropped on this soil. A long-term over- or undersupply will, sooner or later, result in a significant change in the content of plant available nutrients in the soil, but this change may be masked by the release of nutrients from nonavailable compounds.

Field nutrient budgets appear to be a good approach to the evaluation of whether a system is managed in a balanced way or not with regard to important nutrients such as N, P, and K, long before eventual unbalances become detectible by traditional analyses. However, the approach requires comprehensive datasets, which are usually unavailable under practical conditions.

Acknowledgments

H. Riley is gratefully acknowledged for critically reading the paper and T. Gaardløs for his skilled technical assistance. The project was funded jointly by the Norwegian Institute for Agricultural and Environmental Research, the Research Council of Norway, and Yara International ASA.

References

[1] R. Eltun, "The Apelsvoll cropping system experiment. III. Yield and grain quality of cereals," *Norwegian Journal of Agriculture*, vol. 10, no. 1, pp. 7–22, 1996.

[2] A. Korsaeth and R. Eltun, "Nitrogen mass balances in conventional, integrated and ecological cropping systems and the relationship between balance calculations and nitrogen runoff in an 8-year field experiment in Norway," *Agriculture, Ecosystems and Environment*, vol. 79, no. 2-3, pp. 199–214, 2000.

[3] G. Lien, O. Flaten, A. Korsaeth et al., "Comparison of risk in organic, integrated and conventional cropping systems in Eastern Norway," *Journal of Farm Management*, vol. 12, no. 7, pp. 385–401, 2006.

[4] T. A. Breland and R. Eltun, "Soil microbial biomass and mineralization of carbon and nitrogen in ecological, integrated and conventional forage and arable cropping systems," *Biology and Fertility of Soils*, vol. 30, no. 3, pp. 193–201, 1999.

[5] H. Riley, R. Pommeresche, R. Eltun, S. Hansen, and A. Korsaeth, "Soil structure, organic matter and earthworm activity in a comparison of cropping systems with contrasting tillage, rotations, fertilizer levels and manure use," *Agriculture, Ecosystems and Environment*, vol. 124, no. 3-4, pp. 275–284, 2008.

[6] A. Korsaeth, "Relations between nitrogen leaching and food productivity in organic and conventional cropping systems in a long-term field study," *Agriculture, Ecosystems and Environment*, vol. 127, no. 3-4, pp. 177–188, 2008.

[7] B. T. Christensen and A. E. Johnston, "Chapter 18 Soil organic matter and soil quality-Lessons learned from long-term experiments at Askov and Rothamsted," *Developments in Soil Science*, vol. 25, no. C, pp. 399–430, 1997.

[8] E. Ekeberg and H. Riley, "The long-term fertilizer trials at Møystad, S:E: Norway," SP Report 29, 100th Anniversary Workshop Askov Experimental Station, 1995.

[9] N. A. Fettell and H. S. Gill, "Long-term effects of tillage, stubble, and nitrogen management on properties of a red-brown earth," *Australian Journal of Experimental Agriculture*, vol. 35, no. 7, pp. 923–928, 1995.

[10] P. R. Poulton, "The importance of long-term trials in understanding sustainable farming systems: the Rothamsted experience," *Australian Journal of Experimental Agriculture*, vol. 35, no. 7, pp. 825–834, 1995.

[11] W. R. Raun, G. V. Johnson, S. B. Phillips, and R. L. Westerman, "Effect of long-term N fertilization on soil organic C and total N in continuous wheat under conventional tillage in Oklahoma," *Soil and Tillage Research*, vol. 47, no. 3-4, pp. 323–330, 1998.

[12] J. Leifeld, R. Reiser, and H. R. Oberholzer, "Consequences of conventional versus organic farming on soil carbon: results from a 27-year field experiment," *Agronomy Journal*, vol. 101, no. 5, pp. 1204–1218, 2009.

[13] C. F. Drury, T. O. Oloya, D. J. McKenney, E. G. Gregorich, C. S. Tan, and C. L. vanLuyk, "Long-term effects of fertilization and rotation on denitrification and soil carbon," *Soil Science Society of America Journal*, vol. 62, no. 6, pp. 1572–1579, 1998.

[14] C. M. Penfold, M. S. Miyan, T. G. Reeves, and I. T. Grierson, "Biological farming for sustainable agricultural production," *Australian Journal of Experimental Agriculture*, vol. 35, no. 7, pp. 849–856, 1995.

[15] P. M. Berry, E. A. Stockdale, R. Sylvester-Bradley et al., "N, P and K budgets for crop rotations on nine organic farms in the UK," *Soil Use and Management*, vol. 19, no. 2, pp. 112–118, 2003.

[16] O. Oenema, H. Kros, and W. De Vries, "Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies," *European Journal of Agronomy*, vol. 20, no. 1-2, pp. 3–16, 2003.

[17] O. Oenema and M. Heinen, "Uncertainties in nutrient budget due to biases and errors," in *Nutrient Disequilibria in Agroecosystems: Concepts and Case Studies*, E. M. A. Smaling, O. Oenema, and L. O. Fresco, Eds., pp. 75–97, CAB International, Wallingford, UK, 1999.

[18] W. De Vries, J. Kros, O. Oenema, and J. De Klein, "Uncertainties in the fate of nitrogen II: a quantitative assessment of the uncertainties in major nitrogen fluxes in the Netherlands," *Nutrient Cycling in Agroecosystems*, vol. 66, no. 1, pp. 71–102, 2003.

[19] A. Korsaeth, "Runoff and leaching losses of N, P, and K in the period 2001–2011 in a long-term experiment with conventional and organic crop rotations," In press.

[20] WRB, *World Reference Base for Soil Resources*, FAO, Rome, Italy, 1998.

[21] H. Riley and R. Eltun, "The Apelsvoll cropping system experiment. II. Soil characteristics," *Norwegian Journal of Agricultural Sciences*, vol. 8, pp. 317–333, 1994.

[22] H. Egnér, H. Riehm, and W. R. Domingo, "Untersuchungen über die chemische Bodenanalyse als Grundlage für die Beurteilung de Nährstoffzustandes der Böden. II," *Kungliga Lantbrukshögskolans Annaler*, vol. 26, pp. 199–215, 1960.

[23] W. Aas, S. Solberg, S. Manø, and K. E. Yttrei, "Overvåking av langtransportert forurenset luft og nedbør," Report NILU OR 22/2007, Kjeller, Norway, 2007.

[24] IPCC National Greenhouse Gas Inventories Programme, "Agriculture, forestry and other land use," in *IPCC Guidelines for National Greenhouse Gas Inventories*, H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe, Eds., vol. 4, chapter 11, pp. 1–11, IGES, Japan, 2006.

[25] L. Larsson, M. Ferm, Å. Kasimir-Klemedtsson, and L. Klemedtsson, "Ammonia and nitrous oxide emissions from grass and alfalfa mulches," *Nutrient Cycling in Agroecosystems*, vol. 51, no. 1, pp. 41–46, 1998.

[26] K. A. Gomez and A. A. Gomez, *Statistical Procedures for Agricultural Research*, Wiley & Sons, New York, NY, USA, 2nd edition, 1984.

[27] H. Riley and E. Ekeberg, "Effects of depth and time of ploughing on yields of spring cereals and potatoes and on soil properties of a morainic loam soil," *Acta Agriculturae Scandinavica - Section B Soil and Plant Science*, vol. 48, no. 4, pp. 193–200, 1998.

[28] R. Eltun, A. Korsaeth, and O. Nordheim, "A comparison of environmental, soil fertility, yield, and economical effects in six cropping systems based on an 8-year experiment in Norway," *Agriculture, Ecosystems and Environment*, vol. 90, no. 2, pp. 155–168, 2002.

[29] L. Bergström, H. Kirchmann, H. Aronsson, G. Torstensson, and L. Mattsson, "Use Efficiency and Leaching of Nutrients in Organic and Conventional Cropping Systems in Sweden," in *Organic Crop Production—Ambitions and Limitations*, H. Kirchmann and L. Bergström, Eds., pp. 117–141, Springer, New York, NY, USA, 2008.

[30] H. Riley and M. Bakkegard, "Declines of soil organic matter content under arable cropping in southeast Norway," *Acta Agriculturae Scandinavica Section B*, vol. 56, no. 3, pp. 217–223, 2006.

[31] G. Torstensson, H. Aronsson, and L. Bergstrom, "Nutrient use efficiencies and leaching of organic and conventional cropping systems in Sweden," *Agronomy Journal*, vol. 98, no. 3, pp. 603–615, 2006.

[32] D. P. Heenan, W. J. McGhie, F. M. Thomson, and K. Y. Chan, "Decline in soil organic carbon and total nitrogen in relation to tillage, stubble management, and rotation," *Australian Journal of Experimental Agriculture*, vol. 35, no. 7, pp. 877–884, 1995.

[33] G. Uhlen, "Long-term effects of fertilizers, manure, straw and crop rotation on total-N and total-C in soil," *Acta Agriculturae Scandinavica, Section B*, vol. 41, pp. 119–127, 1991.

[34] H. Steinshamn, E. Thuen, M. A. Bleken, U. T. Brenoe, G. Ekerholt, and C. Yri, "Utilization of nitrogen (N) and phosphorus (P) in an organic dairy farming system in Norway," *Agriculture, Ecosystems and Environment*, vol. 104, no. 3, pp. 509–522, 2004.

[35] E. Syväsalo, K. Regina, E. Turtola, R. Lemola, and M. Esala, "Fluxes of nitrous oxide and methane, and nitrogen leaching from organically and conventionally cultivated sandy soil in western Finland," *Agriculture, Ecosystems and Environment*, vol. 113, no. 1–4, pp. 342–348, 2006.

[36] D. J. Greenwood, F. Gastal, G. Lemaire, A. Draycott, P. Millard, and J. J. Neeteson, "Growth rate and % N of field grown crops: theory and experiments," *Annals of Botany*, vol. 67, no. 2, pp. 181–190, 1991.

[37] F. E. Khasawneh and E. C. Doll, "The use of phosphate rock for direct application to soils," *Advances in Agronomy*, vol. 30, no. C, pp. 159–206, 1979.

[38] M. Odlare, M. Pell, and K. Svensson, "Changes in soil chemical and microbiological properties during 4 years of application of various organic residues," *Waste Management*, vol. 28, no. 7, pp. 1246–1253, 2008.

[39] H. Bengtsson, I. Öborn, S. Jonsson, I. Nilsson, and A. Andersson, "Field balances of some mineral nutrients and trace elements in organic and conventional dairy farming—a case study at Örjebyn, Sweden," *European Journal of Agronomy*, vol. 20, no. 1-2, pp. 101–116, 2003.

[40] J. Verloop, J. Oenema, S. L. G. Burgers, H. F. M. Aarts, and H. van Keulen, "P-equilibrium fertilization in an intensive dairy farming system: effects on soil-P status, crop yield and P leaching," *Nutrient Cycling in Agroecosystems*, vol. 87, no. 3, pp. 369–382, 2010.

[41] H. Riley, "Long-term fertilizer trials on loam soil at Møystad, south-eastern Norway: crop yields, nutrient balances and soil chemical analyses from 1983 to 2003," *Acta Agriculturae Scandinavica Section B*, vol. 57, no. 2, pp. 140–154, 2007.

[42] L. Taiz and E. Zeiger, *Plant Physiology*, Sinauer Associates, Sunderland, Mass, USA, 3rd edition, 2002.

[43] S. Mathews, J. P. I. Tritschler, and S. C. Miyasaka, "Phosphorus management and sustainability," in *Grass for Daily Cattle*, J. H. Cherney and D. J. R. Cherney, Eds., pp. 193–222, CABI International, Wallingford, UK, 1998.

[44] S. D. Heming, "Potassium balances for arable soils in southern England 1986–1999," *Soil Use and Management*, vol. 20, no. 4, pp. 410–417, 2004.

[45] L. Blake, S. Mercik, M. Koerschens et al., "Potassium content in soil, uptake in plants and the potassium balance in three European long-term field experiments," *Plant and Soil*, vol. 216, no. 1-2, pp. 1–14, 1999.

[46] I. Öborn, Y. Andrist-Rangel, M. Askekaard, C. A. Grant, C. A. Watson, and A. C. Edwards, "Critical aspects of potassium management in agricultural systems," *Soil Use and Management*, vol. 21, pp. 102–112, 2005.

[47] A. F. Øgaard and S. Hansen, "Potassium uptake and requirement in organic grassland farming," *Nutrient Cycling in Agroecosystems*, vol. 87, no. 1, pp. 137–149, 2010.

[48] Y. Andrist-Rangel, A. C. Edwards, S. Hillier, and I. Öborn, "Long-term K dynamics in organic and conventional mixed cropping systems as related to management and soil properties," *Agriculture, Ecosystems and Environment*, vol. 122, no. 4, pp. 413–426, 2007.

[49] J. Holmqvist, A. F. Øgaard, I. Öborn, A. C. Edwards, L. Mattsson, and H. Sverdrup, "Application of the Profile model to estimate potassium release from mineral weathering in Northern European agricultural soils," *European Journal of Agronomy*, vol. 20, no. 1-2, pp. 149–163, 2003.

[50] H. Kuhlmann, "Importance of the subsoil for the K nutrition of crops," *Plant and Soil*, vol. 127, no. 1, pp. 129–136, 1990.

[51] L. E. De-Bashan and Y. Bashan, "Recent advances in removing phosphorus from wastewater and its future use as fertilizer (1997–2003)," *Water Research*, vol. 38, no. 19, pp. 4222–4246, 2004.

[52] A.-G. Roer, A. Korsaeth, T. M. Henriksen, O. Michelsen, and A. H. Strømman, "The influence of system boundaries on life cycle assessment of grain production in central southeast Norway," *Agricultural Systems*, vol. 111, pp. 75–84, 2012.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 548620, 10 pages
doi:10.1155/2012/548620

Research Article

Impact of No-Tillage and Conventional Tillage Systems on Soil Microbial Communities

Reji P. Mathew,[1] Yucheng Feng,[1] Leonard Githinji,[2] Ramble Ankumah,[2] and Kipling S. Balkcom[3]

[1] *Department of Agronomy and Soils, Auburn University, Auburn, AL 36849, USA*
[2] *Department of Agricultural and Environmental Sciences, Tuskegee University, Tuskegee, AL 36088, USA*
[3] *USDA-ARS, Auburn, AL 36849, USA*

Correspondence should be addressed to Yucheng Feng, yfeng@auburn.edu

Received 6 December 2011; Revised 4 April 2012; Accepted 5 April 2012

Academic Editor: Philip White

Soil management practices influence soil physical and chemical characteristics and bring about changes in the soil microbial community structure and function. In this study, the effects of long-term conventional and no-tillage practices on microbial community structure, enzyme activities, and selected physicochemical properties were determined in a continuous corn system on a Decatur silt loam soil. The long-term no-tillage treatment resulted in higher soil carbon and nitrogen contents, viable microbial biomass, and phosphatase activities at the 0–5 cm depth than the conventional tillage treatment. Soil microbial community structure assessed using phospholipid fatty acid (PLFA) analysis and automated ribosomal intergenic spacer analysis (ARISA) varied by tillage practice and soil depth. The abundance of PLFAs indicative of fungi, bacteria, arbuscular mycorrhizal fungi, and actinobacteria was consistently higher in the no-till surface soil. Results of principal components analysis based on soil physicochemical and enzyme variables were in agreement with those based on PLFA and ARISA profiles. Soil organic carbon was positively correlated with most of the PLFA biomarkers. These results indicate that tillage practice and soil depth were two important factors affecting soil microbial community structure and activity, and conservation tillage practices improve both physicochemical and microbiological properties of soil.

1. Introduction

Tillage systems influence physical, chemical, and biological properties of soil and have a major impact on soil productivity and sustainability. Conventional tillage practices may adversely affect long-term soil productivity due to erosion and loss of organic matter in soils. Sustainable soil management can be practiced through conservation tillage (including no-tillage), high crop residue return, and crop rotation [1]. Studies conducted under a wide range of climatic conditions, soil types, and crop rotation systems showed that soils under no-tillage and reduced tillage have significantly higher soil organic matter contents compared with conventionally tilled soils [2].

Conservation tillage is defined as a tillage system in which at least 30% of crop residues are left in the field and is an important conservation practice to reduce soil erosion [3]. The advantages of conservation tillage practices over conventional tillage include (1) reducing cultivation cost; (2) allowing crop residues to act as an insulator and reducing soil temperature fluctuation; (3) building up soil organic matter; (4) conserving soil moisture [4, 5].

Different tillage practices cause changes in soil physical properties, such as bulk density [6], water holding capacity [7], pore size distribution [8], and aggregation [9]. Stratification of soil organic matter and differences in nutrient distribution have also been observed in long-term conservation tillage systems [10, 11]. Thus, altered soil physical and chemical conditions under conservation tillage create significantly different habitats for microorganisms and result in shifts of soil microbial community structure [10–13]. Conventional tillage can lead to soil microbial communities

dominated by aerobic microorganisms, while conservation tillage practices increase microbial population and activity [11] as well as microbial biomass [10, 14].

Several studies have examined the effects of tillage practices on soil microbial communities in different cropping systems. In a long-term continuous cotton system, the tillage treatment effect varied by soil depth and over time; the impact of treatments was more pronounced during the fallow period and early in the growing season [12]. Although fungal dominance is commonly assumed in no-till soils, the relative abundance of fungi over bacteria is not consistently greater in the Northern Great Plain soils under long-term no-till practices compared with intensive tillage [13]. Ibekwe et al. [15] used biochemical- (i.e., PLFA) and nucleic-acid-based approaches to study the effect of tillage on soil microbial communities in four eastern Washington State soils. PLFA and denaturing gradient gel electrophoresis (DGGE) analyses showed a common pattern of clustering from the four soils and revealed that soil microbial communities respond more to soil management than annual precipitation.

Various culture-independent methods are available for characterizing soil microbial communities; these methods vary in their sensitivity for detecting microbial community changes. Polyphasic approaches are often used to study soil microbial communities due to the extraordinary magnitude of community size and diversity. PLFAs are a major constituent of cell membranes and have been used to identify individual species of bacteria and fungi. Since they are degraded rapidly upon cell death, PLFAs can be used to characterize living microbial biomass. PLFA analysis also provides insights into the broad scale structure of both bacteria and eukaryotic microorganisms [16]. The automated ribosomal analysis (ARISA) is a nucleic-acid-based method, which has a finer resolution for bacterial and fungal communities. This method involves polymerase chain reaction (PCR) amplification of the intergenic region between the small and large subunit ribosomal RNA genes [17]. Since the intergenic region exhibits considerable heterogeneity in both length and nucleotide sequence, ARISA has been used to provide rapid estimation of microbial diversity and community composition.

Soil enzymes play key biochemical functions in the decomposition of organic matter in the soil [18, 19]. They are process level indicators, which reflect past soil biological activity as influenced by soil management. Phosphatases are a broad group of enzymes that are capable of catalyzing hydrolysis of esters and anhydrides of phosphoric acid and have been reported to be good indicators of soil fertility [20, 21]. Phosphatases play key roles in phosphorus cycling, including degradation of phospholipids.

Conservation tillage techniques are widely used in the southeastern United States to conserve soil moisture, nutrients, and structure, providing habitats and substrates for biota, especially microorganisms, which are responsible for mineralization of soil nutrients. In this study, the effects of conventional and no-tillage practices on soil microbial communities were investigated in a continuous corn production system by determining microbial community structure using PLFA analysis and ARISA as well as microbial activities as indicated by soil phosphatases. The central hypothesis was that long-term use of no-tillage practices would cause shifts in soil microbial community structure relative to conventional tillage practices.

2. Materials and Methods

2.1. Study Site and Soil Sampling. The study site was located at the Tennessee Valley Research and Extension Center in Belle Mina, Alabama, USA. The soil type was a Decatur silt loam (Fine, kaolinitic, thermic Rhodic Paleudults). The field experiment was arranged in a randomized complete block factorial design of four replications with tillage being the main factor. The no-tillage plots were established in 1990 and conventionally tilled plots in 1994 from previously established no-till plots. Conventional tillage involved disking and chisel plowing in the fall followed by disking and field cultivating in the spring. Cotton was planted at the study site until 2003 and corn from 2004. Winter rye was seeded in the fall in no-tillage plots and terminated before spring planting with glyphosate application. A detailed description on the history of the field experiment can be found in Schwab et al. [4]. Soil sampling was performed in April of 2008 prior to planting to minimize the effect of plant growth on microbial communities in order to observe the tillage treatment effect. Soil cores (40 to 45 cores) were collected using tube samplers (2.5 cm in diameter) from randomly selected locations in each plot. Soil cores were separated into two depths (0–5 and 5–15 cm) in the field, composited by depth and thoroughly mixed. Field-moist samples were transported to the laboratory on ice and then passed through a 4 mm sieve within 24 hours. Three additional intact soil cores were collected from each plot for bulk density determination at two depths.

2.2. Characterization of Soil Physical and Chemical Properties. Subsamples from each of the 16 composite samples were taken for gravimetric moisture content determination and chemical analysis after air drying. Total carbon and nitrogen were analyzed using a TruSpec CN analyzer (Leco Corp., St. Joseph, MI, USA). Since there is no appreciable carbonate carbon in this inherently acid soil, the total carbon content is equivalent to the soil organic carbon content. Soil pH was measured using 1 : 1 soil/water and 1 : 2 soil/0.01 M $CaCl_2$ suspensions. Bulk density was determined by measuring the moisture loss from intact soil cores of a known volume after drying at 105°C for 24 hours.

2.3. Soil Phosphatase Activities. Air-dried soil samples passed through a 2 mm sieve were used to analyze phosphomonoesterases (acid and alkaline phosphatases) and phosphodiesterase activities as described by Tabatabai [22]. The methods are based on colorimetric determination of *p*-nitrophenol released by phosphatase activity when soil is incubated with buffered substrates at each enzyme's optimal pH [22]. Acid and alkaline phosphatase assays were performed in a modified universal buffer containing 10 mM *p*-nitrophenyl phosphate at pH 6.5 and pH 11, respectively.

Phosphodiesterase assay was performed at pH 8 with 10 mM *p*-nitrophenyl phosphate serving as the substrate. All analyses were done in triplicate.

2.4. Soil Microbial Community Analyses. The homogenized subsamples were taken for extraction of lipids and DNA. Field moist soil samples were stored at 4°C for no more than two weeks before lipid extraction and at −20°C until soil DNA extraction.

2.4.1. Phospholipid Fatty Acid (PLFA) Analysis. Phospholipid fatty acid analysis was performed as described by Feng et al. [12]. It involved extraction of total lipids from soil, fractionation of total lipids, derivatization of fatty acids to form fatty acid methyl esters (FAMEs), and GC analysis of FAMEs. Briefly, duplicate field moist soil samples (8 g dry weight) from each of the 16 composite samples were used for extracting total lipids using a single-phase citrate buffer-chloroform-methanol solution (1 : 2 : 0.8 v/v/v, pH 4). The phospholipids were separated from neutral lipids and glycolipids using silicic acid column chromatography. The phospholipids were then subjected to a mild alkaline methanolysis, and resulting FAMEs were extracted using hexane and dried under nitrogen gas. The FAMEs containing 19 : 0 methyl ester as an internal standard were analyzed using a Hewlett Packard 5890 gas chromatograph with a 25 m HP Ultra 2 capillary column and a flame ionization detector. FAME peaks were identified using the MIDI peak identification software (MIDI, Inc., Newark, DE, USA) and quantified based on the internal standard added. The nomenclature for fatty acids used here was described by Feng et al. [12].

2.4.2. Automated Ribosomal Intergenic Spacer Analysis (ARISA). ARISA involved total community DNA extraction from soil, PCR amplification using fluorescence-tagged oligonucleotide primers targeting intergenic transcribed spacer region, automated electrophoresis, laser detection of fluorescent DNA fragments, and analysis of banding patterns. Total soil DNA was extracted from 8 g of moist soil using a PowerMax Soil DNA Kit (MoBio Labs Inc., Carlsbad, CA, USA) following the manufacturer's instructions. The extracted DNA was quantified using a NanoDrop ND-1000 Spectrophotometer (Thermo Fisher Scientific, Wilmington, DE, USA) and stored at −80°C until use. Both bacterial and fungal ARISAs were performed to determine soil microbial community structure.

The bacterial primers used in the PCR reactions were ITSF (5′-GTCGTAACAAGGTAGCCGTA-3′) and ITSReub (5′-GCCAAGGCATCCACC-3′) [23]. The reaction mixture contained 12.5 μL of 2X GoTaq colorless master mix (Promega, Madison, WI, USA), 25 μg of bovine serum albumin (Sigma-Aldrich Co., St. Louis, MO, USA), 0.2 μM of ITSF primer, 0.2 μM of ITSF primer labeled with IRD800 fluorochrome (LI-COR, Lincoln, Nebraska), 0.4 μM of ITSReub primer, 5 μL of template DNA (∼20 ng), and nuclease-free water to make the final volume to 25 μL. Amplification was performed on a Biometra T-Gradient thermocycler (Whatmann, Goettingen, Germany) using the following cycling parameters: 3 min at 94°C, 30 cycles of 60 s at 94°C, 30 s at 55°C and 60 s at 72°C, and a final 5 min at 72°C [24].

The fungal automated intergenic spacer analyses were performed using ITS1F (5′-CTTGGTCATTTAGAGGAA-GTAA-3′) and 3126T (5′-ATATGCTTAAGTTCAGCG-GGT-3′) [25, 26]. The reaction mixture (25 μL) consisted of 12.5 μL of 2X GoTaq colorless master mix, 25 μg of bovine serum albumin, 0.3 μM of ITS1F primer, 0.1 μM of ITS1F primer labeled with IRD800 fluorochrome, 0.4 μM of 3126T primer, and 5 μL of template DNA (∼20 ng). The thermocycling conditions were as follows: 4 min at 95°C, 35 cycles of 60 s at 95°C, 30 s at 53°C and 60 s at 72°C, and a final 7 min at 72°C [27, 28].

A total of 5 μL amplified PCR products (2.5 μL from each replicate) was mixed with 2.5 μL of stop buffer (LI-COR Blue Stop Solution), denatured at 95°C for 2 min, and then placed on ice. The denatured PCR products (0.8–1 μL) were loaded on 6% polyacrylamide gel along with 0.8 μL of the IRD800 50–700 bp sizing standard (LI-COR). ARISA fragments were resolved under denaturing conditions for 9 hours at 1,500 V using the LI-COR 4300 sequencer. Laser-scanned banding pattern image from the LI-COR sequencer was converted to 8-bit TIFF using Kodak 1D Image Analysis Software (Eastman Kodak Co., Rochester, NY, USA).

2.5. Data Analysis. All microbial parameters were converted to unit weight of dry soil prior to data analysis. Data for general soil physicochemical and biological properties were analyzed using PROC MIXED and multiple comparison procedure as well as principal components analysis. The mole percent distribution of PLFAs was analyzed using principal components analysis (PROC PRINCOMP, SAS ver.9.1.3). Analysis of PLFA profiles was performed using a set of 50 fatty acids that were present in most of the samples. Bacterial biomass was calculated using the sum of 15 bacterial markers, that are, 14:0, 15:0, a15:0, i15:0, i16:0, 16:1ω5, 16:1ω7, 16:1ω9, 17:0, a17:0, i17:0, 18:0, 18:1ω7, cy17:0, and cy19:0 [29, 30]. Fungal biomass was assessed using 18:2ω6, 9 [31] and physiological stress by the ratio of cy19:0/18:1ω7 [32, 33]. The fungal to bacterial PLFA ratio was calculated using 18:2ω6, 9/sum of bacterial markers [30, 34]. Gram-negative to Gram-positive bacteria were calculated using (i15:0 + a15:0 + i16:0 + 10Me16:0)/(16:1ω7 + 18:1ω7 + cy19:0). The PLFA biomarkers and ratios were also analyzed using PROC MIXED and multiple comparison procedure.

ARISA-banding pattern images were processed using the software BIONUMERICS Ver. 5.0 (Applied Maths, Belgium). Each image was normalized using the 50–700 bp sizing standard as the external reference standard, which allowed for comparison of multiple gels. Levels of similarity between DNA fingerprints were compared using a densitometric curve-based method with the cosine coefficient after the conversion, normalization, and background subtraction with mathematical algorithms of banding patterns. Dendrograms were developed using cluster analysis performed with the cosine similarity coefficient and unweighted pair-group method using average linkages (UPGMA). The position tolerance was set at an optimization of 0.5%, and band comparison was made using a position tolerance of 1%. Principal

Table 1: Selected chemical and physical properties of soils from no-till (NT) and conventional-till (CT) treatments*.

Tillage treatment	Depth (cm)	Organic C (%)	Total N(%)	C/N ratio	Bulk density (Mg m^{-3})	Soil pH (1 : 2 $CaCl_2$)	Soil moisture content
NT	0–5	1.94a	0.13a	14.9a	1.52b	6.1a	0.25a
NT	5–15	0.84b	0.07b	11.7b	1.65a	5.9a	0.18b
CT	0–5	0.92b	0.08b	11.0b	1.53b	6.1a	0.15c
CT	5–15	0.76b	0.07b	10.9b	1.66a	6.2a	0.12d

* Means ($n = 4$) followed by the same letter in a column are not significantly different (Tukey, $P \leq 0.05$).

Table 2: Total PLFAs and phosphatase† activities in no-till (NT) and conventional-till (CT) soils*.

Tillage treatment	Depth (cm)	Total PLFAs (nmol g^{-1})	Acid P	Alk P	PDE
			(μg of p-nitrophenol g^{-1} hr^{-1})		
NT	0–5	104a	367a	321a	132a
NT	5–15	38b	307ab	44c	36b
CT	0–5	39b	200b	89b	32b
CT	5–15	30c	202b	87b	34b

† Acid P, acid phosphatase; Alk P, alkaline phosphatase; PDE, phosphodiesterase.
*Means ($n = 4$) followed by the same letter in a column are not significantly different (Tukey, $P \leq 0.05$).

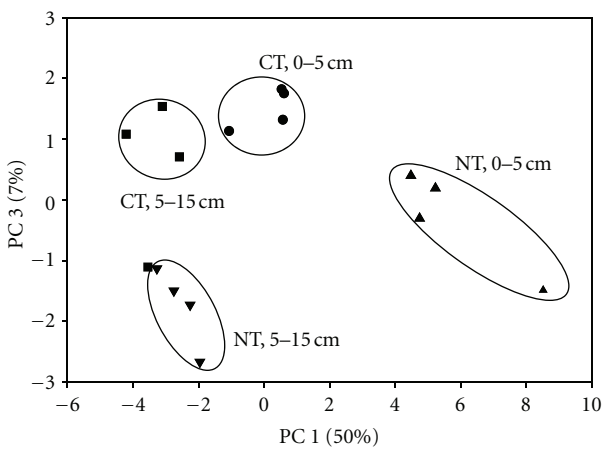


Figure 1: Principal components analysis of PLFA profiles.

components analysis was used to determine distribution of fingerprint patterns according to different tillage treatment and soil depth.

3. Results

3.1. Soil Physicochemical and Biochemical Properties. Physicochemical characteristics of surface soils differed between tillage treatments (Table 1). Soil organic C, total N, and C/N ratios were significantly higher in the no-till treatment than the conventional tillage treatment at the 0–5 cm depth but not at the lower depth. Depth effects were observed only in the no-till treatment. Bulk density for surface soil in both no-till and conventional-till treatments was lower compared with the subsurface soil although no significant difference was observed between tillage treatments. Soil pH values did not vary by tillage treatment or soil depth.

Total PLFA concentrations, an indicator of viable microbial biomass, ranged from 30 nmol/g of soil for the conventional-till treatment at the 5–15 cm depth to 104 nmol/g of soil for the no-till treatment at the 0–5 cm depth (Table 2). The total PLFA concentration in the no-till surface soil was 2.7 times higher than in the conventionally tilled soil. As soil depth increased, total PLFA concentrations decreased in both tillage treatments. Soil phosphatase activities showed a similar trend with no-till soil having significantly higher activities than conventionally tilled soil at the 0 to 5 cm depth (Table 2). In the no-till treatment, the enzyme activities were significantly higher at the 0 to 5 cm than at the 5 to 15 cm depth except for acid phosphatase. Among three soil phosphatases, acid phosphatase activity was the highest, ranging from 200 to 367 μg of p-nitrophenol g^{-1} hr^{-1}. Alkaline phosphatase activities ranged from 44 to 321 and phosphodiesterase from 32 to 132 μg of p-nitrophenol g^{-1} hr^{-1}.

3.2. PLFA. Principal components analysis of PLFA profiles showed that 81% of the total sample variation was explained by the first three principal components (PCs). PC 1 explained 50% of the total variation and separated the soil depth effect. PC 3 explained 7% of the variation and separated the tillage effect (Figure 1). The influential fatty acids for the first principal component (Table 3) were an actinobacterial biomarker (10Me16:0), an aerobic bacterial biomarker (16:1ω7), and fungal biomarkers (18:1ω9 and 18:2ω6, 9). The third principal component was influenced mostly by a nonspecific fatty acid (i17:1), an anaerobic bacterial biomarker (cy19:0), and an actinobacterial biomarker (10Me16:0) (Table 3).

The relative abundance of fungal biomarker (18:2ω6, 9) as indicated by mole percentage did not show tillage treatment effect; however, the concentration of this biomarker was higher in no-till than conventionally tilled soil at the surface depth (Table 4). The sum of bacterial PLFAs showed a similar trend. Similar to the relative abundance of fungal

Table 3: PLFA having scores > | ± 0.23| for the first and third principal components.

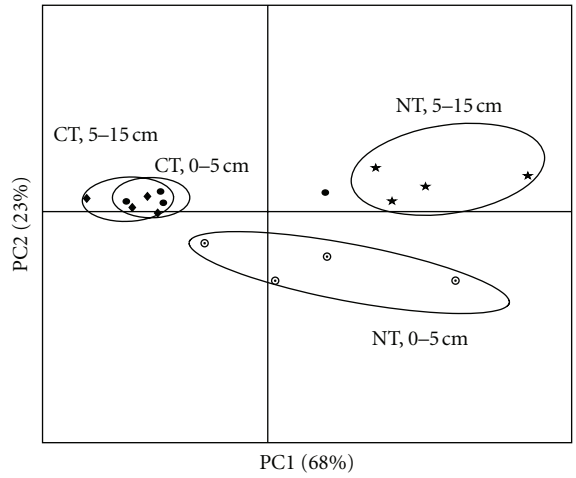
Fatty acid	Score	Specificity as a biomarker*
PC 1		
10Me16:0	−0.65	Actinobacteria
16:1ω7	0.32	Aerobic bacteria
18:1ω9	0.29	Fungi
18:2ω6,9	0.23	Fungi
PC 3		
i17:1	−0.51	Nonspecific
cy19:0	−0.34	Anaerobic bacteria
10Me16:0	0.30	Actinobacteria

* Source: Findlay [48] and Paul and Clark [49].

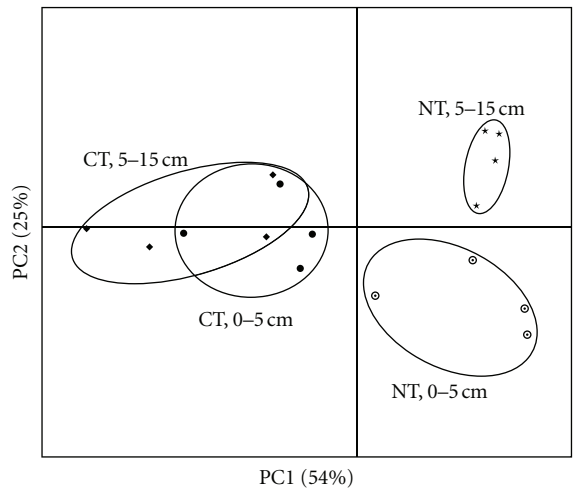
and bacterial PLFAs, the fungal to bacterial PLFA ratios showed depth but not tillage treatment effects. Although arbuscular mycorrhizal (AM) fungi proportions only showed the depth effect, concentrations of the AM fungal biomarker (16:1ω5) showed both tillage and depth effects. The relative abundance of the actinobacterial biomarker (10Me18:0) was similar across tillage treatments and soil depths, whereas its concentrations differed by tillage and depth. Gram-positive to Gram-negative bacterial PLFA ratios (Table 4) and the stress indicator ratios (cy19:0/18:1ω7, data not shown) did not show any significant difference for tillage treatment or depth.

3.3. ARISA. Principal components analysis of bacterial ARISA profiles showed that the first and second principal components explained 68% and 23% of the total sample variation, respectively (Figure 2(a)). The first principal component separated the no-tillage from conventional tillage treatment, and the second principal component separated the no-till treatment by soil depth. There was no depth separation for the conventional tillage treatment. Principal components analysis of fungal ARISA profiles showed that the first and second principal components explained 54% and 25% of the total sample variation, respectively (Figure 2(b)). The first principal component separated the tillage effect, while the second principal component separated the surface and subsurface soil for the no-till treatment.

3.4. Interactions between Soil Physicochemical and Biochemical Variables. Correlation and multivariate analyses were performed to determine interactions between soil physicochemical and biochemical variables. Acid and alkaline phosphatases as well as phosphodiesterase activities were positively correlated to soil organic carbon and soil moisture contents (Table 5). Soil bulk density was negatively correlated with alkaline phosphatase ($r = -0.56$) and phosphodiesterase ($r = -0.46$) activities but had no significant correlation with acid phosphatase activities. Total PLFAs were highly correlated with soil organic carbon ($r = 0.98$) and moisture content ($r = 0.87$). The fungal to bacterial PLFA ratios and proportions of AM fungal biomarker as well as the



(a) Bacteria



(b) Fungi

Figure 2: Principal components analyses of bacterial (a) and fungal (b) ARISA profiles.

fungal biomarker were also positively correlated with soil organic carbon (Table 5). Bacterial PLFA proportions were negatively correlated with both soil organic carbon and moisture content but positively correlated with bulk density. The fungal biomarker and the fungal to bacterial PLFA ratio were negatively correlated with soil bulk density. The relative abundance of AM fungal biomarker was positively correlated with soil moisture content.

Multivariate analysis using selected soil physicochemical and enzyme variables (i.e., soil organic carbon, total nitrogen, soil moisture, soil pH, bulk density, acid and alkaline phosphatases, and phosphodiesterase) also revealed tillage and depth effects (Figure 3). Principal components analysis showed that the first principal component explained 68% of the total sample variation and the second principal component 17%. Data points for the no-tillage treatment at the surface depth formed a distinct cluster by themselves. Data points for the conventional tillage treatment at both depths clustered together, whereas those for the no-till

Table 4: PLFA biomarkers and ratios† in no-till (NT) and conventional-till (CT) soils*.

Tillage treatment	Depth (cm)	Fungi/bacteria	G+/G− bacteria†	Fungi		Bacteria		AM fungi		Actinobacteria	
				(mol%)	(nmol g^{-1})	(mol%)	(nmol g^{-1})	(mol%)	(nmol g^{-1})	(mol%)	(nmol g^{-1})
NT	0–5	0.08a	1.48a	3.97a	4.47a	53.1a	50.9a	3.89a	4.32a	2.18a	2.09a
NT	5–15	0.03b	1.76a	2.29b	0.76b	56.8b	21.3b	2.93b	1.13bc	2.61a	1.11b
CT	0–5	0.07a	1.54a	3.87a	1.41b	53.0a	20.9b	3.27ab	1.17b	2.41a	0.99bc
CT	5–15	0.04b	1.84a	2.00b	0.68b	57.0b	16.9c	2.83b	0.83c	2.90a	0.77c

† G+/G− bacteria: ratio of Gram-positive to Gram-negative bacterial PLFA.

* Means ($n = 4$) followed by the same letter in a column are not significantly different (Tukey, $P \leq 0.05$).

Table 5: Correlation coefficients between soil physicochemical and biochemical variables determined in the study.

Soil property	Phosphatase activity†			PLFA biomarkers and ratios				
	Acid P	Alk P	PDE	Total PLFA	Fungi	Bacteria	Fungi/bacteria	AM fungi
Soil organic carbon	0.72	0.95	0.92	0.98	0.53	−0.65	0.56	0.60
Soil moisture content	0.77	0.84	0.90	0.87	NS	−0.39	NS	0.45
Bulk density	NS*	−0.56	−0.46	−0.53	−0.62	0.49	−0.60	NS

† Acid P, acid phosphatase; Alk P, alkaline phosphatase; PDE, phosphodiesterase;
*NS: No significant correlation ($P \leq 0.05$).

Table 6: Soil physicochemical and enzyme variables having scores $\geq |\pm 0.38|$ for the first two principal components.

Soil properties	Score
PC 1	
Soil organic carbon	0.42
Total nitrogen	0.41
Alkaline phosphatase	0.41
Phosphodiesterase	0.41
Soil moisture	0.38
PC 2	
Soil pH (1 : 2 $CaCl_2$)	0.81

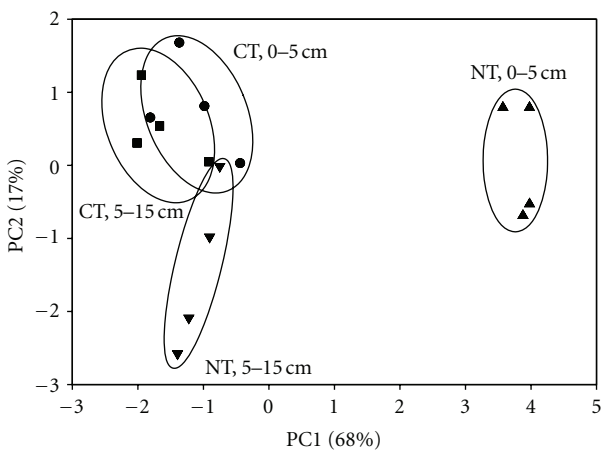


Figure 3: Principal components analysis using soil physicochemical and enzyme variables.

treatment formed two clusters separated by soil depth. The influential variables for the first principal component were soil organic carbon, total nitrogen, alkaline phosphatase, phosphodiesterase, and soil moisture and that for the second principal component was soil pH (Table 6).

4. Discussion

Changes in soil characteristics associated with adoption of conservation tillage systems generally result in improved soil quality, especially in the southeastern USA where soils are inherently low in fertility and susceptible to aggregate disruption and erosion. In this study, soil under the long-term no-till treatment had higher soil carbon and nitrogen contents, total PLFAs, and phosphatase activities at the 0–5 depth than that under the conventional-till treatment. Tillage treatment effects were less pronounced at the 5–15 cm depth. These observations are in agreement with previous findings reported by, for example, Ceja-Navarro et al. [35], Drijber et al. [36], Ekenler and Tabatabai [37], Feng et al. [12], Helgason et al. [13], and Ibekwe et al. [15]. Total PLFAs in the no-till surface soil were much higher than those reported in a previous study during the fallow period [12] conducted on the same soil type although organic carbon contents at the two sites were similar. This may be attributed the difference in the cropping systems: continuous cotton with no winter cover crop in the previous study versus continuous corn with rye as a winter cover crop in this study. Cotton is known to generate lesser residues than corn [38], and the rye cover crop provided additional organic matter input to the soil. Three years of corn/rye cropping system perhaps were not long enough for observing a significant change in soil organic matter; the increase in microbial biomass as indicated by total PLFAs, however, provides another line of evidence that microorganisms are sensitive and early indicators for soil quality evaluation. The findings of tillage treatment and depth effects on phosphatase activities were consistent with the study of Ekenler and Tabatabai [37]. Soil enzymes have been suggested as soil quality indicators owing to their relationship to soil biology and rapid response to changes in soil management and ease of measurement [39].

In no-till soils, the accumulation of crop residues on the soil surface results in enrichment of soil organic matter in the surface layer and as a consequence increased abundance of microorganisms. This study demonstrated a consistent increase in the abundance of fungi, bacteria, arbuscular mycorrhizal fungi, and actinobacteria in the no-till surface soil. Similar to other reports (e.g., Feng et al. [12]; Helgason et al. [13]; Pankhurst et al. [40]), this study did not show a fungal dominance in the no-till soil as indicated by the ratio of fungal to bacterial PLFAs. The relative abundance of fungi under no-till practices has been shown to be greater than that under conventional-till practices when fungal biomass was determined by measuring hyphal length [41]. This discrepancy may be attributed to differences in the methods used. As pointed out by Helgason et al. [13], microscopic measurements of fungal hyphal length performed by Frey et al. [41] include both viable and nonviable fungal hyphae. PLFA analysis on the other hand provides a measure of viable microbial biomass. Additional factors to be taken into account include that (1) different groups of microorganisms share overlapping PLFAs also contribute to the discrepancy

and (2) phospholipid concentrations in fungi are lower than those in bacteria. Nevertheless, comparison of fungal to bacterial PLFA ratios between tillage treatments is warranted.

Polyphasic approaches are often used to study soil microbial communities. PLFA analysis has been shown to be the best approach to discern a treatment effect on soil microbial community and be able to differentiate treatments that are not resolved by PCR-based methods in some cases [42]. In this study, both PLFA analysis and ARISA clearly demonstrated the shift in soil microbial communities associated with tillage practices. These findings are consistent with those reported by Drijber et al. [36], Feng et al. [12], and Peixoto et al. [43]. The observed changes in soil microbial communities can be attributed to favorable physical and chemical conditions under the no-tillage system for microbial activities. A closer examination of principal components analysis results for PLFA and ARISA profiles (Figures 1 and 2) revealed that the depth effect for conventionally tilled soil was more pronounced in PLFA analysis. This suggests that in addition to bacteria and fungi, microfauna (e.g., protozoa and nematodes) may contribute to the discrimination of the subtle difference between soil depths in the relatively well mixed conventionally tilled soil since eukaryotic organisms other than fungi contribute to the soil PLFAs.

ARISA is an automated DNA fingerprinting method targeting the intergenic spacer regions of bacteria and fungi in PCR; it is highly reproducible and effective in detecting changes in soil microbial community structure. Bacterial and fungal ARISA have previously been used in studies conducted on agricultural and forest soils [44, 45]. To our knowledge, this is the first time that ARISA was used to determine the impact of tillage practices on soil microbial communities. Although it provides information on genetic community structure of soil bacteria and fungi, the intergenic spacer regions targeted by ARISA cannot be used to identify dominant organisms. Little information is available regarding the specific microorganisms affected by different tillage practices. Ceja-Navarro et al. [35] conducted phylogenetic and multivariate analyses to determine the effects of zero tillage and conventional tillage on soil bacterial communities in a long-term maize-wheat rotation experiment. They found that bacterial communities under zero tillage and crop residue retention have the highest level of diversity and richness. Zero tillage has a positive effect on members of *Rhizobiales* and crop residue retention increases fluorescent *Pseudomonas* spp. and *Burkholderiales* group. In a rice-soybean rotation study, impact of conventional and no-tillage with and without cover crops on soil bacterial community structure was determined using PCR-DGGE without identification of bands through DNA sequencing [43]. Responses of bacterial communities to cultivation, tillage, and soil depth but not to cover cropping were detected.

Results of principal components analysis based on soil physicochemical and enzyme variables (Figure 3) were in general agreement with those based on PLFA and ARISA profiles. Soil organic carbon was the most influential factor for PC 1, confirming its critical role in the no-till system. Soil organic carbon was correlated with all biochemical variables except for the relative abundance of bacterial biomarkers. A negative correlation between soil organic carbon and bacterial PLFAs has also been observed by Zornoza et al. [46] and Helgason et al. [13]. Lauber et al. [47] quantified microbial communities by quantitative PCR and also reported lack of correlation between soil carbon and bacterial population. They showed that soil pH and texture are better predictors of soil bacteria.

5. Conclusions

In this study, soil under the long-term no-till treatment had higher soil carbon and nitrogen contents, total PLFAs, and phosphatase activities at the 0–5 cm depth than that under the conventional tillage treatment. Differences between tillage treatments at the 5–15 cm depth were negligible with the exception of alkaline phosphatase activities. Soil microbial communities shifted with tillage treatment and soil depth. Tillage practice and soil depth were two important factors affecting soil microbial communities. PLFA analysis and ARISA showed comparable results on treatment effects. PLFA profiles, however, detected differences in microbial communities associated with soil depth in the conventional tillage treatment. This study demonstrated that tillage systems influence soil microbial communities along with soil physicochemical properties. Further research is needed to determine the influence of tillage-induced changes on soil microbial community composition (i.e., the identity of key organisms) and their dynamics.

Acknowledgments

The authors would like to thank Dr. Zhanjiang (John) Liu for the use of LI-COR 4300 sequencer in the Fish Molecular Genetics and Biotechnology Laboratory at Auburn University and Dr. Edzard Van Santen for helping with statistical analysis. This work was supported, in part, by grants from the Alabama Agricultural Land-Grant Alliance and the Alabama Agricultural Experiment Station.

References

[1] P. R. Hobbs, K. Sayre, and R. Gupta, "The role of conservation agriculture in sustainable agriculture," *Philosophical Transactions of the Royal Society B*, vol. 363, no. 1491, pp. 543–555, 2008.

[2] R. Alvarez, "A review of nitrogen fertilizer and conservation tillage effects on soil organic carbon storage," *Soil Use and Management*, vol. 21, no. 1, pp. 38–52, 2005.

[3] N. D. Uri, "Factors affecting the use of conservation tillage in the United States," *Water, Air, and Soil Pollution*, vol. 116, no. 3-4, pp. 621–638, 1999.

[4] E. B. Schwab, D. W. Reeves, C. H. Burmester, and R. L. Raper, "Conservation tillage systems for cotton in the Tennessee Valley," *Soil Science Society of America Journal*, vol. 66, no. 2, pp. 569–577, 2002.

[5] T. O. West and W. M. Post, "Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis," *Soil Science Society of America Journal*, vol. 66, no. 6, pp. 1930–1946, 2002.

[6] M. M. Wander, M. G. Bidart, and S. Aref, "Tillage impacts on depth distribution of total and particulate organic matter in three Illinois soils," *Soil Science Society of America Journal*, vol. 62, no. 6, pp. 1704–1711, 1998.

[7] M. D. Trojan and D. R. Linden, "Macroporosity and hydraulic properties of earthworm-affected soils as influenced by tillage and residue management," *Soil Science Society of America Journal*, vol. 62, no. 6, pp. 1687–1692, 1998.

[8] R. H. Azooz, M. A. Arshad, and A. J. Franzluebbers, "Pore size distribution and hydraulic conductivity affected by tillage in Northwestern Canada," *Soil Science Society of America Journal*, vol. 60, no. 4, pp. 1197–1201, 1996.

[9] K. Y. Chan and J. A. Mead, "Surface physical properties of a sandy loam soil under different tillage practices," *Australian Journal of Soil Research*, vol. 26, no. 3, pp. 549–559, 1988.

[10] E. Kandeler, D. Tscherko, and H. Spiegel, "Long-term monitoring of microbial biomass, N mineralisation and enzyme activities of a chernozem under different tillage management," *Biology and Fertility of Soils*, vol. 28, no. 4, pp. 343–351, 1999.

[11] T. E. Staley, "Soil microbial biomass alterations during the maize silage growing season relative to tillage method," *Soil Science Society of America Journal*, vol. 63, no. 6, pp. 1845–1847, 1999.

[12] Y. Feng, A. C. Motta, D. W. Reeves, C. H. Burmester, E. Van Santen, and J. A. Osborne, "Soil microbial communities under conventional-till and no-till continuous cotton systems," *Soil Biology and Biochemistry*, vol. 35, no. 12, pp. 1693–1703, 2003.

[13] B. L. Helgason, F. L. Walley, and J. J. Germida, "Fungal and bacterial abundance in long-term no-till and intensive-till soils of the Northern Great Plains," *Soil Science Society of America Journal*, vol. 73, no. 1, pp. 120–127, 2009.

[14] E. L. Balota, A. Colozzi-Filho, D. S. Andrade, and R. P. Dick, "Microbial biomass in soils under different tillage and crop rotation systems," *Biology and Fertility of Soils*, vol. 38, no. 1, pp. 15–20, 2003.

[15] A. M. Ibekwe, A. C. Kennedy, P. S. Frohne, S. K. Papiernik, C. H. Yang, and D. E. Crowley, "Microbial diversity along a transect of agronomic zones," *FEMS Microbiology Ecology*, vol. 39, no. 3, pp. 183–191, 2002.

[16] A. Frostegard, A. Tunlid, and E. Baath, "Phospholipid fatty acid composition, biomass, and activity of microbial communities from two soil types experimentally exposed to different heavy metals," *Applied and Environmental Microbiology*, vol. 59, no. 11, pp. 3605–3617, 1993.

[17] M. M. Fisher and E. W. Triplett, "Automated approach for ribosomal intergenic spacer analysis of microbial diversity and its application to freshwater bacterial communities," *Applied and Environmental Microbiology*, vol. 65, no. 10, pp. 4630–4636, 1999.

[18] R. G. Burns, "Extracellular enzyme-substrate interactions in soil," in *Microbes in their Natural Environment*, R. W. J. H. Slater and J. W. T. Wimpenny, Eds., pp. 249–298, Cambridge University Press, London, UK, 1983.

[19] R. L. Sinsabaugh, R. K. Antibus, and A. E. Linkins, "An enzymic approach to the analysis of microbial activity during plant litter decomposition," *Agriculture, Ecosystems and Environment*, vol. 34, no. 1–4, pp. 43–54, 1991.

[20] R. P. Dick, J. A. Sandor, and N. S. Eash, "Soil enzyme activities after 1500 years of terrace agriculture in the Colca Valley, Peru," *Agriculture, Ecosystems and Environment*, vol. 50, no. 2, pp. 123–131, 1994.

[21] W. A. Dick and M. A. Tabatabai, "Potential uses of soil enzymes," in *Soil Microbial Ecology: Applications in Agricultural and Environmental Management*, F. B. Metting Jr, Ed., pp. 95–127, Marcel Dekker, New York, NY, USA, 1992.

[22] M. A. Tabatabai, "Soil enzymes," in *Methods of Soil Analysis. Part 2-Chemical and Microbiological Properties*, A. L. Page, R. H. Miller, and D. R. Keeney, Eds., pp. 775–883, Soil Science Society of America, Madison, Wis, USA, 1994.

[23] M. Cardinale, L. Brusetti, P. Quatrini et al., "Comparison of different primer sets for use in automated ribosomal intergenic spacer analysis of complex bacterial communities," *Applied and Environmental Microbiology*, vol. 70, no. 10, pp. 6147–6156, 2004.

[24] L. Ranjard, F. Poly, J. Combrisson et al., "Heterogeneous cell density and genetic structure of bacterial pools associated with various soil microenvironments as determined by enumeration and DNA fingerprinting approach (RISA)," *Microbial Ecology*, vol. 39, no. 4, pp. 263–272, 2000.

[25] B. Nicolardot, L. Bouziri, F. Bastian, and L. Ranjard, "A microcosm experiment to evaluate the influence of location and quality of plant residues on residue decomposition and genetic structure of soil microbial communities," *Soil Biology and Biochemistry*, vol. 39, no. 7, pp. 1631–1644, 2007.

[26] L. Ranjard, A. Echairi, V. Nowak, D. P. H. Lejon, R. Nouaïm, and R. Chaussod, "Field and microcosm experiments to evaluate the effects of agricultural Cu treatment on the density and genetic structure of microbial communities in two different soils," *FEMS Microbiology Ecology*, vol. 58, no. 2, pp. 303–315, 2006.

[27] N. Kennedy, J. Connolly, and N. Clipson, "Impact of lime, nitrogen and plant species on fungal community structure in grassland microcosms," *Environmental Microbiology*, vol. 7, no. 6, pp. 780–788, 2005.

[28] N. Kennedy, S. Edwards, and N. Clipson, "Soil bacterial and fungal community structure across a range of unimproved and semi-improved upland grasslands," *Microbial Ecology*, vol. 50, no. 3, pp. 463–473, 2005.

[29] A. Frostegård and E. Bååth, "The use of phospholipid fatty acid analysis to estimate bacterial and fungal biomass in soil," *Biology and Fertility of Soils*, vol. 22, no. 1-2, pp. 59–65, 1996.

[30] T. A. Spedding, C. Hamel, G. R. Mehuys, and C. A. Madramootoo, "Soil microbial dynamics in maize-growing soil under different tillage and residue management systems," *Soil Biology and Biochemistry*, vol. 36, no. 3, pp. 499–512, 2004.

[31] C. Kaiser, A. Frank, B. Wild, M. Koranda, and A. Richter, "Negligible contribution from roots to soil-borne phospholipid fatty acid fungal biomarkers 18:2ω6,9 and 18:1ω9," *Soil Biology and Biochemistry*, vol. 42, no. 9, pp. 1650–1652, 2010.

[32] L. E. Jackson, F. J. Calderon, K. L. Steenwerth, K. M. Scow, and D. E. Rolston, "Responses of soil microbial processes and community structure to tillage events and implications for soil quality," *Geoderma*, vol. 114, no. 3-4, pp. 305–317, 2003.

[33] J. Moore-Kucera and R. P. Dick, "PLFA profiling of microbial community structure and seasonal shifts in soils of a Douglas-fir chronosequence," *Microbial Ecology*, vol. 55, no. 3, pp. 500–511, 2008.

[34] S. A. Boyle, R. R. Yarwood, P. J. Bottomley, and D. D. Myrold, "Bacterial and fungal contributions to soil nitrogen cycling under Douglas fir and red alder at two sites in Oregon," *Soil Biology and Biochemistry*, vol. 40, no. 2, pp. 443–451, 2008.

[35] J. A. Ceja-Navarro, F. N. Rivera-Orduña, L. Patiño-Zúñiga et al., "Phylogenetic and multivariate analyses to determine the effects of different tillage and residue management practices on soil bacterial communities," *Applied and Environmental Microbiology*, vol. 76, no. 11, pp. 3685–3691, 2010.

[36] R. A. Drijber, J. W. Doran, A. M. Parkhurst, and D. J. Lyon, "Changes in soil microbial community structure with tillage under long-term wheat-fallow management," *Soil Biology and Biochemistry*, vol. 32, no. 10, pp. 1419–1430, 2000.

[37] M. Ekenler and M. A. Tabatabai, "Responses of phosphatases and arylsulfatase in soils to liming and tillage systems," *Journal of Plant Nutrition and Soil Science*, vol. 166, no. 3, pp. 281–290, 2003.

[38] R. Lal, "Soil carbon sequestration impacts on global climate change and food security," *Science*, vol. 304, no. 5677, pp. 1623–1627, 2004.

[39] R. P. Dick, D. P. Breakwell, and R. F. Turco, "Soil enzyme activities and biodiversity measurements as integrative microbiological indicators," in *Methods for Assessing Soil Quality*, J. W. Doran and A. J. Jones, Eds., pp. 247–271, Soil Science Society of America, Madison,Wis, USA, 1996.

[40] C. E. Pankhurst, C. A. Kirkby, B. G. Hawke, and B. D. Harch, "Impact of a change in tillage and crop residue management practice on soil chemical and microbiological properties in a cereal-producing red duplex soil in NSW, Australia," *Biology and Fertility of Soils*, vol. 35, no. 3, pp. 189–196, 2002.

[41] S. D. Frey, E. T. Elliott, and K. Paustian, "Bacterial and fungal abundance and biomass in conventional and no-tillage agroecosystems along two climatic gradients," *Soil Biology and Biochemistry*, vol. 31, no. 4, pp. 573–585, 1999.

[42] P. W. Ramsey, M. C. Rillig, K. P. Feris, W. E. Holben, and J. E. Gannon, "Choice of methods for soil microbial community analysis: PLFA maximizes power compared to CLPP and PCR-based approaches," *Pedobiologia*, vol. 50, no. 3, pp. 275–280, 2006.

[43] R. S. Peixoto, H. L. C. Coutinho, B. Madari et al., "Soil aggregation and bacterial community structure as affected by tillage and cover cropping in the Brazilian Cerrados," *Soil and Tillage Research*, vol. 90, no. 1-2, pp. 16–28, 2006.

[44] N. C. Prevost-Boure, P. A. Maron, L. Ranjard et al., "Seasonal dynamics of the bacterial community in forest soils under different quantities of leaf litter," *Applied Soil Ecology*, vol. 47, no. 1, pp. 14–23, 2011.

[45] L. Ranjard, F. Poly, J. C. Lata, C. Mougel, J. Thioulouse, and S. Nazaret, "Characterization of bacterial and fungal soil communities by automated ribosomal intergenic spacer analysis fingerprints: biological and methodological variability," *Applied and Environmental Microbiology*, vol. 67, no. 10, pp. 4479–4487, 2001.

[46] R. Zornoza, C. Guerrero, J. Mataix-Solera, K. M. Scow, V. Arcenegui, and J. Mataix-Beneyto, "Changes in soil microbial community structure following the abandonment of agricultural terraces in mountainous areas of Eastern Spain," *Applied Soil Ecology*, vol. 42, no. 3, pp. 315–323, 2009.

[47] C. L. Lauber, M. S. Strickland, M. A. Bradford, and N. Fierer, "The influence of soil properties on the structure of bacterial and fungal communities across land-use types," *Soil Biology and Biochemistry*, vol. 40, no. 9, pp. 2407–2415, 2008.

[48] R. H. Findlay, "Determination of microbial community structure using phospholipid fatty acid profiles," in *Molecular Microbial Ecology Manual*, G. A. Kowalchuk II, Ed., pp. 983–1004, Kluwer Academic Publishers, Dordrecht, The Netherlands, 2004.

[49] E. A. Paul and F. E. Clark, *Soil Microbiology and Biochemistry*, Academic Press, San Diego, Calif, USA, 1996.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 826236, 9 pages
doi:10.1155/2012/826236

Research Article

Evolution of Soil Biochemical Parameters in Rainfed Crops: Effect of Organic and Mineral Fertilization

Marta M. Moreno,[1] Carmen Moreno,[1] Carlos Lacasta,[2] and Ramón Meco[3]

[1] *E. Ingenieros Agrónomos, Universidad de Castilla-La Mancha (UCLM), Ronda de Calatrava 7, 13071 Ciudad Real, Spain*
[2] *CSIC, Centro de Ciencias Medioambientales, Finca Experimental "La Higueruela", Santa Olalla, 45530 Toledo, Spain*
[3] *Servicio de Investigación Agraria, Consejería de Agricultura y Medio Ambiente de la Junta de Comunidades de Castilla-La Mancha, Pintor Matías Moreno 4, 45071 Toledo, Spain*

Correspondence should be addressed to Marta M. Moreno, martamaria.moreno@uclm.es

Received 4 December 2011; Accepted 10 April 2012

Academic Editor: Rosario García Moreno

In organic farming, crop fertilization is largely based on the decomposition of organic matter and biological fixation of nutrients. It is therefore necessary to develop studies conducted to know and understand the soil biological processes for the natural nutrient supplies. The effect of three fertilizer managements (chemical with synthetic fertilizers, organic with 2500 kg compost ha^{-1}, and no fertilizer) in a rainfed crop rotation (durum wheat-fallow-barley-vetch as green manure) on different soil biochemical parameters in semi-arid conditions was investigated. Soil organic matter, microbial biomass carbon, organic matter mineralization, CO_2 production-to-ATP ratio, and NO_3-N content were analysed. Fertilization was only applied to cereals. The results showed the scarce effect of the organic fertilization on soil quality, which resulted more dependent on weather conditions. Only soil organic matter and NO_3-N were affected by fertilization (significantly higher in the inorganic treatment, 1.28 g 100 g^{-1} and 17.3 ppm, resp.). Soil organic matter was maintained throughout the study period by the inclusion of a legume in the cropping system and the burying of crop residues. In fallow, soil microbial biomass carbon increased considerably (816 ng g^{-1}), and NO_3-N at the end of this period was around 35 ppm, equivalent to 100 kg N ha^{-1}.

1. Introduction

Conventional farming has been important for improving food to meet human demands but has been largely dependent on intensive inputs of synthetic fertilizers and pesticides [1, 2], both from an economic and energetic point of view. In recent years, the relationship between agriculture and the environment has changed, and concerns regarding the sustainability of agricultural production systems have come to the fore [3]. In this context, organic or ecological farming, focused on the environment and public health, is increasing worldwide [4]. Organic farming avoids the application of synthetic biocides and fertilizers [5, 6], promotes the use of renewable resources to prevent pollution [7], may reduce some negative effects attributed to conventional farming, and may have potential benefits in enhancing soil quality [2]. Thus, plant production in organic farming mainly depends on nutrient release as a function of the mineralization processes in soils. Therefore, to get an active soil microflora and an important amount of available nutrients is crucial in these productive systems, being the goal "fertilizing the soil rather than the plant" a priority among organic farmers to assure sufficient nutrient mineralization [8].

The incorporation of organic residues to soils causes a revival of biological and biochemical properties, stimulating microbial growth and metabolic activity as a result of the contributions of new labile carbon sources will serve as a substrate for soil microorganisms. This process is accompanied by an increase of the respiratory rate, releasing CO_2 as a reflect of the catabolic processes carried out from the organic supplies [9].

Soil quality is considered as "the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity,

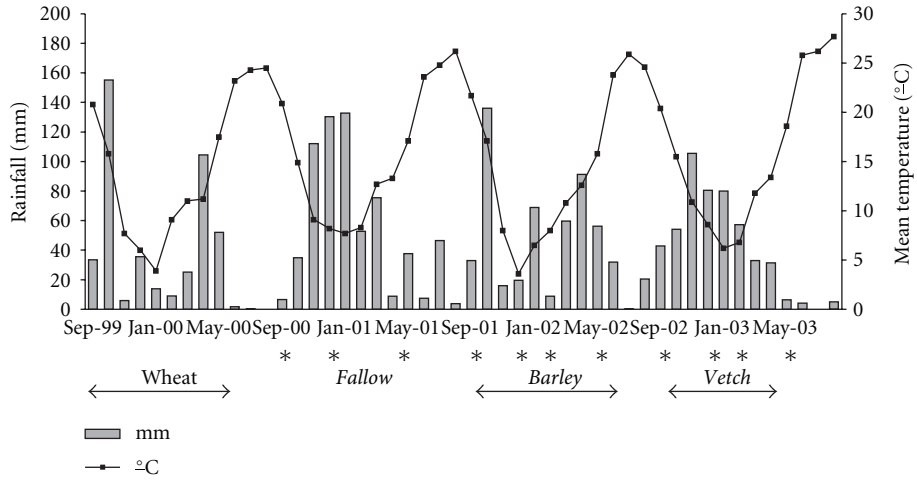


Figure 1: Distribution of rainfall and mean temperature during the field experiment. Crop rotation in the experimental parcels is shown (1999/2000–2002/2003). In italic the crop sequences considered in this study (2000/2001–2002/2003). *Dates of soil sampling.

maintain or enhance water and air quality, and support human health and habitation" [10]. Soil quality is responsible for determining soil ecological functions, such as decomposition and formation of soil organic matter [11], and determines the sustainability and productivity of agroecosystems [12]. Apart from the inherent soil quality, the dynamic soil quality changes in response to human use and management [8, 13].

Both the quantity and quality of labile soil organic matter (SOM) play a key role in the functioning and sustainability of agricultural systems, due to its significant impact on the physical, chemical, and biological soil properties [14, 15]. Quantity provides information about the amount of labile C substrate available to support microbial activity, while quality is related to SOM dynamics and nutrient (C or N) supply [14]. SOM is often used as an indicator of soil quality, it generally requires long periods (5–10 years) to detect changes as a result of possible alterations. In contrast, changes in soil microbial biomass carbon (SMBC) and nitrogen (SMBN) and in microbial functions rapidly reflect the impacts of the agricultural management and may change within shorter periods of time, before any changes in chemical or physical parameters are noticed [8, 16]. SMBC provides information on the dimensions of the biomass but not about their metabolic activity, being the soil respiration (CO_2 production) a measurement of this activity. In general, SMBC increases with increasing total organic carbon content, although this relationship may be affected by macroclimate, soil moisture, soil temperature regimes, and crop rotation [12].

After water, nitrogen is the most limiting factor for crop growth. For this reason, in the 1950–1990 period, the use of chemical nitrogen fertilizers increased about 10 times, leading to an important increase in cereal yields. However, the application of these fertilizers and other industrial and anthropogenic activities have altered the basic conditions of the natural nitrogen cycle and contributed to nitrate contamination of terrestrial and aquatic ecosystems with great risk to human health [17].

The aim of the present study was to assess the effect of different fertilization managements (organic, chemical, and no fertilization) applied to cereal crops in a rainfed crop rotation (durum wheat-fallow-barley-vetch as green manure) on soil microbiological properties, organic matter, and nitrate content in a semiarid environment over a 3-year period.

2. Material and Methods

2.1. Site Description and Experimental Design. Field experiments began in the season 1996/97 at La Higueruela Experimental Farm (4°26′ W, 40°04′ N, altitude 450 m), property of the Spanish National Research Council, Santa Olalla, Toledo, in the semiarid region of Castilla-La Mancha (central Spain).

Experiments were based on a typical 4-year crop rotation for semiarid environments: durum wheat (*Triticum durum* L.)-fallow-barley (*Hordeum vulgare* L.)-vetch (*Vicia sativa* L.) as green manure. In this study, the data corresponding to the period 2000/01 to 2002/03 were included (fallow-barley-vetch as green manure) (Figure 1), once finished a complete crop rotation, in order to avoid the possible interferences resulted from the previous uses of the soils.

The climate of the study region is semiarid Mediterranean, with a four-month drought period in summer coinciding with the highest temperatures. From 1975 to 1998, the average annual rainfall and temperature were 445 mm and 14.4°C, respectively.

The soil at the experimental site is classified as a Luvisol [18], with very differentiated horizons: A horizon (0–20 cm, sandy-loam); B horizon (20–60 cm, clay accumulation); C horizon (60–90 cm, calcium carbonate accumulation); R horizon (>90 cm). The main physical and chemical properties of the experiment parcel soils (0–20 cm) at the beginning of the study (year 2000) are presented in Table 1, summarized in a sandy-loam texture, a neutral pH, low organic matter and calcium levels, normal magnesium and potassium contents, and a high phosphorus level.

Trials were designed as randomised complete blocks with three managements of fertilization (chemical, organic, and no fertilization) and four replications. Elemental plots had a size of 100 m^2 (20 × 5 m). Fertilization was only applied to the cereal crops, as follows—(i) chemical (inorganic) fertilization: a total of 72-45-45 N-P_2O_5-K_2O kg ha^{-1}, distributed at sowing in a complex form (300 kg ha^{-1} of the 8-15-15 complex) and as a top-dressing at the tillering stage (Duramon 26, 13% ammoniacal N, 11% urea N, 200 kg ha^{-1}) (estimated from the average crop extractions and the yield crops in this area); (ii) organic fertilization: 2500 kg ha^{-1} of compost applied at sowing, allowed for organic farming use, with the following compounds expressed as kg ha^{-1}: organic matter, 1528; total nitrogen, 70; P_2O_5, 28; K_2O, 111; magnesium, 12; (iii) no fertilization treatment as control.

2.2. Plant Material and Culture Conditions. In the experiments, the crop varieties used were Hispanic for barley, Oscar for durum wheat, and Senda for vetch. All the crops were sown in November with a conventional sower (row distance 15 cm). The amounts of seeds used were 140 kg ha^{-1} for cereals and 100 kg ha^{-1} for vetch. The cultivation practices followed were similar to those employed by local growers, adapted to the type of soil and weed incidence, and so forth. Tillage consisted of two or three cultivator operations prior the crop sowings. In cereals, once the crops were combine-harvested after reaching physiological maturity in June-July, straw was uniformly incorporated to the same plots where it was produced with a disc harrow. The vetch was also incorporated into the soil with a disc harrow at flowering state (end of April-beginning of May) to be used as green manure. No weed control was practiced in any plot.

In the control treatment, the fertilization for the entire crop sequence only consisted in the N provided by the vetch crop and the cereal straw.

2.3. Soil Determinations. Soil samples 20 cm depth were taken in September and December 2000, May, September, and December 2001, February, May, September, and December 2002, and February and May 2003 (Figure 1), coinciding with the end of the meteorological seasons in Mediterranean climates.

In each sample, soil microbiological parameters, organic matter (Walkley-Black), and nitrate (NO_3-N) [19] contents were quantified. The soil microbiological parameters analysed, measured according to the methodology proposed by Maire et al. [20], were the following.

(i) SMBC: it expresses the total amount of microbial in soil, mainly bacteria and fungi, measured as the ATP (adenosine triphosphate) content. This parameter could not be measured in October 2001.

(ii) SOM mineralization (SOMM): this quantity is the sum of CO_2 production during the 15 days of incubation, expressed on an organic matter basis.

(iii) CO_2 production-to-ATP ratio, or specific activity of the microbial biomass. This ratio is similar to the more generally used metabolic quotient (qCO_2), which has been repeatedly used as a stress indicator [21], is related to the disponibility of nutrients, especially organic carbon, and it is often used as an indicator for assessing the influence of the environment conditions on soil microbial communities [22].

Table 1: Initial soil physical and chemical characteristics in the experimental parcel (year 2000).

Soil parameter	
Sand (2–0.05 mm) (g 100 g^{-1})	68.1
Silt (0.05–0.002 mm) (g 100 g^{-1})	14.8
Clay (<0.002 mm) (g 100 g^{-1})	17.1
pH (1 : 2.5 soil : water)	6.6
Organic matter (Walkley-Black) (g 100 g^{-1})	1.29
C/N	9.18
Phosphorus (Olsen) (mg kg^{-1})	186
Potassium (ammonium acetate extract) (mg kg^{-1})	168
Calcium (ammonium acetate extract) (mg kg^{-1})	1381
Magnesium (ammonium acetate extract) (mg kg^{-1})	150

2.4. Biomass Measurement. Aboveground biomass was controlled for the entire study, including crop plants and weeds. For this purpose, two samples per elemental plot consisting on three adjacent rows 0.5 m long (equivalent to 0.23 m^2) were taken at harvest and dried in a forced air oven at 60°C until constant weight. Each season, total biomass incorporated into the soil was calculated from the buried biomass of the previous crop (cereal straw, forage legumes, and root biomass, estimated in 20% of the total aboveground biomass according to unpublished data from the research group). In the organic treatment, the compost supply was also considered.

2.5. Statistical Analysis. Data were analysed by a two-way analysis of variance (ANOVA) with fertilization treatment and sampling date as main factors. The significance of the corresponding interaction between both factors was studied, and a Duncan's multiple-range test ($P < 0.05$) was applied to the significant results in each case. The statistical analysis was performed with the statistical package InfoStat 2007, professional version.

3. Results

3.1. Climate. The monthly rainfall and mean temperature during the period 1999/2000–2002/2003 are shown in Figure 1. The average seasonal (1 September–31 August) rainfall was 532.2 mm, irregularly distributed intra-annually in timing and amount. Rainfall was similar in all autumns (average 184 mm), but however it varied considerably in the other seasons; thus, rainfall in summer ranged from 2.0 mm in 2000/2001 to 57.7 mm in 2001/2002, in winter from 58 mm in 1999/2000 to 315.9 mm in 2000/2001, and in spring from 70.5 mm in 2002/2003 to 207.1 mm in

2001/2002. The average annual temperature was 15.2°C (winter, 6.9°C; spring, 13.8°C; summer, 24.2°C; autumn, 15.2°C).

3.2. Crop Biomass and Organic Matter Supplies. Aboveground vegetal biomass in the different crops was significantly affected by the fertilization treatment over the study period (Table 2), being in all cases higher in the chemical treatment. In relation to the total inputs of organic matter to the soil (Table 3), no differences among treatments were found; however, it should be noticed that the compost dose was also considered as an input in the organic treatment, being lower the amount of vegetal biomass incorporated into the soil in this treatment, especially from the barley crop.

3.3. Soil Biological Parameters. The ANOVA for the soil parameters analysed is summarized in Table 4. In general, no interactions between the fertilization treatment and the sampling date were significant in any case, which allowed to study the simple effects of each factor. Differences in all soil variables analysed were observed among the sampling dates, mainly as result of variation in weather conditions; however, the effect of the fertilization treatment was only significant on the SOM and NO_3-N variables.

The soil results obtained throughout the whole experiment are presented in Table 5. Averaged across sampling dates, SOM was significantly higher with chemical fertilization (1.28 g 100 g^{-1}) than with compost or no fertilization (1.19 and 1.20 g 100 g^{-1}, resp.). In the inorganic treatment, the highest SOM corresponded to the barley crop, when fertilization took place.

SOMM ranged from 358 μg OM g^{-1} 15 d^{-1} in the inorganic treatment to 337 μg OM g^{-1} 15 d^{-1} in the control (Table 5), although no differences among treatments were found. Throughout the study period, it is remarkable the high average SOMM reached in Sept 2000 (721 μg OM g^{-1} 15 d^{-1}). During the fallow period, SOMM was high and practically constant (Dec 2000 and May 2001). The lowest values were obtained at the end of this season (Sept 2001, 184 μg OM g^{-1} 15 d^{-1}).

In relation to SMBC, the highest levels were recorded in spring and autumn, and especially during the fallow period (Table 5). After May 2001, once the maximum SMBC was reached, an important decreased in this parameter was observed and remained during the following samplings. However, SMBC increased again in September 2002, especially in the organic fertilizer treatment, and in May 2003, coinciding with the end of the vetch crop.

The efficiency of soil microorganisms (CO_2 to ATP ratio) was higher in the cooler months during the vetch crop (Table 5, Figure 1), when the microbial biomass decreased and the soil organic matter mineralization (measured at laboratory) presented similar values than those reached in spring and autumn.

The average soil NO_3-N content was significantly higher in inorganically (17.3 ppm) than in organically (14.1 ppm) or no-fertilized soils (15.0 ppm) (Table 5). However, despite the general results in relation to the averaged soil NO_3-N contents, statistical differences among treatments were only observed in May 2002, probably as result of the low rainfall registered in the previous winter (97.1 mm) (Figure 1), which could limit the leaching of nitrates and therefore to improve the chemical fertilizer efficiency. NO_3-N remained in low levels during fallow and reached the maximum values at the end of this period in the three fertilization treatments (34.8 ppm average, equivalent to 100 kg ha^{-1} in the 0.20 m surface soil layer) (Table 5). In general, during the barley growing season, the soil NO_3-N content was higher with chemical fertilization than in the other two treatments. In each case, soil NO_3-N levels decreased as the crop cycle advanced as consequence of plant extractions.

4. Discussion

It is well documented that the treatments which supply more carbon to the system will generate a higher amount of soil organic matter and therefore a higher soil microbial biomass and microbial activity [2, 12, 23–25]. However, in this study the increase of the carbon supply did not result in an increase of the soil parameters measured (SOM, SOMM, SMBC, and CO_2 to ATP ratio), being more affected by the specific meteorological conditions than for the type or amount of the fertilizers employed.

During the barley crop, soils did not receive external organic matter supplies from crop residues because this crop followed a fallow period, and only the organic treatment received the compost as organic supply. However, this external input of organic matter did not result in an increase of SOM. It could be explained by the dry winter which characterized that season (2001/2002), appropriate conditions for limiting the leaching of nitrates and getting a good efficiency of chemical fertilizers, and therefore for obtaining a great plant biomass production which could later be incorporated into the soil in the inorganic treatment, effect which was also observed for vetch crop. For this reason, the total biomass accumulated was higher in the chemical than in the other fertilization treatments, although the total inputs of organic matter to the soils were similar among them. Pardo et al. [26], however, did not find differences among chemical, organic and no-fertilizer treatments in cereal and vetch biomass in a similar approach.

In relation to SOMM, the highest values were reached in the first sampling (higher CO_2 release rate) because at that date all crop residues from the previous wheat crop were still undecomposed (or only partially decomposed) as a result of the scarce rainfall registered in the previous summer (2.0 mm). Consequently, when the soil sample was placed at the laboratory under the appropriate temperature and humidity conditions for biological activity, the large amount of carbon contained in the sample produced a large CO_2 release. Thus, SOMM at the laboratory was always higher at the end of summer provided that a source of organic matter was previously added, because in these cases the soil sample contained a certain amount of carbon which could be mineralised. During the fallow period, SOMM was high and practically constant, when crop residues

Table 2: Aboveground vegetal biomass (kg ha^{-1}) in the different crops over the period 1999–2003.

Treatment	1999/2000 Wheat	2000/2001 *Fallow*	2001/2002 *Barley*	2002/2003 *Vetch*	Total accumulated[1]
Chemical	9313 a	0	14201 a	3032 a	17233 a
Organic	9988 a	0	10419 b	2406 b	12825 b
No fertilization	10073 a	0	12668 ab	2098 b	14766 b
Average	9792	0	12429	2512	14941

In italic the crop sequence considered in this study ([1]also for total accumulated data). Different small letters in the same column indicate significant differences at $P < 0.05$.

Table 3: Total inputs of organic matter (kg ha^{-1}) incorporated into the soil over the period 2000–2003.

Treatment	2000/2001 Fallow	2001/2002 Barley	2002/2003 Vetch	Total accumulated
Chemical	8049 a	0	10740 a	18789 a
Organic	8792 a	2500	8024 b	19316 a
No fertilization	9267 a	0	9358 ab	18625 a
Average	8703	833	9374	18910

Different small letters in the same column indicate significant differences at $P < 0.05$.

remained undecomposed in the soil and, however, under the appropriate conditions at the laboratory, the decomposition process began with the corresponding CO_2 emission. The low SOMM measured at the end of the fallow period indicated that the mineralization process of the crop residues had already taken place.

SMBC was also affected by the weather conditions; thus, the highest levels were recorded in spring and autumn, when temperature and humidity conditions were suitable for the microorganisms to grow, and especially during the fallow period due to the presence of organic carbon from the previous wheat crop. This effect was also observed at the beginning of the vetch crop and especially in the organic fertilizer treatment (appropriate humidity and temperature conditions, and organic carbon from both the barley crop and the organic fertilization). However, the low temperatures reached in winter led to a low activity of soil microorganisms. In summer, high SMBC was only observed when it was rainy and there were crop residues rich in carbon. During the barley crop, the low SMBC values were probably as result of the lack of organic carbon from the previous fallow period and the low temperatures reached that season.

The analysis of the results obtained in this trial suggests that inputs of 2500 kg ha^{-1} of compost (organic fertilization) did not lead to an improvement of the soil biochemical parameters. These results contrast with most studies developed in similar environments; thus, García-Galavís et al. [27] and Marinari et al. [24] indicated that soils organically fertilized usually had more active microbial populations than soils with chemical fertilization, which means an improvement in soil quality. However, most of these studies were performed with high amounts of compost, greater than 20 t ha^{-1}, and often in irrigated crops, which even resulted in NO_3-N plant levels higher than the maximum allowed [28].

Soil moisture is an important factor for microorganisms growing and plant residue decomposition [29]. When high soil moisture is present, then there is less oxygen available for microbial growing. In the same way, Calabria et al. [30] found the lowest SOMM rates under dryland conditions. In autumn and spring, inorganic N is immobilized (increase of SMBC), and once the carbon present in plant residues is consumed, microbial biomass decreases and nitrogen releases when moisture levels for nitrification are appropriate. In dry summers the nitrification process is limited, being reactivated during the following autumn, and decreases again in winter due to low temperatures. The results obtained by Farrus et al. [28] in different lettuce growing seasons under organic and no fertilizer treatments also support the effect of the meteorological conditions on soil chemical and biological parameters; thus, they did not find differences among treatments when lettuce grew in spring-summer because temperatures favored the organic matter mineralization process.

NO_3-N remained in low levels during fallow either because it was still immobilized, forming organic compounds (not mineralized), or because it had become part of the microbial biomass. This parameter reached the highest values at the end of summer, with levels around 35 ppm (equivalent to 100 kg N ha^{-1}) regardless of the fertilization treatment, enough amounts for rainfed cereal crops in semiarid environments (2000 kg ha^{-1} grain yields) [3]. It explains why the cereal-fallow rotation has been historically used by farmers in rainfed crops, even before the appearance of external fertilizers.

Application of compost did not result in an increase of vegetal biomass and soil NO_3-N content, which indicates that the organic nitrogen supplied by the compost (70 kg ha^{-1}) was not transformed into nitrate, in agreement with Pardo et al. [26]. Van Faassen and Van Dijk [31] also reported that mineralization of organic nitrogen from manures is rather variable depending on manure type and soil properties.

In general, SOM at the end of the study maintained their initial levels, in concordance with Pardo et al. [26], which indicates that crop rotations and burying the vetch

Table 4: Summary of the analysis of variance (ANOVA) for soil biological variables over the experiment.

Source of variation	SOM[1]			SOMM[2]			SMBC[3]			CO_2 to ATP ratio			NO_3-N		
	d.f.	M.S.	*F*	d.f.	M.S.	*F*	d.f.	M.S.	*F*	d.f.	M.S.	*F*			
Fertilization (FT)	2	0.12	3.22*	2	4952.73	0.44	2	23063.92	1.42	2	12.60	2.25	2	118.33	7.46**
Sampling date (SD)	10	0.07	2.02*	10	261707.83	23.40**	9	662241.99	40.90**	9	87.53	15.61**	10	766.00	48.29**
FT × SD	20	0.02	0.61	20	7208.98	0.64	18	19833.67	1.22	18	5.28	0.94	20	12.25	0.77
Error	99	0.04		99	11183.04		90	16192.64		90	5.61		99	15.86	
Total	131			131			119			119			131		

[1]SOM: soil organic matter. [2]SOMM: soil organic matter mineralization. [3]SMBC: soil microbial biomass carbon.
d.f.: degrees of freedom. M.S.: mean square. *F*: *F*-statistic. * Significant at $P < 0.05$. ** Significant at $P < 0.01$.

Table 5: Evolution of the soil biological parameters (SOM[1], SOMM[2], SMBC[3], CO_2 to ATP ratio, NO_3-N) according to the fertilization treatments over the field experiment.

Soil parameter	Treatment	Sept-00	Dec-00	May-01	Sept-01	Dec-01	Feb-02	May-02	Sept-02	Dec-02	Feb-03	May-03	Average
SOM[1] (g 100 g^{-1})	Chemical	1.23 a	1.23 a	0.99 a	1.23 a	1.31 a	1.43 a	1.43 a	1.40 a	1.25 a	1.29 a	1.31 a	1.28 a
	Organic	1.17 a	1.21 a	1.01 a	1.17 a	1.28 a	1.14 b	1.17 b	1.26 a	1.35 a	1.21 a	1.12 a	1.19 b
	No fertilization	1.15 a	1.12 a	1.10 a	1.14 a	1.15 a	1.27 ab	1.23 b	1.27 a	1.28 a	1.16 a	1.31 a	1.20 b
	Average	1.18 AB	1.19 AB	1.03 B	1.18 AB	1.25 A	1.28 A	1.28 A	1.31 A	1.29 A	1.22 A	1.25 A	1.22
SOMM[2] (μg OM g^{-1} 15 d^{-1})	Chemical	739 a	280 a	260 a	200 a	253 a	260 a	334 a	304 a	380 a	412 a	520 a	358 a
	Organic	793 a	259 a	281 a	174 a	257 a	241 a	282 ab	362 a	340 a	324 a	495 a	346 a
	No fertilization	631 a	315 a	337 a	178 a	224 a	270 a	258 b	255 a	367 a	430 a	447 a	337 a
	Average	721 A	284 DE	293 DE	184 E	245 DE	257 DE	291 DE	307 CDE	362 CD	388 C	487 B	347
SMBC[3] (ATP, ng g^{-1})	Chemical	970 a	561 a	823 a	—	224 a	252 a	299 a	330 b	205 a	353 a	462 a	448 a
	Organic	757 b	381 b	790 a	—	243 a	197 a	222 a	569 a	215 a	214 a	453 a	404 a
	No fertilization	856 ab	409 b	835 a	—	248 a	265 a	265 a	391 ab	239 a	390 a	531 a	443 a
	Average	861 A	450 B	816 A	—	238 C	238 C	262 C	430 B	219 C	319 C	482 B	432
CO_2 to ATP ratio (μg CO_2 μg^{-1} ATP)	Chemical	4.85 a	2.61 a	1.81 a	—	7.79 a	7.41 a	7.97 a	6.81 a	13.84 a	7.82 a	7.77 a	6.87 a
	Organic	6.07 a	3.73 a	2.10 a	—	7.03 a	8.25 a	8.22 a	4.42 a	10.36 a	11.61 a	7.33 a	6.91 a
	No fertilization	4.60 a	3.65 a	2.28 a	—	6.05 a	7.21 a	6.53 a	4.86a a	9.29 a	8.84 a	5.86 a	5.92 a
	Average	5.17 DE	3.33 EF	2.07 F	—	6.95 CD	7.62 BC	7.57 BC	5.36 D	11.16 A	9.42 AB	6.98 CD	6.57
NO_3-N (ppm)	Chemical	6.2 a	9.4 a	8.3 a	35.8 a	25.3 a	23.6 a	17.4 a	15.5 a	14.6 a	14.6 a	19.3 a	17.3 a
	Organic	4.6 a	9.2 a	7.6 a	32.2 a	22.0 a	15.8 a	12.3 b	12.0 a	12.2 a	11.5 a	15.5 a	14.1 b
	No fertilization	4.7 a	10.4 a	7.7 a	36.5 a	17.6 a	16.8 a	12.3 b	15.7 a	13.3 a	13.1 a	17.2 a	15.0 b
	Average	5.2 G	9.7 F	7.8 FG	34.8 A	21.6 B	18.7 BC	14.0 DE	14.4 DE	13.4 E	13.1 E	17.3 CD	15.5

[1]SOM: soil organic matter. [2]SOMM: soil organic matter mineralization. [3]SMBC: soil microbial biomass carbon.
Different small letters in the same column for each variable indicate significant differences at $P < 0.05$. Different capital letters in the same row indicate significant differences at $P < 0.05$.

and the crop residues after harvesting help to maintain the soil fertility.

5. Conclusions

The study of different soil biochemical parameters in a rainfed crop-rotation in a sandy-loam soil indicates that the improvement of soil quality depended more on the weather conditions than on the type of fertilizers used. Organic fertilization consisting of 2500 kg compost ha^{-1} did not improve the quantity and quality of soil organic matter, being similar to those observed when chemical or no fertilizer were used. However, the inclusion of a legume in the cropping system and the burying of crop residues after harvesting were enough to maintain the initial soil organic matter levels. Including a fallow period in a crop rotation is advisable because the soil microorganism populations increased (appropriate moisture and temperature conditions and not competition with crops). Additionally, at the end of fallow soil NO_3-N contents were high and enough for cereal growth in semiarid environments.

Acknowledgments

The research was supported by projects INIA SC 96-081 and RTA 01-108. The authors would like to thank Luis Martín de Eugenio and José Ramón Vadillo for performing the field operations and helping to compile the corresponding data over the study period.

References

[1] C. Tu, J. B. Ristaino, and S. Hu, "Soil microbial biomass and activity in organic tomato farming systems: effects of organic inputs and straw mulching," *Soil Biology and Biochemistry*, vol. 38, no. 2, pp. 247–255, 2006.

[2] A. S. F. Araújo, V. B. Santos, and R. T. R. Monteiro, "Responses of soil microbial biomass and activity for practices of organic and conventional farming systems in Piauí state, Brazil," *European Journal of Soil Biology*, vol. 44, no. 2, pp. 225–230, 2008.

[3] M. M. Moreno, C. Lacasta, R. Meco, and C. Moreno, "Rainfed crop energy balance of different farming systems and crop rotations in a semi-arid environment: results of a long-term trial," *Soil and Tillage Research*, vol. 114, no. 1, pp. 18–27, 2011.

[4] S. Melero, J. C. R. Porras, J. F. Herencia, and E. Madejon, "Chemical and biochemical properties in a silty loam soil under conventional and organic management," *Soil and Tillage Research*, vol. 90, no. 1-2, pp. 162–170, 2006.

[5] C. A. Helander and K. Delin, "Evaluation of farming systems according to valuation indices developed within a European network on integrated and ecological arable farming systems," *European Journal of Agronomy*, vol. 21, no. 1, pp. 53–67, 2004.

[6] U. Jørgensen, T. Dalgaard, and E. S. Kristensen, "Biomass energy in organic farming—the potential role of short rotation coppice," *Biomass and Bioenergy*, vol. 28, no. 2, pp. 237–248, 2005.

[7] IFOAM Norms, *International Federation of Organic Agriculture Movements*, 2002.

[8] A. Fließbach and P. Mäder, "Microbial biomass and size-density fractions differ between soils of organic and conventional agricultural systems," *Soil Biology and Biochemistry*, vol. 32, no. 6, pp. 757–768, 2000.

[9] J. C. García Gil, *Efectos residuales y acumulativos producidos por la aplicación de compost de residuos urbanos y lodos de depuradoras sobre agrosistemas mediterráneos degradados*, Ph.D. thesis, Science Faculty, Autonomous University of Madrid, Madrid, Spain, 2001.

[10] D. L. Karlen, M. J. Mausbach, J. W. Doran, R. G. Cline, R. F. Harris, and G. E. Schuman, "Soil quality: a concept, definition, and framework for evaluation," *Soil Science Society of America Journal*, vol. 61, no. 1, pp. 4–10, 1997.

[11] J. W. Doran, M. Sarrantonio, and M. A. Liebig, "Soil health and sustainability," in *Advances in Agronomy*, D. L. Sparks, Ed., vol. 56, pp. 25–37, Academic Press, San Diego, Calif, USA, 1996.

[12] S. Melero, E. Madejón, J. C. Ruiz, and J. F. Herencia, "Chemical and biochemical properties of a clay soil under dryland agriculture system as affected by organic fertilization," *European Journal of Agronomy*, vol. 26, no. 3, pp. 327–334, 2007.

[13] M. R. Carter, E. G. Gregorich, D. W. Anderson, J. W. Doran, H. H. Janzen, and F. J. Pierce, "Chapter 1 Concepts of soil quality and their significance," *Developments in Soil Science*, vol. 25, pp. 1–19, 1997.

[14] E. E. Marriott and M. Wander, "Qualitative and quantitative differences in particulate organic matter fractions in organic and conventional farming systems," *Soil Biology and Biochemistry*, vol. 38, no. 7, pp. 1527–1536, 2006.

[15] D. Rotenberg, A. J. Wells, E. J. Chapman, A. E. Whitfield, R. M. Goodman, and L. R. Cooperband, "Soil properties associated with organic matter-mediated suppression of bean root rot in field soil amended with fresh and composted paper mill residuals," *Soil Biology and Biochemistry*, vol. 39, no. 11, pp. 2936–2948, 2007.

[16] A. P. Silva, L. C. Babujia, J. C. Franchini, R. A. Souza, and M. Hungria, "Microbial biomass under various soil- and crop-management systems in short- and long-term experiments in Brazil," *Field Crops Research*, vol. 119, no. 1, pp. 20–26, 2010.

[17] M. Fernández-Pascual, M. de María, and M. R. de Felipe, "Fijación biológica de nitrógeno: factores limitantes," in *Ciencia y Medio Ambiente*, F. Valladares, Ed., pp. 195–202, CSIC, Madrid, Spain, 2002.

[18] USDA, *Keys to Soil Taxonomy, Tenth Edition*, Soil Survey Staff, United States Department of Agriculture NRCS, 2006.

[19] J. H. Wetters and K. L. Uglum, "Direct spectrophotometric simultaneous determination of nitrite and nitrate in the ultraviolet," *Analytical Chemistry*, vol. 42, no. 3, pp. 335–340, 1970.

[20] N. Maire, D. Borcard, E. Laczkó, and W. Matthey, "Organic matter cycling in grassland soils of the Swiss Jura mountains: biodiversity and strategies of the living communities," *Soil Biology and Biochemistry*, vol. 31, no. 9, pp. 1281–1293, 1999.

[21] D. A. Wardle and A. Ghani, "A critique of the microbial metabolic quotient (qCO_2) as a bioindicator of disturbance and ecosystem development," *Soil Biology and Biochemistry*, vol. 27, no. 12, pp. 1601–1610, 1995.

[22] T. H. Anderson and K. H. Domsch, "Application of ecophysiological quotients (qCO_2 and qD) on microbial biomasses from soils of different cropping histories," *Soil Biology and Biochemistry*, vol. 22, no. 2, pp. 251–255, 1990.

[23] E. Madejón, R. López, J. M. Murillo, and F. Cabrera, "Agricultural use of three (sugar-beet) vinasse composts: effect

on crops and chemical properties of a Cambisol soil in the Guadalquivir river valley (SW Spain)," *Agriculture, Ecosystems and Environment*, vol. 84, no. 1, pp. 55–65, 2001.

[24] S. Marinari, G. Masciandaro, B. Ceccanti, and S. Grego, "Evolution of soil organic matter changes using pyrolysis and metabolic indices: a comparison between organic and mineral fertilization," *Bioresource Technology*, vol. 98, no. 13, pp. 2495–2502, 2007.

[25] Q. R. Wang, Y. C. Li, and W. Klassen, "Changes of soil microbial biomass carbon and nitrogen with cover crops and irrigation in a tomato field," *Journal of Plant Nutrition*, vol. 30, no. 4, pp. 623–639, 2007.

[26] G. Pardo, J. Cavero, J. Aibar, and C. Zaragoza, "Nutrient evolution in soil and cereal yield under different fertilization type in dryland," *Nutrient Cycling in Agroecosystems*, vol. 84, no. 3, pp. 267–279, 2009.

[27] P. A. García-Galavís, C. Santamaría, J. C. Ruiz, and A. Daza, "Efecto beneficioso de la agricultura ecológica sobre los microorganismos del suelo," in *Agroecología: Referente Para la Transición de los Sistemas Agrarios, VI SEAE Congress*, pp. 1143–1151, Madrid, Spain, 2004.

[28] E. Farrus, M. Adrover, A. Forss, and J. Vadell, "Comparación de tres fuentes de materia orgánica sobre las características del suelo," in *Agroecología: Referente Para la Transición de los Sistemas Agrarios, VI SEAE Congress*, pp. 1111–1123, Madrid, Spain, 2004.

[29] Z. Huang, Z. Xu, and C. Chen, "Effect of mulching on labile soil organic matter pools, microbial community functional diversity and nitrogen transformations in two hardwood plantations of subtropical Australia," *Applied Soil Ecology*, vol. 40, no. 2, pp. 229–239, 2008.

[30] C. Calabria, I. Bautista, and M. Valero, "Índices biológicos de disponibilidad de nitrógeno en suelos de la Comunidad Valenciana," in *Agroecología: Referente para la Transición de los Sistemas Agrarios, VI SEAE Congress*, pp. 1017–1032, Madrid, Spain, 2004.

[31] H. G. Van Faassen and H. Van Dijk, "Manure as a source of nitrogen and phosphorus in soils," in *Animal Manure on Grassland and Fodder Crops, Fertilizer or Waste Development in Plant and Soil Sciences*, H. G. van der Meer, R. J. Unwen, T. A. van Dijk, and G. C. Ennik, Eds., pp. 27–45, Martinus Nijhoff Publishers, Dordrecht, The Netherlands, 1987.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 157068, 10 pages
doi:10.1155/2012/157068

Research Article

Nitrogen and Phosphorus Changes in Soil and Soil Water after Cultivation

Mark Watkins, Hayley Castlehouse, Murray Hannah, and David M. Nash

Future Farming Systems Research Division, Department of Primary Industries, 1301 Hazeldean Road Ellinbank, VIC 3821, Australia

Correspondence should be addressed to David M. Nash, david.nash@dpi.vic.gov.au

Received 24 November 2011; Accepted 14 March 2012

Academic Editor: Rosario García Moreno

Untilled dairy pasture has the potential to release more phosphorus to the environment than a regularly ploughed pasture. In this paper we report the initial results of a study comparing the effects of cultivation, phosphorus (P) fertiliser (10, 35, and 100 kg P/ha), and two types of vegetation (ryegrass (*Lolium perenne*) or ryegrass mixed with clover (*Trifolium repens*)) in a randomised complete block design. Phosphorus was measured in soil samples taken from depths of 0–20 mm and 0–100 mm. Waters extracted from the 0–20 mm samples were also analysed. In all cases, the P concentrations (Olsen P, Colwell P, Total P, $CaCl_2$ extractable P, Dissolved Reactive P, and Total Dissolved P) in the top 20 mm declined with ploughing. Dissolved Reactive P measured in the soil water was 70% less overall in the ploughed plots compared with the unploughed plots, and by 35 weeks after P treatments the decrease in Dissolved Reactive P was 66%. The effects of the fertiliser and pasture treatments were inconclusive. The data suggest that ploughing can lower the risk of P exports from intensive dairy farms in the trial area.

1. Introduction

Excessive phosphorus (P) in surface waters is a major environmental issue in Australia [1]. In many freshwater ecosystems, P limits primary production and excessive P inputs contribute to eutrophication and the development of cyanobacterial blooms [2] which can be hazardous to human health [3–5]. This is especially true in the Gippsland Region of south-eastern Australia which contains the Tambo, Mitchell, Thomson, and Latrobe rivers and an estuarine lakes system of international significance, the Gippsland Lakes [6]. Agricultural enterprises, particularly dairy farms [7–9], contribute to excessive P concentrations and the associated increasing prevalence of algal blooms in the Gippsland Lakes [10, 11]. For the Victorian coastal plains (Gippsland through to Melbourne at less than 200 m elevation), annual 75th percentile targets for total P (TP) and total nitrogen (TN) have been established for nutrients in rivers and streams; these targets are 0.045 mg/L and 0.6 mg/L, respectively [12].

A broad range of strategies have been instigated to lessen P inflows to the Gippsland Lakes [13, 14]. These strategies include on-farm measures such as minimising water lost to drains, construction of reuse ponds, stock exclusion from waterways, control of soil erosion, and management of fertiliser application and timing, as well as various off-farm measures to reuse irrigation drain water and minimise inputs from stream erosion, forestry activities, and industrial discharges. Prominent amongst these strategies has been the implementation of Best Management Practices (BMPs) to lessen dissolved P exports from dairy pastures. However, it is becoming increasingly clear that current BMPs [15] will not achieve the targeted 40% reduction [14] in P exports, especially from dairy farms.

Numerous studies have demonstrated that, surface-applied fertilisers, decaying plant material, and wastes from grazing animals increase P and N concentrations [16] and lower P adsorptive capacity in surface soils [17–20], thereby increasing P export potential [21, 22]. Consequently, in well-managed dairy pastures in the Gippsland Region P and N are generally exported in dissolved ($<0.45\,\mu$m) rather than particulate forms [9, 17, 23] with the times between fertiliser application and runoff, and grazing and runoff, and a year-dependant base component, explaining most of the between storm variation in P [24–26]. Cultivation is one way of lessening soil nutrient stratification, increasing P adsorption

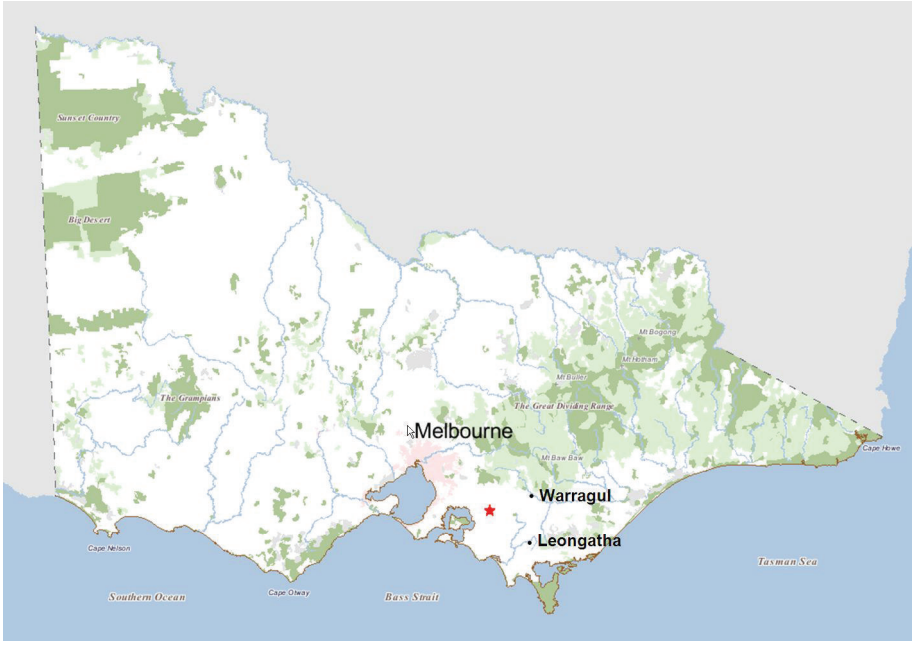


Figure 1: Location of the experimental site at Poowong, VIC, Australia.

near the soil surface and potentially lowering P exports [27–30].

Studies on laser grading, an extreme form of cultivation, have been carried out in the Gippsland area. In the initial study, three years after laser grading, surface soil P adsorption had increased, soil test P decreased, soil water P and N concentrations decreased and total P (TP) and total N (TN) in irrigation runoff decreased by 41% and 36%, respectively [31]. These studies were repeated on commercial farms. In the first year, cultivation had lowered soil and soil water P concentrations (in some cases by more than 50%) and lowered P concentrations in runoff [32]. However, while the P export potential (estimated from soil tests) from cultivation remained lower for the duration of the study, albeit at a declining margin, there were no treatment effects on overland flow P concentrations after the first year. The lack of a treatment effect when the "background" sources of P in the soil had declined appears to be primarily the result of grazing effects and variability between sites, bays, and years, particularly during the drought experienced by this region at the time. There were no effects of cultivation on N exports in overland flow or soil water, presumably due to N fertiliser additions that continued during the study.

Pastures used for dairying in Gippsland are commonly a mix of ryegrass (*Lolium spp.*) and clover (*Trifolium repens*). Clover fixation of atmospheric nitrogen enhances soil N fertility, but clover requires a higher soil P (Olsen P of >15 mg/kg) compared with ryegrass (Olsen P > 12 mg/kg) [33, 34]. This suggests that it may be possible to grow adequate ryegrass pasture at lower soil P, and hence lower P export potential, if nitrogen fertiliser replaces the nitrogen fixing function of clover. The lower required soil P concentration would thereby enhance the benefits of cultivation in mitigating P exports. In this study we examined the effects of destratification of soils by mouldboard ploughing along with the effects of three P fertiliser application rates on pasture that is either a monoculture of ryegrass or mixed sward of ryegrass and clover. The quality and yield of the pasture were assessed for each treatment in addition to measures of P and N export potential.

It was hypothesised that P concentrations in the top 20 mm soil could be reduced by cultivation and that lower application of P fertiliser would reduce P in soil water. In addition, it was hypothesised that a ryegrass monoculture may be more viable than a ryegrass clover mix in a lower soil P environment and that differences in soil N and P may be observed between these two vegetative types.

2. Method

The experimental site was located near Poowong, Victoria, Australia (−38.2782°, 145.7404°, Figure 1) in an undulating low hills landscape with a slope of 4%. The soil type is typically a Grey Dermosol according to the Australian Soil Classification [35], but in poorer drained lower lying areas can be a Dermosolic Hydrosol (indicating that the soil profile is saturated for a number of months in most years). The surface soil texture is a light fine sandy clay loam. The average annual rainfall in the Poowong area is *c.* 1100 mm (30 year mean to 1991 is 1143 mm [36]). Rainfall and irrigation for the trial period are shown in Figure 2.

The experiment had a randomised complete block design with 12 treatments consisting of the complete factorial combinations of two types of sod preparation (mouldboard ploughed or unploughed), two types of vegetation (ryegrass monoculture or a mixed sward of white clover and ryegrass), and three rates of phosphorus fertiliser (10, 35, 100 kg/ha). The trial site (c. 0.5 ha) was divided up into three blocks of

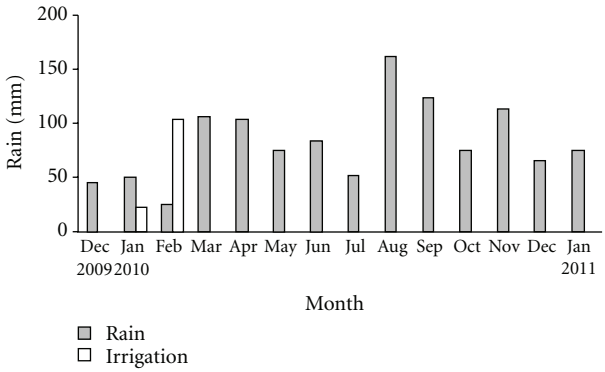


Figure 2: Monthly rainfall and irrigation for the trial period.

twelve plots (12 × 6 m). Blocks were arranged with a 3 m buffer between each block.

Prior to treatment imposition, plots were sprayed with glyphosphate and heavily grazed to remove vegetation. Ploughed plots were mouldboard ploughed on November 20, 2009. As is common in the area, the ploughed plots were then levelled using a series of iron bars drawn behind a tractor on November 24, 2009. All plots were seeded the following day with 25 kg/ha perennial ryegrass (*Lolium perenne*) for the monoculture plots, and 25 kg/ha ryegrass and 5 kg/ha white clover (*Trifolium repens*, Kopu II) for the mixed sward plots. To aid establishment, a basal application of fertiliser was applied four weeks after ploughing at a rate of 15 kg/ha P, 40 kg/ha K (Incitec Pivot SuperfectPot 2 and 1 Mo: 5.9% P, 16.7% K, 7.3% S, 0.025% Mo) and 30 kg/ha N (Incitec Pivot granulated urea: 46% N). Fertiliser P (Incitec Pivot Triple Super: 20.7% N, 1.0% S) treatments were applied on March 30, 2010. All plots received a basal fertiliser application for sulphur (60 kg/ha Gypsum) and potassium (Incitec Pivot Muriate of Potash 20 kg/ha) on the same day. Rust occurred in the ryegrass, and pure ryegrass plots subsequently received supplementary applications of urea to stimulate vegetative growth and alleviate the symptoms (February 4 and June 4, 2010). The herbicide MCPA (2-methyl-4-chlorophenoxyacetic acid) was applied to ryegrass only plots to control unwanted paspalum on 26 March and 27 July 2010. MCPA could not be applied to the mixed sward plots as clover is sensitive to MCPA. It is of note that sowing was delayed due to wet conditions in late spring (which prevented ground preparation) and dry conditions during the summer of 2009-2010 necessitated irrigation to establish the pasture (122 mm applied in January and February 2010, Figure 2).

Commencing fifteen weeks after seeding, plots were grazed twelve times with 500 cows for an average of 45 minutes (i.e., until the animals stopped actively grazing). After grazing, manure was immediately removed by shovel from each plot to minimise the effects of manure on between plot and within plot variation. Where possible, on the following day remaining vegetation was mown to 80 mm pasture height to promote even growth in keeping with local farming practice [37]. A Gianni Ferrari GT20 catcher ride-on mower (JSB Equipment Pty Ltd, Carrum Downs, Australia) was used to cut the pasture and all grass clippings were removed off-site. Mowing was not possible between September 2010 and January 2011 due to very wet conditions and animal treading damage (i.e., pugging or poaching) which created an uneven soil surface. A roller was used to address these problems (December 23, 2010; January 15, 2011).

Soil was sampled at 0–20 mm and 0–100 mm depths from each plot after cultivation but before P fertiliser treatments were applied (November 4, 2009; March 10, 2010) and approximately quarterly thereafter (May 14, 2010, July 5, 2010, October 4, 2010, November 29, 2010). Additional samples from 0–300 mm deep were sampled annually (December 3, 2009, November 1, 2010). Samples were recovered from 20 mm and 100 mm depths at each sampling date using stainless steel corers (25 mm ID). Samples to 300 mm were recovered using similar corers and sectioned (0–20 mm, 20–50 mm, 50–75 mm, 75–100 mm, 100–150 mm, 150–200 mm, 200–300 mm) to yield seven samples per plot. Soils were not sampled in the four-week period following fertiliser application.

For the 20 mm sampling, a minimum of fifty cores from each plot were bulked to provide a composite sample which was subsequently used for both soil and soil water analyses. Ten cores from each plot were bulked to provide the 100 mm samples, while for the 300 mm samples five cores from each depth interval were bulked. All soil samples were returned to the laboratory in insulated containers (5°C) where stones and large roots were removed and the samples thoroughly mixed. After mixing, a 50 g portion was taken from each sample to measure soil moisture and a 400 g portion of the 20 mm samples was taken for soil water analyses. The remaining soil was oven-dried at 40°C and sieved to 2 mm. The dried, sieved soil was stored in polyethylene containers at room temperature prior to further analysis.

Soil moisture was determined at 105°C. Soil samples were analysed for Olsen P, Colwell P, Calcium Chloride Extractable P ($CaCl_2$ P), Total P (TP), Total N (TN), P Buffer Index with Colwell Fertility Correction (PBI_{+ColP}), Skene Potassium (Skene K), Available Sulphur (CPC S, calcium phosphate plus charcoal extractable sulphur), Total Carbon (TC), and Oxidisable Organic Carbon (OOC, Walkley-Black method) [38]. Organic P was estimated as the additional P released after persulphate digestion of the Colwell P extract (LaChat Quickchem method 10-115-01-1-E, Hach, Loveland, CO, USA).

Soil water was extracted within 12 hours of sampling by centrifuging (GT20, Spintron Pty Ltd, Dandenong, Australia) 400 g of soil for 15 minutes at 1500 rpm (500 g) [39]. The soil water was then stored refrigerated (4°C) in polypropylene containers (Techno Plas Pty Ltd, St Marys, Australia) prior to chemical analysis. Dissolved reactive P (DRP) was measured within 24 hours, while all other analyses were completed within 7 days. The samples were analysed using a LaChat Quickchem flow injection analyser, and QuikChem methods (Hach, Loveland, CO, USA). A portion of each sample was filtered through a 0.45 μm membrane filter (Sartorius Minisart) for analysis of DRP (phosphoromolybdenum blue method 10-115-01-1-A), nitrate/nitrite (NO_x, cadmium reduction method 13-107-04-1-B), ammonia (NH_3, blue indophenol method 10-107-06-1-A), and

Table 1: Initial soil properties two weeks prior to ploughing (November 2 and 5, 2009).

	Total N (mg/kg)	Olsen P (mg/kg)	Colwell P (mg/kg)	Organic P (mg/kg)	$CaCl_2$ P (mg/kg)	NH_4^+ (mg/kg N)	NO_3^- (mg/kg N)
0–20 mm depth (November 2, 2009)							
Mean	5200	54	182	272	1.3	30	5.4
Minimum	4400	34	100	160	0.2	11	0.5
Maximum	6300	66	250	380	2.8	79	19
0–100 mm depth (November 5, 2009)							
Mean	4400	41	132	208	0.7	20	7.3
Minimum	3800	30	96	140	0.2	8.3	0.37
Maximum	5600	54	180	290	1.5	60	27
Relative percentage difference of means comparing the 0–20 mm and 0–100 mm depths							
	18.8	32.4	38.3	30.8	78.2	48.4	−26.0

TDP (ammonium persulphate digestion followed by phosphomolybdenum blue method 10-115-01-1-E). Unfiltered samples were analysed for TP (ammonium persulphate digestion followed by phosphoromolydenum blue method 10-115-01-1-E) and TN (ammonium persulphate digestion followed by diazotised sulphanilamide with NED method 10-107-04-1-A). Dissolved Unreactive P (DUP) was calculated as the difference between TDP and DRP; Particulate P (PP) was calculated as the difference in TP and TDP. Particulate N (PN) was calculated as the difference between TN and TDN, Dissolved Inorganic N (DIN) was calculated as the sum of ammonia and NO_x, and Dissolved Organic Nitrogen (DON) was calculated as the difference between TDN and DIN.

Pasture samples were collected biannually (May 27, 2010, January 27, 2011) by mowing a diagonal strip with a Toro 20332 lawn mower (Toro Australia Pty. Ltd. Beverly, Australia) to 80 mm high with a catcher, tipping the contents onto a plastic tarpaulin, weighing the contents plus tarpaulin and then subsampling. The pasture was then freeze-dried (Dynavac FD600, Gardner Denver Industries Australia Pty Ltd, Dandenong South, Australia) except for a 1 kg portion which was dried at 60°C to determine dry matter. The freeze dried samples were stored at room temperature in polyethylene bags prior to analysis by near infrared reflectance spectrometry to determine protein, carbohydrate, fibre, fat, and energy (Dairy One, Ithaca, NY).

2.1. Statistical Methods. The effect of cultivation, vegetation, and P fertilising levels was analysed by ANOVA using Genstat (Release 13, 2010, VSN International Ltd, Hemel Hempstead, UK) statistical software. Treatment and blocking structures were as follows:

$$\begin{aligned} &\text{Treatment: (Initial/(Sample Date} * \text{P Fertiliser Rate))} \\ &\qquad * \text{Vegetation} * \text{Cultivation,} \\ &\text{Block: Sample Date} * \text{(Block/Plot),} \end{aligned} \tag{1}$$

where "$*$" is the crossing operator, and "/" is the nesting operator. The initial factor identifies samples taken before and after P fertiliser treatment. All samples were subsequent to cultivation treatment application and establishment of pastures. All data were checked for outliers and normality of distribution and constant variance using graphs of residuals versus fitted values, histograms and normal quantile plots of residuals. If necessary, data were transformed to meet distributional assumptions using a generalised log transformation:

$$y = \ln\left(x + \sqrt{x + \lambda}\right), \tag{2}$$

where x is the analyte concentration [40], the parameter λ being estimated by maximum likelihood. Means were compared using least significant difference at the 5% level. Back-transformed means were calculated ($x = [e^y - (\lambda/e^y)]/2$) for presentation on the analyte concentration scale.

3. Results and Discussion

Monthly rainfalls during the trial period are shown in Figure 2. A total of 1151 mm fell over the trial area in 2010. High rainfall in winter and spring caused waterlogging (i.e., saturation of the soil and standing water for >10 days on occasions) which are both characteristic of this area.

Soil test results prior to ploughing are presented in Table 1. Based on these results, the soil would be classified as having moderate to high soil P fertility [41] and moderate P sorption [42] which is typical of soils and farms in this region (John Gallienne, *personal communication,* June 2011). A comparison of the 0–20 mm and 0–100 mm soil test results suggests that prior to ploughing the soil had higher concentrations of P nearer the surface. For example, in the 0–20 mm and 0–100 mm depths, concentrations were 182 and 132 mg Colwell P/kg, 54 and 41 mg Olsen P/kg, 1.3 and 0.7 mg $CaCl_2$ P/kg. Generally, N concentrations were also higher near the surface. For instance, TN and NH_4^+ were 5.2 and 4.4 g N/kg, and 30 and 20 mg N/kg, respectively. However, NO_3^- concentration was lower at the 0–20 mm depth (5.4 mg N/kg) compared to the 0–100 mm depth (7.3 mg N/kg). This is possibly due to the effects of denitrification [43] and leaching [44] moving N to lower depths following high spring rainfall (November 2, 2011 for 0–20 mm, November 5, 2010 for 0–100 mm). The stratification at this site is similar to that measured in the Adelaide Hills, South Australia [17], but less than that measured on irrigated

pastures of the Macalister irrigation district in south eastern Victoria, Australia [45]. This probably reflects prior fertiliser applications and operational management. The site for this study was a managed dairy pasture and, in the five years prior to this study being implemented, approximately 270 kg/ha single superphosphate, 130 kg/ha muriate of potash, and 100 kg/ha of urea were applied annually.

After ploughing and sowing, but prior to the P treatments being implemented, surface soil water was sampled (0–20 mm). Ploughing lowered concentrations of P ($P < 0.001$) but increased concentrations of N ($P < 0.001$). For example, DRP and TDP concentrations for ploughed and unploughed plots were 0.1 and 1.3 mg/L DRP and 0.6 and 3.3 mg/L TDP, respectively. The equivalent N data for ploughed and unploughed plots for concentrations were 60.0 and 30.5 mg/L N for TDN, 10.9 and 2.8 mg/L N for NH_3, and 33.2 and 8.2 mg/L N for NO_x. Decreased P concentrations are consistent with other studies [18, 45–47] and probably reflect the relocation of topsoil away from the soil surface, in addition to increased P adsorption where ploughing brought fresh clay material to the surface. Increased N concentrations following ploughing are consistent with organic matter disturbance and aeration stimulating the microbial population resulting in increased ammonification and nitrification [48, 49].

Ploughing also affected soil water P tests after the P fertiliser treatments were applied ($P < 0.001$) for the subsequent four samplings (May 17, 2010, July 7, 2010, October 4, 2010, and November 29, 2010) (Table 2). For example, the mean DRP and TDP concentrations in ploughed and unploughed plots were 0.25 and 0.8 mg/L P, and 0.51 and 1.52 mg/L P, respectively. Unlike the results prior to P fertiliser treatments being implemented, N concentrations were lower in the ploughed versus the unploughed plots (TDN, $P < 0.001$; NH_3, $P = 0.014$; and NO_x, $P = 0.023$). Nitrogen concentrations for ploughed and unploughed plots were 10.0 and 13.1 mg/L N for TDN, 0.4 and 0.5 mg/L N for NH_3, and 1.2 and 1.6 mg/L N for NO_x. These results are consistent with ploughing having stimulated the rapid decomposition of organic matter, exhausting this source of N near the soil surface prior to the fertiliser treatments being initiated.

The results of the 0–20 mm soil testes were consistent with the soil water tests (Table 3). Olsen P, Colwell P, Organic P, and $CaCl_2$ P concentrations decreased for ploughed versus unploughed plots ($P < 0.001$). The respective means were 35.8 and 66.4 mg/kg P for Olsen P, 99.4 and 205.4 mg/kg P for Colwell P, 150.7 and 314.5 mg/kg P for Organic P, and 0.50 and 1.8 mg/L P for $CaCl_2$ P.

There were no vegetation effects for the soil water. With the exception of NH_4^+, the same was true of the soil test results in the 0–20 mm soil samples. The concentration of NH_4^+ ($P = 0.003$) was higher for the mixed sward plots (11.0 mg/kg N) compared to the monoculture plots (8.5 mg/kg N). This was a surprising result as urea had been added to the ryegrass monoculture in response to the rust infestation. Presumably NH_4^+ released from urea had been nitrified and leached from the profile prior to sampling, whereas N fixation by the clover in the mixed sward increased NH_4^+ concentrations as has also been reported to occur in Canterbury, New Zealand [50] and Terang, south-west Victoria [51].

Higher P fertiliser application rates increased the concentration of P in the soil water ($P < 0.001$). For example, at the fertilising rates of 10, 35 and 100 kg/ha P, concentrations were 0.27, 0.36, and 0.91 mg/L for DRP, and 0.62, 0.73, and 1.52 mg/L for TDP. The results of the 0–20 mm soil tests results were consistent with the soil water tests ($P < 0.001$) as they increased in concentration with increased fertiliser rates. At fertiliser rates of 10, 35, and 100 kg/ha P, the soil test concentrations were 512, 539 and 623 mg TP/kg, 122, 137 and 198 mg Colwell P/kg, 41.4, 46.2 and 65.6 mg Olsen P/kg, 194, 217 and 287 kg Organic P/kg, and 0.66, 0.83 and 1.6 mg $CaCl_2$ P/kg, respectively. Similar trends have been found elsewhere [8, 9]. In the same study, Robertson and Nash also found that $CaCl_2$ P concentrations were not always higher for higher P fertiliser rates and concluded that $CaCl_2$ P data would not be a useful environmental test in the pasture systems that they studied [8].

Over the period of four samplings after all treatments were implemented, N initially decreased and then tended to increase. For example, TDN ($P = 0.001$) concentrations over the four sampling dates were 19.91, 8.59, 9.35, and 10.77 mg/L N, and NO_x ($P < 0.001$) concentrations were 4.39, 1.77, 0.41, and 1.17 mg/L N. Concentrations of NO_2^- ($P < 0.001$) remained unchanged. With the exception of DUP, which decreased over the first three samplings, and then doubled relative to the first sample (0.39, 0.29, 0.04, 0.77 mg/L DUP, $P = 0.001$), P concentrations decreased. The changes to DUP may in part be related to between sampling variation in the parameters from which it was derived. Another study [8], in which the highest fertiliser rate was only 23 kg P/ha, also found DUP did not have a consistent trend with P fertilising rate. Over the same period, 0–20 mm soil test results for P, TN, and NO_3^- decreased ($P < 0.001$ for Olsen P, Colwell P, and NO_3^-; $P = 0.004$ for TN; and $P = 0.002$ for Organic P). Sampling date did not affect NH_4^+ which has also been reported occurring in simulated cultivation trials near Brisbane, Queensland [49].

Aside from the main treatment effects, after fertiliser treatments, there were also treatment interactions. For all soil water P analyses, there were Sample Date by P Fertiliser Rate interactions where decay rates decreased as the P fertiliser application rates were decreased ($P = 0.02$ for TN, $P < 0.001$ for others, see Figure 3). As has been seen previously with laser grading [32], a Sample Date by Cultivation interaction was also observed ($P < 0.001$) where the effects of cultivation diminished over time. For example, at the first sampling (May 17, 2011) after all treatments had been applied, the DRP concentration in the ploughed and unploughed plots was 0.40 and 2.36 mg/L while six months later (November 29, 2010) the concentrations had decreased to 0.10 and 0.31 mg/L, respectively (Figure 3). TDP concentrations exhibited a similar trend except only the first three samplings showed a decrease in concentration. The fourth sample showed an increase in TDP concentrations, possibly due to waterlogging and pugging that occurred after the third sampling. A Vegetation by Cultivation interaction for both DIN

Table 2: Mean values of phosphorus and nitrogen concentration in soil water after fertiliser treatments were applied. Values in brackets represent significance at 5% level. Values before the comma compare ploughed versus unploughed within a fertiliser treatment, while values after the comma compare fertiliser treatments with identical sod preparation. Within a row, items with the same letter are not significantly different.

Treatments		Concentration (mg/L)											
		DRP	TDP	TP	TDN	TN	NH_3	NO_x	DUP	PP	PN	DIN	DON
10 kg/ha P	Unploughed	0.55 (a, a)	1.2 (a, a)	1.6 (a, a)	13.2 (a, a)	14.6 (a, a)	0.6 (a, a)	1.5 (a, a)	0.52 (a, a)	0.24 (a, a)	1.26 (a, a)	2.5 (a, a)	10.2 (a, a)
	Ploughed	0.13 (b, a)	0.3 (b, a)	0.4 (b, a)	11.3 (a, a)	12.0 (a, a)	0.4 (a, a)	1.2 (a, a)	0.16 (b, a)	0.1 (b, a)	0.5 (b, a)	1.8 (b, a)	9.2 (a, a)
35 kg/ha P	Unploughed	0.62 (a, a)	1.3 (a, a)	1.7 (a, a)	13.2 (a, a)	14.7 (a, a)	0.5 (a, a)	1.7 (a, a)	0.53 (a, a)	0.24 (a, a)	1.19 (a, a)	2.5 (a, a)	9.9 (a, a)
	Ploughed	0.21 (b, b)	0.4 (b, a)	0.5 (b, a)	9.8 (b, a)	10.4 (b, a)	0.4 (a, a)	1.3 (a, a)	0.15 (b, a)	0.09 (b, a)	0.51 (b, a)	1.9 (b, a)	7.7 (b, a)
100 kg/ha P	Unploughed	1.5 (a, b)	2.3 (a, b)	2.9 (a, b)	13.0 (a, a)	14.6 (a, a)	0.5 (a, a)	1.5 (a, a)	0.39 (a, a)	0.26 (a, a)	1.14 (a, a)	2.2 (a, a)	10.0 (a, a)
	Ploughed	0.55 (b, c)	1.0 (b, b)	1.2 (b, b)	9.0 (b, a)	9.7 (b, a)	0.4 (a, a)	1.2 (a, a)	0.25 (b, a)	0.11 (b, a)	0.56 (b, a)	1.8 (a, a)	7.4 (a, a)

Table 3: Mean values of soil fertility tests for both the 0–20 mm and 0–100 mm soil depths after fertiliser treatments were applied. Values in brackets represent significance at 5% level. Values before the comma compare ploughed versus unploughed within a fertiliser treatment, while values after the comma compare fertiliser treatments with identical sod preparation. Within a row, items with the same letter are not significantly different when comparing 0–20 mm together or comparing 0–100 mm together (significant tests have not been applied between different depths).

Depth	Treatments		Concentration (mg/kg)					
			Olsen P	Colwell P	Organic P	Total P	$CaCl_2$ P	TN
Top 2 cm	10 kg/ha P	Unploughed	58.6 (a, a)	177 (a, a)	281 (a, a)	719 (a, a)	1.5 (a, a)	4600 (a, a)
		Ploughed	24.3 (b, a)	67.4 (b, a)	106 (b, a)	297 (b, a)	0.3 (b, a)	2320 (b, a)
	35 kg/ha P	Unploughed	60.2 (a, a)	187 (a, a)	297 (a, a)	737 (a, a)	1.5 (a, a)	4930 (a, a)
		Ploughed	32.2 (b, a)	86.8 (b, a)	137 (b, a)	330 (b, a)	0.5 (b, b)	2280 (b, a)
	100 kg/ha P	Unploughed	80.3 (a, b)	252 (a, b)	365 (a, a)	799 (a, a)	2.9 (a, b)	4600 (a, a)
		Ploughed	50.9 (b, c)	144 (b, b)	209 (b, b)	441 (b, b)	0.9 (b, c)	2440 (b, a)
Top 10 cm	10 kg/ha P	Unploughed	42.9 (a, a)	126 (a, a)	217 (a, ac)	580 (a, a)	0.6 (a, a)	4000 (a, ac)
		Ploughed	28.9 (b, a)	83.3 (b, a)	145 (b, a)	360 (b, b)	0.5 (a, a)	2700 (b, a)
	35 kg/ha P	Unploughed	43.0 (a, a)	131 (a, a)	218 (a, bc)	593 (a, a)	0.6 (a, a)	4200 (a, a)
		Ploughed	28.4 (b, a)	78.2 (b, a)	140 (b, a)	365 (b, a)	0.4 (a, a)	2900 (b, a)
	100 kg/ha P	Unploughed	49.3 (a, b)	132 (a, a)	225 (a, a)	468 (a, a)	0.7 (a, a)	3400 (a, bc)
		Ploughed	40.4 (b, b)	115 (a, b)	177 (a, a)	463 (a, a)	0.5 (a, a)	3200 (a, b)

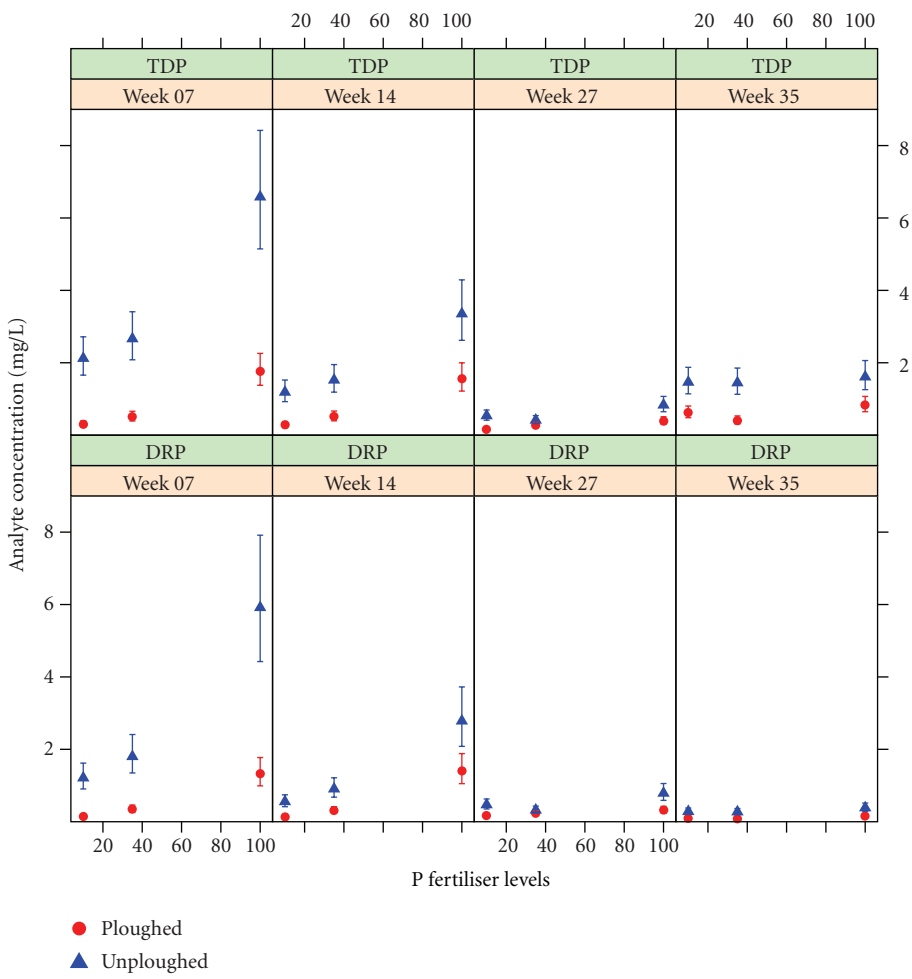


Figure 3: Mean TDP and DRP concentrations in soil water for ploughed and unploughed treatments at different P fertiliser levels showing the decay over the trial period. Week represents the number of weeks since fertiliser treatments began. Error bars represent significance at 5% level.

Table 4: Mean pasture quality tests for vegetation treatments, sampled May 17, 2010.

	Yield (kg·DM/ha)	Dry matter basis				
		Metabolisable energy (MJ/kg)	Crude protein (%)	Neutral detergent fibre (%)	Starch (%)	Crude fat
Monoculture	474	10.4	22.5	48	0.5	5.6
Mixed sward	535	11	25.8	43	1.9	5.3
Significance (P value)	0.008	<0.001	<0.001	<0.001	<0.001	0.019

($P = 0.012$) and NO_x ($P = 0.003$) was also observed. DIN in the mixed sward plots accounted for the main cultivation effect, while the monoculture plots were unchanged between cultivation types. NO_x concentrations were similar, except that in the monoculture plots, unploughed plots had reduced NO_x concentrations versus ploughed plots, even though the mixed sward plots again accounted for the main cultivation effect. A marginal ($P = 0.045$) P Fertiliser by Cultivation interaction for DUP was observed with the interaction most evident at the 100 kg/ha P fertiliser application rate.

Except for NH_4^+ or NO_3^-, the 0–20 mm soils test results had a similar Sample Date by Cultivation interaction ($P < 0.001$) to the soil water analyses. For P and TN, the difference between ploughed and unploughed plots decreased over time. For example, Olsen P concentration in ploughed and unploughed plots was 41.95 and 87.44 mg/kg P in May 2010, and 30.8 and 51.9 in November 2010.

The 0–100 mm soil tests had trends consistent with the 0–20 mm soil tests (Table 2). At the 0–100 mm depth, TP, Colwell P, Olsen P, $CaCl_2$ P and OOC concentrations decreased with ploughing ($P < 0.001$). For example, in the ploughed and unploughed plots the mean concentrations were 396 and 547 mg/kg TP, 92.2 and 128.9 mg/kg Colwell P, 32.6 and 45.1 mg/kg Olsen P, 0.4 and 0.6 mg/kg $CaCl_2$ P, and 7.1 and 9.4 mg/kg OOC, respectively. TN and NH_4^+ concentrations also decreased with ploughing. For example, in the ploughed and unploughed plots the mean concentrations were 2.9 and 3.9 g/kg TN, and 8.5 and 11.5 mg/kg NH_4^+ as N. Only Olsen P increased with P fertiliser application rate, and then only at the highest P fertiliser rate; the concentration of Olsen P was 35.6, 35.7, and 44.8 mg/kg at 10, 35, and 100 kg/ha P, respectively.

The pasture results (May 17, 2010) are summarised in Table 4. Neutral Detergent Fibre (NDF) and Crude Fat (CF) were higher in the monoculture plots compared to the mixed sward plots (NDF, $P < 0.001$; CF, $P = 0.019$). For example, in the monoculture plots, NDF and CF concentrations were 47.9 and 5.6% (dry weight basis), and, in the mixed sward plots, the concentrations were 42.8 and 5.4%. Other test results were higher in the mixed sward plots compared to the monoculture plots. For example, yield, metabolisable energy (ME), crude protein (CP) and starch concentrations on a dry matter basis in the monoculture plots were 474 kg·DM/ha, 10.4 MJ/kg, and 22.5 and 0.5%, while in the mixed sward plots the results were 535 kg·DM/ha, 11.0 MJ/kg, and 25.8 and 1.9%. Ploughing lowered the concentration of CP ($P = 0.032$). The mean CP concentration in the ploughed and unploughed plots was 23.7% and 24.5%, respectively. This effect may be the result of reduced nitrogen in the top layer of soil given that nitrogen has a positive effect on CP [52]. However, such differences are biologically insignificant (B. Wales, *personal communication* 16-8-11) and cultivation effects or interactions may also be due to differences in establishment of the plots, such as weeds like paspalum that were more prevalent in the unploughed plots. There was a P Fertiliser by Cultivation effect ($P = 0.025$) where the concentration of CF increased with increases in P fertiliser rates for the ploughed plots but not for the unploughed plots. ME had a Vegetation by Cultivation interaction ($P = 0.036$) for mixed sward plots with the average ME concentration higher in the unploughed plots compared to the ploughed plots, but there was no significant difference between cultivation types for the monoculture plots. CF also had a Vegetation by Cultivation interaction ($P < 0.001$) where the concentration of CF in the monoculture plots was higher than the mixed sward plots for the cultivated treatments, but lower than the mixed sward plots in the uncultivated treatment.

Cultivation could be incorporated into farm management at times when pasture needs renovating or when a summer forage crop is to be planted. Generally, when pasture renovation is required, seed is over-sown into the pasture. Ploughing is not carried out, due to the added cost of seed bed preparation and the risk of erosion. However if soil P has built up in the upper layers, it could be environmentally advantageous to include ploughing to bury or mix the upper layer of soil with soil from lower in the profile. In doing so, the potential for soil erosion and consequent export of particulate P could potentially outweigh the advantage of ploughing to decrease dissolved P exports. For instance, ploughing would be too high a risk for steep hill slopes (>5%), and renovation areas should only be performed when the potential for heavy precipitation is minimised (early autumn in Gippsland) (J. Gallienne, *personal communication* 17-11-11). In this study, a mouldboard plough was used rather than a power harrow which is more commonly used for commercial cultivation in the Gippsland. To destratify the soil P, the upper soil layer must be either buried lower in the soil profile (e.g., mouldboard plough) or mixed in with the lower layers (e.g., rotary hoe). In contrast, some other types of cultivators (e.g., power harrow) would be less effective as they break up the soil but do not mix material in a vertical plane.

4. Concluding Discussion

As phosphorus exports from intensively grazed land are well above the 0.045 mg/L TP objective values for Victorian coastal plains, mitigation measures need to be identified and

implemented [12]. In this study, we compared potential P and N exports from a pasture soil from West Gippsland with higher P, and generally higher N concentrations in the upper layer of soil. It is from this layer that the majority of P and N is mobilised into overland flow [53]. Cultivation mixed the upper nutrient rich layer further down the soil profile and brought up the less nutrient rich material to the surface. The concentrations of P and N in the upper 20 mm soil layer were lower in the ploughed treatments compared with the unploughed plots ($P < 0.001$). Cultivation also reduced the P and N concentrations in the soil water ($P < 0.001$ for DRP, TDP, and TDN). Overall, mean soil water DRP concentration initially decreased by 70% by cultivation from 0.79 mg/L in the unploughed plots to 0.23 mg/L in the ploughed plots. At six months after treatments had been applied, the effect of ploughing on DRP, expressed as a percentage, was almost identical, 66% (DRP means, unploughed plots: 0.31 mg/L; ploughed plots: 0.10 mg/L). Therefore, the cultivation undertaken potentially reduced the concentration of P that could be mobilised during overland flow events. Pasture type had no significant effect on the results after one year, although the mixed pasture had a higher yield and a higher quality feed than the monoculture plots.

Destratification by ploughing could be incorporated into farm management practice on occasions when pasture requires renovation, or a summer forage crop is to be planted. The cultivating machinery would need to mix or invert the soil in a vertical plane.

Acknowledgments

The authors would like to acknowledge the study was conducted on the property of Brett and Jodie Loughridge who provided enthusiastic help. The authors also wish to acknowledge the advice and monthly consultation with John Gallienne, agricultural consultant. The authors affirm they have no financial interest in the named products used, that is, Incitec Pivot fertilisers and Sartorius Minisart filters.

References

[1] J. R. Davis and K. Koop, "Eutrophication in Australian rivers, reservoirs and estuaries—a Southern hemisphere perspective on the science and its implications," *Hydrobiologia*, vol. 559, no. 1, pp. 23–76, 2006.

[2] M. R. Grace, T. R. Scicluna, C. L. Vithana, P. Symes, and K. P. Lansdown, "Biogeochemistry and cyanobacterial blooms: investigating the relationship in a shallow, polymictic, temperate lake," *Environmental Chemistry*, vol. 7, no. 5, pp. 443–456, 2010.

[3] V. May, "The occurance of toxic cyanophyte blooms in Australia," in *The Water Environment: Algal Toxins and Health*, W. W. Carmichael, Ed., pp. 127–142, Plenum, New York, NY, USA, 1981.

[4] I. R. Falconer, A. M. Beresford, and M. T. C. Runnegar, "Evidence of liver damage by toxin from a bloom of the blue-green alga, *Microcystis aeruginosa*," *Medical Journal of Australia*, vol. 1, no. 11, pp. 511–514, 1983.

[5] A. T. C. Bourke, R. B. Hawes, A. Neilson, and N. D. Stallman, "An outbreak of hepato-enteritis (the Palm Island mystery disease) possibly caused by algal intoxication," *Toxicon*, vol. 21, no. 3, pp. 45–48, 1983.

[6] P. Cottingham, G. Bennison, R. Dunn et al., *Algal Bloom and Nutrient Status of Victorian Surface Waters*, State Water Laboratory of Victoria, Melbourne, Australia, 1995.

[7] R. W. McDowell and R. J. Wilcock, "Water quality and the effects of different pastoral animals," *New Zealand Veterinary Journal*, vol. 56, no. 6, pp. 289–296, 2008.

[8] F. A. Robertson and D. M. Nash, "Phosphorus and nitrogen in soil, plants, and overland flow from sheep-grazed pastures fertilized with different rates of superphosphate," *Agriculture, Ecosystems and Environment*, vol. 126, no. 3-4, pp. 195–208, 2008.

[9] K. Barlow, D. M. Nash, and R. B. Grayson, "Phosphorus export at the paddock, farm-section, and whole farm scale on an irrigated dairy farm in South-Eastern Australia," *Australian Journal of Agricultural Research*, vol. 56, no. 1, pp. 1–9, 2005.

[10] I. T. Webster and Gippsland Coastal Board, *Gippsland Lakes Environmental Study—Assessing Options for Improving Water Quality and Ecological Function*, CSIRO Publishing, Collingwood, Australia, 2001.

[11] R. B. Grayson, K. S. Tan, and A. Western, *Estimation of Sediment and Nutrient Loads into the Gippsland Lakes*, CSIRO Publishing and University of Melbourne, Collingwood, Australia, 2001.

[12] Environment Protection Authority, *Nutrient Objectives for Rivers and Streams—Ecosystem Protection*, Environment Protection Authority, Melbourne, Australia, 2003.

[13] Department of Natural Resources and Environment, *Macalister Irrigation District Nutrient Reduction Plan*, Department of Natural Resources and Environment, Victoria, Australia, 1998.

[14] Environment Protection Authority, *Protecting Water Quality in Central Gippsland. Schedule F5—Waters of the Latrobe and Thomson River Basins and Merriman Creek Catchment and Draft Policy Impact Assessment*, Environment Protection Authority, Melbourne, Australia, 1995.

[15] T. Ladson and J. Tilleard, *BMPs for Reducing Phosphorus Loads to the Gippsland Lakes, Report on Findings from Expert Panel*, Gippsland Lakes Task Force, 2006.

[16] E. Frossard, L. M. Condron, A. Oberson, S. Sinaj, and J. C. Fardeau, "Processes governing phosphorus availability in temperate soils," *Journal of Environmental Quality*, vol. 29, no. 1, pp. 15–23, 2000.

[17] W. J. Dougherty, D. M. Nash, D. J. Chittleborough, J. W. Cox, and N. K. Fleming, "Stratification, forms, and mobility of phosphorus in the topsoil of a Chromosol used for dairying," *Australian Journal of Soil Research*, vol. 44, no. 3, pp. 277–284, 2006.

[18] N. J. Mathers and D. M. Nash, "Effects of tillage practices on soil and water phosphorus and nitrogen fractions in a Chromosol at Rutherglen in Victoria, Australia," *Australian Journal of Soil Research*, vol. 47, no. 1, pp. 46–59, 2009.

[19] A. N. Sharpley, S. J. Smith, and W. R. Bain, "Nitrogen and phosphorus fate from long-term poultry litter applications to Oklahoma soils," *Soil Science Society of America Journal*, vol. 57, no. 4, pp. 1131–1137, 1993.

[20] J. T. Sims, R. R. Simard, and B. C. Joern, "Phosphorus loss in agricultural drainage: Historical perspective and current research," *Journal of Environmental Quality*, vol. 27, no. 2, pp. 277–293, 1998.

[21] S. T. Pierson, M. L. Cabrera, G. K. Evanylo et al., "Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter," *Journal of Environmental Quality*, vol. 30, no. 5, pp. 1784–1789, 2001.

[22] D. H. Pote, T. C. Daniel, D. J. Nichols et al., "Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff," *Journal of Environmental Quality*, vol. 28, no. 1, pp. 170–175, 1999.

[23] D. M. Nash and C. Murdoch, "Phosphorus in runoff from a fertile dairy pasture," *Australian Journal of Soil Research*, vol. 35, no. 2, pp. 419–429, 1997.

[24] D. M. Nash, L. Clemow, M. Hannah, K. Barlow, and P. Gangaiya, "Modelling phosphorus exports from rain-fed and irrigated pastures in Southern Australia," *Australian Journal of Soil Research*, vol. 43, no. 6, pp. 745–755, 2005.

[25] D. M. Nash, M. Hannah, D. Halliwell, and C. Murdoch, "Factors affecting phosphorus export from a pasture-based grazing system," *Journal of Environmental Quality*, vol. 29, no. 4, pp. 1160–1166, 2000.

[26] D. M. Nash and M. Hannah, "Using Monte-Carlo simulations and Bayesian Networks to quantify and demonstrate the impact of fertiliser best management practices," *Environmental Modelling and Software*, vol. 26, no. 9, pp. 1079–1088, 2011.

[27] J. E. Morrison and F. W. Chichester, "Tillage system effects on soil and plant nutrient distributions on vertisols," *Journal of Production Agriculture*, vol. 7, no. 3, pp. 364–373, 1994.

[28] B. Pezzarossa, M. Barbafieri, A. Benetti et al., "Effects of conventional and alternative management systems on soil phosphorus content, soil structure, and corn yield," *Communications in Soil Science and Plant Analysis*, vol. 26, no. 17-18, pp. 2869–2885, 1995.

[29] G. W. Rehm, G. W. Randall, A. J. Scobbie, and J. A. Vetsch, "Impact of fertilizer placement and tillage system on phosphorus distribution in soil," *Soil Science Society of America Journal*, vol. 59, no. 6, pp. 1661–1665, 1995.

[30] A. N. Sharpley, "Soil mixing to decrease surface stratification of phosphorus in manured soils," *Journal of Environmental Quality*, vol. 32, no. 4, pp. 1375–1384, 2003.

[31] D. M. Nash, B. Webb, M. Hannah et al., "Changes in nitrogen and phosphorus concentrations in soil, soil water and surface run-off following grading of irrigation bays used for intensive grazing," *Soil Use and Management*, vol. 23, no. 4, pp. 374–383, 2007.

[32] D. M. Nash and H. Castlehouse, "Improved grazing systems that enhance water quality (phase 2)," Final Report, 2009.

[33] Pastures Australia, *White Clover*, 2011, http://www.pasturepicker.com.au/Html/White_clover.htm.

[34] Pastures Australia, *Perennial Ryegrass*, 2011, http://www.pasturepicker.com.au/Html/Perennial_ryegrass.htm.

[35] R. F. Isbell, *Australian Soil Classification*, CSIRO Publishing, Collingwood, Australia, 2002.

[36] Commonwealth of Australia, B.o.M. *Monthly rainfall, Poowong (Post Office)*, 2011, http://reg.bom.gov.au/jsp/ncc/cdio/weatherData/av?p_nccObsCode=139&p_display_type=dataFile&p_stn_num=086092.

[37] F. Tyndall, *Grazing Dairy Pastures*, Department of Natrual Resources and Environment, Victorian State Government, Victoria, Australia, 2002.

[38] G. E. Rayment and D. J. Lyons, *Soil Chemical Methods—Australasia*, CSIRO Publishing, Collingwood, Australia, 2011.

[39] M. Toifl, D. Nash, F. Roddick, and N. Porter, "Effect of centrifuge conditions on water and total dissolved phosphorus extraction from soil," *Australian Journal of Soil Research*, vol. 41, no. 8, pp. 1533–1542, 2003.

[40] D. M. Rocke, B. Durbin, M. Wilson, and H. D. Kahn, "Modeling uncertainty in the measurement of low-level analytes in environmental analysis," *Ecotoxicology and Environmental Safety*, vol. 56, no. 1, pp. 78–92, 2003.

[41] K. I. Peverill, L. A. Sparrow, and D. J. Reuter, Eds., *Soils Analysis: An Interpretation Manual*, CSIRO Publishing, Collingwood, Australia, 1999.

[42] L. L. Burkitt, P. W. Moody, C. J. P. Gourley, and M. C. Hannah, "A simple phosphorus buffering index for Australian soils," *Australian Journal of Soil Research*, vol. 40, no. 3, pp. 497–513, 2002.

[43] E. W. Russell, *Soil Conditions and Plant Growth*, Longman, 10th edition, 1973.

[44] D. M. Nash, D. Halliwell, and J. Cox, "Hydrological mobilisation of pollutants at the slope/field scale," in *Agriculture, Hydrology and Water Quality*, P. M. Haygarth and S. C. Jarvis, Eds., pp. 225–242, CABI Publishing, Oxon, UK, 2002.

[45] D. M. Nash, B. Webb, M. Hannah et al., "Changes in nitrogen and phosphorus concentrations in soil, soil water and surface run-off following grading of irrigation bays used for intensive grazing," *Soil Use and Management*, vol. 23, no. 4, pp. 374–383, 2007.

[46] A. N. Sharpley, "Soil mixing to decrease surface stratification of phosphorus in manured soils," *Journal of Environmental Quality*, vol. 32, no. 4, pp. 1375–1384, 2003.

[47] D. T. Vu, C. Tang, and R. D. Armstrong, "Tillage system affects phosphorus form and depth distribution in three contrasting Victorian soils," *Australian Journal of Soil Research*, vol. 47, no. 1, pp. 33–45, 2009.

[48] A. L. Heathwaite, P. J. Johnes, and N. E. Peters, "Trends in nutrients," *Hydrological Processes*, vol. 10, no. 2, pp. 263–293, 1996.

[49] P. R. Grace, I. C. MacRae, and R. J. K. Myers, "Temporal changes in microbial biomass and N mineralization under simulated field cultivation," *Soil Biology and Biochemistry*, vol. 25, no. 12, pp. 1745–1753, 1993.

[50] W. R. Cookson, I. S. Cornforth, and J. S. Rowarth, "Winter soil temperature (2–15°C) effects on nitrogen transformations in clover green manure amended or unamended soils; a laboratory and field study," *Soil Biology and Biochemistry*, vol. 34, no. 10, pp. 1401–1415, 2002.

[51] F. R. McKenzie, J. L. Jacobs, P. Riffkin, G. Kearney, and M. McCaskill, "Long-term effects of multiple applications of nitrogen fertiliser on grazed dryland perennial ryegrass/white clover dairy pastures in south-west Victoria. 1. Nitrogen fixation by white clover," *Australian Journal of Agricultural Research*, vol. 54, no. 5, pp. 461–469, 2003.

[52] F. R. McKenzie, J. L. Jacobs, M. J. Ryan, and G. Kearney, "Effect of rate and time of nitrogen application from autumn to midwinter on perennial ryegrass-white clover dairy pastures in Western Victoria. 2. Pasture nutritive value," *Australian Journal of Agricultural Research*, vol. 50, no. 6, pp. 1067–1072, 1999.

[53] L. R. Ahuja and O. R. Lehman, "The extent and nature of rainfall-soil interaction in the release of soluble chemicals to runoff," *Journal of Environmental Quality*, vol. 12, no. 1, pp. 34–40, 1983.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 608275, 9 pages
doi:10.1155/2012/608275

Research Article

Effect of Management Practices on Soil Microstructure and Surface Microrelief

R. Garcia Moreno,[1] T. Burykin,[2] M. C. Diaz Alvarez,[1] and J. W. Crawford[2]

[1] *Centre for Studies and Research on Agricultural and Environmental Risk Management (CEIGRAM), School of Agricultural Engineering, Polytechnic University of Madrid, Ciudad Universitaria s.n., 28040 madrid, Spain*
[2] *Faculty of Agriculture, Food and Natural Resources, Australian Technology Park, University of Sydney, Eveleigh, Sydney, NSW 2015, Australia*

Correspondence should be addressed to R. Garcia Moreno, rosario.garciam@udc.es

Received 1 December 2011; Revised 8 March 2012; Accepted 9 March 2012

Academic Editor: Philip White

Soil surface roughness (SSR) and porosity were evaluated from soils located in two farms belonging to the Plant Breeding Institute of the University of Sidney. The sites differ in their soil management practices; the first site (PBI) was strip-tilled during early fall (May 2010), and the second site (JBP) was under power harrowed tillage at the end of July 2010. Both sites were sampled in mid-August. At each location, SSR was measured for three 1 m^2 subplots using shadow analysis. To evaluate porosity and aggregation, soil samples were scanned using X-ray computed tomography with 5 μm resolution. The results show a strong negative correlation between SSR and porosity, 20.13% SSR and 41.38% porosity at PBI versus 42.00% SSR and 18.35% porosity at JBP. However, soil images show that when soil surface roughness is higher due to conservation and soil management practices, the processes of macroaggregation and structural porosity are enhanced. Further research must be conducted on SSR and porosity in different types of soils, as they provide complementary information on the evaluation of soil erosion susceptibility.

1. Introduction

Soil surface roughness (SSR), which describes the microvariations in soil elevations primarily resulting from tillage practices and textural porosity, is one of the major factors affecting wind and water erosion [1–4]. SSR is a direct indicator of the degradation of soil microstructure, which is mainly due to a loss of physical, chemical, and biological properties [1, 2, 5]. In this case, SSR is closely related to erosion, which is the primary cause for the loss of soil structure and organic matter, and it leads to a decrease in soil productivity and reduced fauna diversity [4, 6].

SSR promotes soil biota activity, which plays an important role in the rehabilitation of sealed soil surfaces and the restructuring of soils, particularly after compaction events [7]. SSR is mainly affected by management practices and, depending on the techniques used, SSR can increase the number and variability of microorganisms through the improvement of soil porosity and flow water in the vadose zone [8]. The increase in microorganism activity is very important in most biogeochemical cycles within soils because it improves the physical and biological state of the soil [4, 9, 10]. Thus, tillage influences the development of different types of microorganisms. Techniques that conserve pore systems tend to enhance the activity of microorganisms and conserve the biota that are beneficial to the development of crops [7, 11, 12].

However, the study of soil porosity and its relation to other properties is complicated because soil is one of the most complex materials on all scales [12, 13].

By defining soil structure as the arrangement of particles and associated pores in soils, ranging from nanometres to centimetres, we can demonstrate the biological influence on the stabilisation of aggregates at macro-to-micro scales and their relationship with soil surface processes [14]. This assumption was proposed by Oades [14], who found that

degradation of cultivated land starts with the breakdown of aggregates from larger structures and the corresponding pores, which control drainage and aeration (>30 μm).

Soil porosity includes textural porosity and structural porosity. Textural porosity is also known as matrix, intraaggregate, or intrapedal porosity. It is produced by the voids between primary mineral particles. Structural porosity, also called interaggregate or interpedal porosity, is produced by the pores between aggregates or soil blocks. The morphology and interconnection of structural porosity is closely related to the shape, size, and stability of aggregates and blocks, and it is generally also related to the soil genesis and type of soil use. Knowledge of the pore size distribution allows the evaluation of connectivity and soil flow properties [9, 10, 15].

Dexter [9] found that management factors, such as tillage, cropping, and compaction, have great influence on structural porosity and showed that soils with poorer physical quality produce more clods. There is a strong relation between small aggregates (<8 mm) and large aggregates (>32 mm) because from an agronomic point of view, tillage must create fine aggregates to obtain optimal plant emergence. From this point of view, large aggregates or clods have no agronomic significance and may create problems for soil management, causing soil aggregation to be very important in preventing soil erosion [1].

The aggregate hierarchy proposed by Hadas [16] and reviewed by Dexter [17] represents one of the most frequently applied theories due to its simplicity. The lowest hierarchical order is the combination of single mineral particles, such as clay plates, into a basic type of compound particle (e.g., a domain of clay plates). The next hierarchical order is formed by larger compound particles, such as clusters of domains. Once these clusters come together to form microaggregates, they enter the next hierarchical order, and so on. We find greater compaction and higher homogeneous properties for orders of decreasing size. The existence of different orders of aggregates depends on soil properties [16].

The stability of micro- and macro-aggregates also presents different properties. Although macroaggregate stability is positively correlated to different types of structures of organic matter, microaggregate stability fails to correlate to any type of organic matter [15, 18]. In some cases, humic substances may play an important role in the cohesion between clay particles through links with polyvalent cations [15, 19]. However, studies presenting soils with different pedogenesis have shown that microaggregate stability is mainly associated with soil mineralogy [18].

High-quality soil systems assure that physical, chemical, and biological properties are appropriated for soil conservation, thus avoiding soil degradation in a production system. Thus, any management practice that increases soil aggregation improves the structure of soils and permits the development of textural soil, which positively influences the function of soil ecosystems [20, 21]. This is reinforced by Hati et al. [22], showing that one of the main elements to assure soil fertility is the maintenance of optimum soil physical conditions through the applications of conservation practices. Hati et al. [22] corroborated that the decline of organic matter content in soil is associated with the physical degradation of soil.

Six et al. [23] corroborated the former theory. They found that tillage operations strongly influence the organic matter associated with aggregates because tillage operations increase the aggregate formation and the associated organic matter leads to the stabilisation of soil structure by increasing the structural porosity. In fact, a lack of tillage leads to an increase in particulate organic matter related to micro- and macro-aggregates compared to conventional tillage operations.

Kravchenko et al. [19] studied the organic matter of soils handled by different tillage systems over 15 years and found that soils managed with no tillage conserved the highest organic matters. When plots were conventionally tilled, the plots covered with residual vegetal matter conserved more organic matter than bare soils. The authors demonstrated that conservation techniques are able to increase the soil organic matter. Soil texture also plays an important role in the conservation of organic matter. In general, soils with higher clay contents have a positive correlation with the organic matter content, although the mineralisation of the organic matter is increased in more coarsely textured soils [24].

Therefore, research has shown that the net productivity of cultivated soils comes from the addition of numerous physical, chemical, and biological processes, which occur on the micrometre-to-metre scale, where soil porosity plays a very important role in facilitating the processes [25, 26]. Observations of soil in thin sections are an essential tool for evaluating biota activity in relation to the description and quantification of soil porosity, its potential impact on soil formation and conservation, its influence on soil connectivity [3, 27] and associated degradation processes [19, 24].

Microstructures are used to determine most of the processes in soils [12, 21, 28]. Computer tomography analysis and three-dimensional images of soil microstructures have been crucial to understanding the role of organic matter, its relation to soil aggregate stability and its influence on soil porosity. Macro- and micro-pores seem to play a very important role in organic matter degradation and conservation through aggregate stabilisation and porosity distribution [11, 12, 21, 25, 28].

Progress in computer tomography (CT) technology offers the possibility of imaging the nondestructive three-dimensional structure of soils at resolutions relevant to the interactions between physical, chemical, and biological soil properties [3, 26, 29]. Unfortunately, current scanners and image processing software are not able to resolve microbial cells; however, they can reproduce organic matter and its distribution through the pore system, which can be an indirect indicator of microbial activity [3].

Several authors [26, 29] have found that the morphology of the soil pore distribution is an important factor in understanding soil processes and formation at the small scale. X-ray computed tomography (CT) has been used to study aggregate stability in relation to pore architecture as a means of investigating microbial activity. These authors

concluded that stability, rather than morphology, of pores plays an important role in the formation of aggregates at the microstructure scale.

Soil surface roughness has also been related to texture, porosity and organic matter [30]; however, its relation with the formation of aggregates has been poorly studied.

Based on previous studies [1, 2, 12, 13, 28], the aim of the present study is to evaluate the influence of soil management practices on soil porosity and soil surface roughness as well as the relationship between both indicators. Soil cores were studied using X-ray computed tomography (CT) to visualise soil pore spaces in three dimensions, and soil surface roughness was measured using shadow analysis. The evaluation is suited to integrated SSR characterisation related to the estimation of soil pore spaces and the visualisation of the architecture of aggregates as a function of soil management practices.

2. Materials and Methods

2.1. Experimental Sites. The field experiments were conducted at two sites belonging to the Breeding Plant Institute of the University of Sidney under different management practices (Figure 1): one at the main facilities of the experimental farm of the Plant Breeding Institute (PBI) and the second at John B. Pye (JBP) Farm (Figure 2). Both sites are located 65 km southwest of Sidney, Australia, 34°00′51″S 150°40′49″E and are characterised as red dermosol with a clay loamy texture according to the Australian Soil Classification [31].

The main difference between the two sites was the management practices (Table 1). The sites were chosen based on their differences in crop management in order to relate conservation practices to soil surface roughness and microstructure. The first site (PBI) was strip-tilled beginning on May 2010 during early fall. The site was characterised as having bare soils, being highly eroded, and having no cover protection. The JBP plots were characterised by conservation management practices. The JBP soils were power harrowed to 5 cm at the end of July and covered with cereal crop remains incorporated into the soil and prepared for sowing cereals. The images and samples were taken on August 13. Surface images of both soils at the time of sampling are presented in Figures 2 and 3.

Soil surface roughness and microstructure were measured for samples from three randomly chosen 1 m^2 subplots within each site using shadow analysis and a tomography scan. The samples for porosity measurement were taken at the soil surface level. Sidney Observatory Hill recorded 249.6 mm for Fall 2010, below the historical average of 397.1 mm. The regional value recorded for the New South Wales region was 141.8 mm, slightly below the historical average of 142.7 mm [32].

2.2. Soil Surface Roughness Data. Soil surface roughness was measured at three 1 m^2 subplots from each site. For each subplot, three images were taken from the south, north, and west. Thus, nine images from each site were compared and measured for soil surface roughness. The images taken from the south at each location are shown in Figures 2 and 3.

Figure 1: Sampling locations: the blue dot indicates PBI, and the red dot indicates JBP.

(a)

(b)

Figure 2: Experimental sites at PBI (a) and JBP (b).

Shadow analysis is based on the assumption that the lengths of shadows cast at a given angle in bright daylight are proportional to soil microrelief [1, 2]. The images were captured with a Panasonic DMC-FT2 digital camera at a resolution of 14.50 MP.

All photos were taken at a solar angle of 40° to preclude any possibility of sunlight-induced differences during the same day. The test fields were close enough to ensure that the angle of incident light was the same. This angle was verified before images were taken.

Table 1: Differences in management practices for the experimental sites.

Location	Soil texture	Management	Coverage	Date of sampling
PBI	Clay loam	Strip tillage (last week of July)	Bare soil	August 13, 2010
JBP	Clay loam	Power harrow (first week of May)	Straw cover	

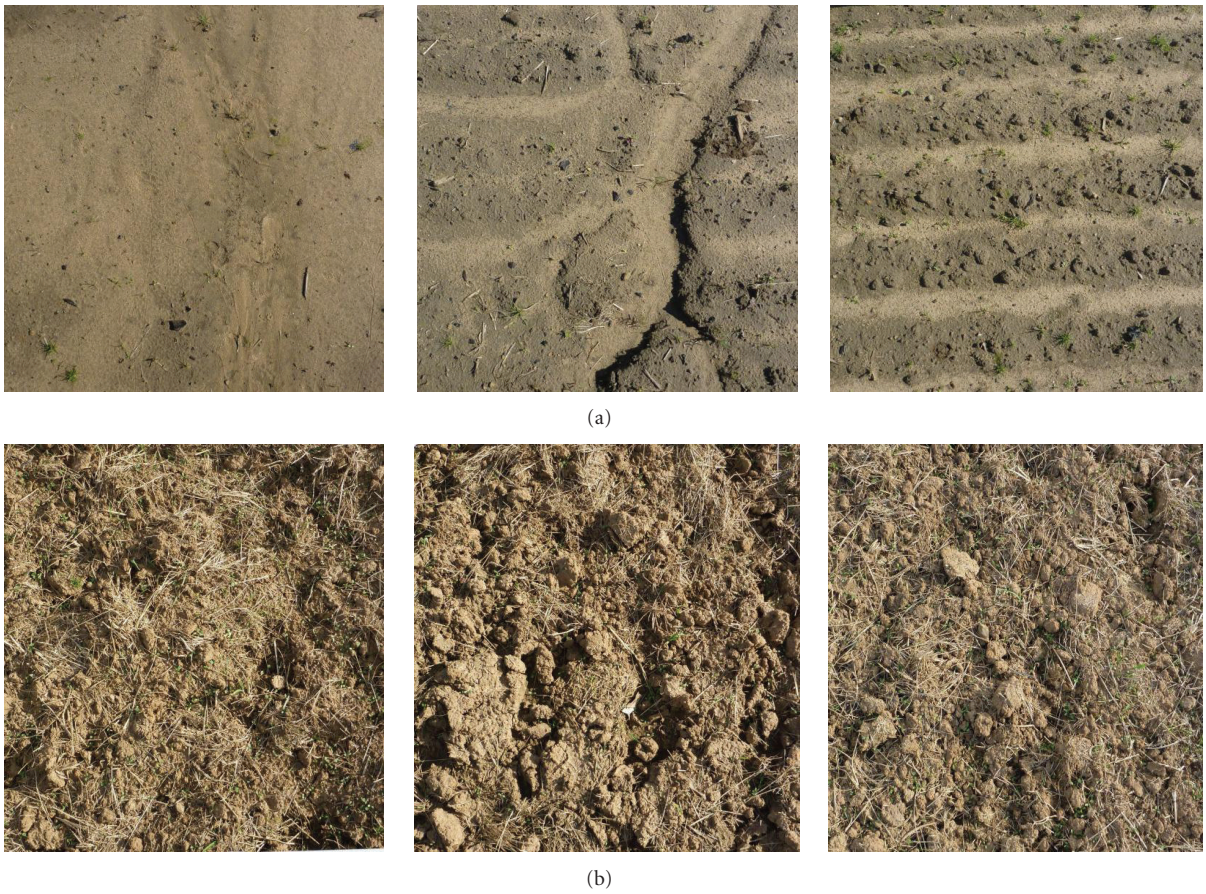


Figure 3: Soil surface roughness (SSR) images of PBI plots (1 cob, 2 cob, and 3 cob on (a) L-R) and JBP plots (1 jbp, 2 jbp, and 3 jbp on (b) L-R). Each measurement was taken three times.

A frame of 1 m^2 was used to take the images and assure that the same area was chosen for each subplot reading. The camera was set on a Slik tripod to photograph the entire 1.0 m^2 area in a single frame and to assure that the camera lens was placed parallel to the soil surface at a height of 1.65 m. The focal angle and the distance from the lens to the ground were constant throughout to ensure that the resolution would be the same in all photos. The shadows cast by the soil microrelief were analysed with byte map histograms using Corel Draw Photo Paint (Corel Corporation 1992–1996) software. After identification on the histogram, the shaded points were converted to a black surface against a white background. The shadow index was then computed as the percentage of blackout of the total number of pixels. The results from each subplot were an average measurement of the three-directional images. Pore network and voxel-based soil porosity were evaluated using computed tomography.

2.3. Computed Tomography Measurements. The images for tomography measurements were taken using an XRADIA micro-XCT-400 that belongs to the Australian Centre for microscopy and microanalysis, University of Sidney (Figure 4). In addition to the quantification of porosity, this technique allows visualisation of the pore distribution and shape as well as connectivity and aggregation.

The samples used in the CT scanner were taken from the soil surface at the six field locations. The height of each sample was 4 mm, and the diameter was 2 mm. The samples were fixed to the structure of the scan without any further preparation. Each sample was taken from the subplot at the same location where the soil surface roughness was measured. The resolution of the images was 5 μm, and a threshold algorithm was applied to convert the greyscale images to binary images, with black corresponding to the pore space and white and grey to soil matrix, including organic matter.

Table 2: Soil surface roughness (SSR) and porosity at the soil surface, expressed in percentages. The values in parentheses are the standard deviations.

Location	Sample ID	Porosity (%)	SSR (%)
PBI	1N		14.6 (2.36)
	1W		14.08 (1.41)
	1S		15.50 (0.01)
	total 1	38.07	14.25 (1.41)
	2N		22.22 (0.16)
	2W		21.78 (0.84)
	2S		25.58 (1.06)
	total 2	42.92	23.19 (1.96)
	3N		21.32 (0.36)
	3W		25.46 (0.63)
	3S		22.10 (0.15)
	total 3	45.01	22.96 (1.82)
	Total	**42.00 (3.56)**	**20.13 (5.10)**
JBP	1N		44.60 (1.13)
	1W		43.50 (1.12)
	1S		42.68 (0.00)
	total 1	17.55	43.60 (1.02)
	2N		39.20 (2.38)
	2W		42.27 (0.52)
	2S		39.68 (0.31)
	total 2	18.23	40.39 (1.84)
	3N		40.83 (2.35)
	3W		41.58 (3.51)
	3S		38.08 (0.48)
	Total 3	19.28	40.16 (2.15)
	Total	**18.35 (0.87)**	**41.38 (1.92)**

This equipment functioned as a high-resolution, non-destructive 3D X-ray imaging system with a spatial resolution of $<1\,\mu$ and $0.56\,\mu$m pixel size. This technique provides the full three-dimensional structure of soil samples with minimal resolution dependence on the size and its preparation. The images were then evaluated using image analysis software, which allowed the exploration of soil porosity and the visualisation of soil microstructure.

The image processing began with reduction in greyscale to reduce brightness. A sigma filter was applied to reduce noise and preserve structure within images. The smoothed images were segmented to produce binary images for the analysis of porosity and interconnectivity. For calculation, the images were subjected to greyscale segmentation using different threshold values. The percentage porosity was calculated as a percentage of black space, and the differences between two locations were statistically compared using analysis of variance.

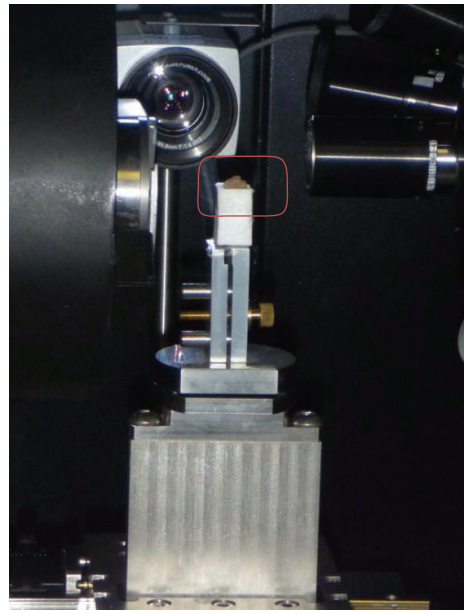


Figure 4: CT Scanner (Xradia CTX 400) with sample.

3. Results

Table 2 shows the results of SSR for each plot and from each direction expressed in percentage of shadows. There are no significant differences in measurements for the same subplot at sampling positions (N, S, and W). For this reason, only the averages of each subplot are considered, allowing direct comparison with the porosity percentages. Results from different directions are in agreement with results from previous studies at different geographical locations, where this technique was validated with other well-tested methods [1, 2].

Comparing the results in Table 2 with the resulting images of SSR patterns for both sites (Figure 2) shows that the numerical results express the roughness differently from the images of each subplot.

Based on both sources of roughness measurement, the SSR results from PBI facilities were lower because they showed a higher degree of erosion than the plots at JBP, and the PBI plots were not tilled since the beginning of fall. Additionally, the degree of erosion of the PBI subplots varies, mainly due to rainfall. The images of the first subplot show the highest degree of erosion, followed by the second and third subplots. The last two subplots show very similar results for the percentage of shadows. At the second subplot, the erosion is primarily related to the ridges created by rainfall erosion, and at the last subplot, the remaining erosion is created by tillage operations and the increase in oriented soil surface roughness.

In contrast, the results from JBP subplots give higher, more consistent levels of roughness. The percentages of shadows obtained for JBP follow those for PBI. These values are due to the type and later time of tillage and conservational practices, including the addition of grass residuals to reduce erosion.

Table 3: Analyses of variance and covariance for SSR and porosity percentages.

(a)

Statistics of regression	
Multiple correlation	0.993902605
Coefficient R^2	0.987842388
Adjusted R^2	0.983789851
Standard error	1.790239741
Observations	5

(b)

	Degrees of freedom	SS	Average of the squared deviations	F	F Critique value
Regression	1	781.237405	781.237405	243.7589899	0.000571022
Residual	3	9.61487499	3.20495833		
Total	4	790.85228			
Covariance		−120.3479056			
Corr. Coef.		−0.904242715			

The soil surface roughness was compared to the porosities to study their potential relation and to observe how different management approaches influence the inner structures of soil, particularly porosity and its influence on erosion.

The resulting inner structures found in the sampling aggregates from different sites and subplots are shown in Figure 6. These images represent the complete aggregate and a visualisation of the inner core of samples.

The images of samples from the JBP site present a stronger degree of aggregation than samples from PBI (Figures 5 and 6). The degree of cementation is high in clumps from JBP, indicating that these soils are less susceptible to erosion than soils with a low degree of aggregation, such as the PBI samples. Because both soils have the same texture, the higher degree of aggregation in the PBI samples seems to be produced by a higher presence of organic matter and organisms. The higher presence of macroaggregates and greater SSR percentages (Figures 5 and 6) is related to tillage practices and residual organic matter used as cover. Thus, the porosity at JBP is distributed in well-organised channels. These results are based on the literature reviews and indicate that the observed porosity is mainly related to structural porosity with the organisation of interaggregates [16, 17].

The percentages of porosity found in both experimental sites (Table 2) show that the porosity at PBI is almost twice that calculated for JBP. However, examination of the inner structure (Figure 6) shows that PBI samples lack aggregation and are more susceptible to erosion. Therefore, PBI has a higher percentage of textural porosity, where particles form intraaggregates with cohesion mainly associated with soil mineralogy and where microaggregates are the dominant hierarchical order [18]. This result is supported by the lowest percentage of SSR showing less resistance to erosion [1, 2].

4. Discussion

Comparing the results obtained for SSR with porosity values of the experimental plots shows that plots with higher percentages of SSR have lower percentages of porosity, as is the case for the PBI samples.

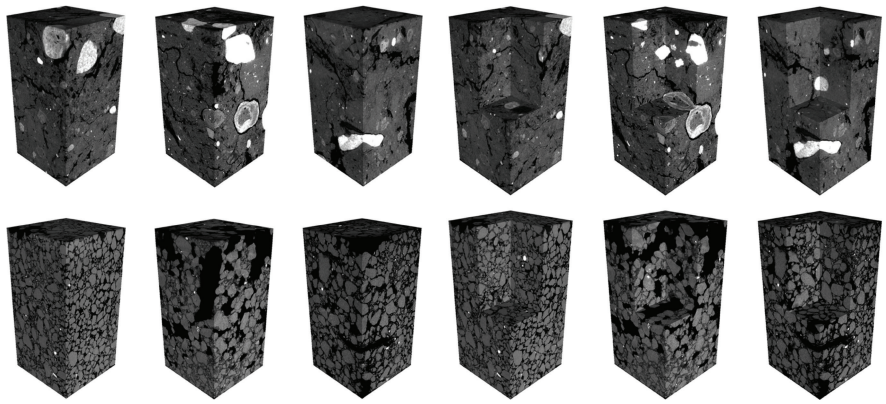
The regression analysis was used to compare SSR and porosity, indicating a negative correlation with $R^2 = 0.99$, significant at $P < 0.0015$ (Figure 7; Table 3).

The resulting relation between the two parameters corroborates the erosion properties of both sites. Soils from PBI show low aggregation, high porosity, very low microrelief of the soil surface, and low resistance to erosion, most of which is induced by rainfall, as indicated by the presence of ridges. The 3D images show that the high porosity is mainly due to textural porosity with a low degree of aggregation, and the absence of vegetal coverture and low percentage of SSR make the soil highly sensitive to erosion.

However, the images of subplots from JBP show that soil management practices have increased the SSR associated with the formation of macroaggregates and the presence of interaggregates [16, 17, 19]. Structural porosity seems to prevail over textural porosity here. These properties give the JBP plots a high degree of stabilisation, with the formation of macroaggregates due to the presence of organic matter and possibly microorganisms (Figures 5 and 6) [15, 19], and a lower susceptibility to erosion than PBI [19].

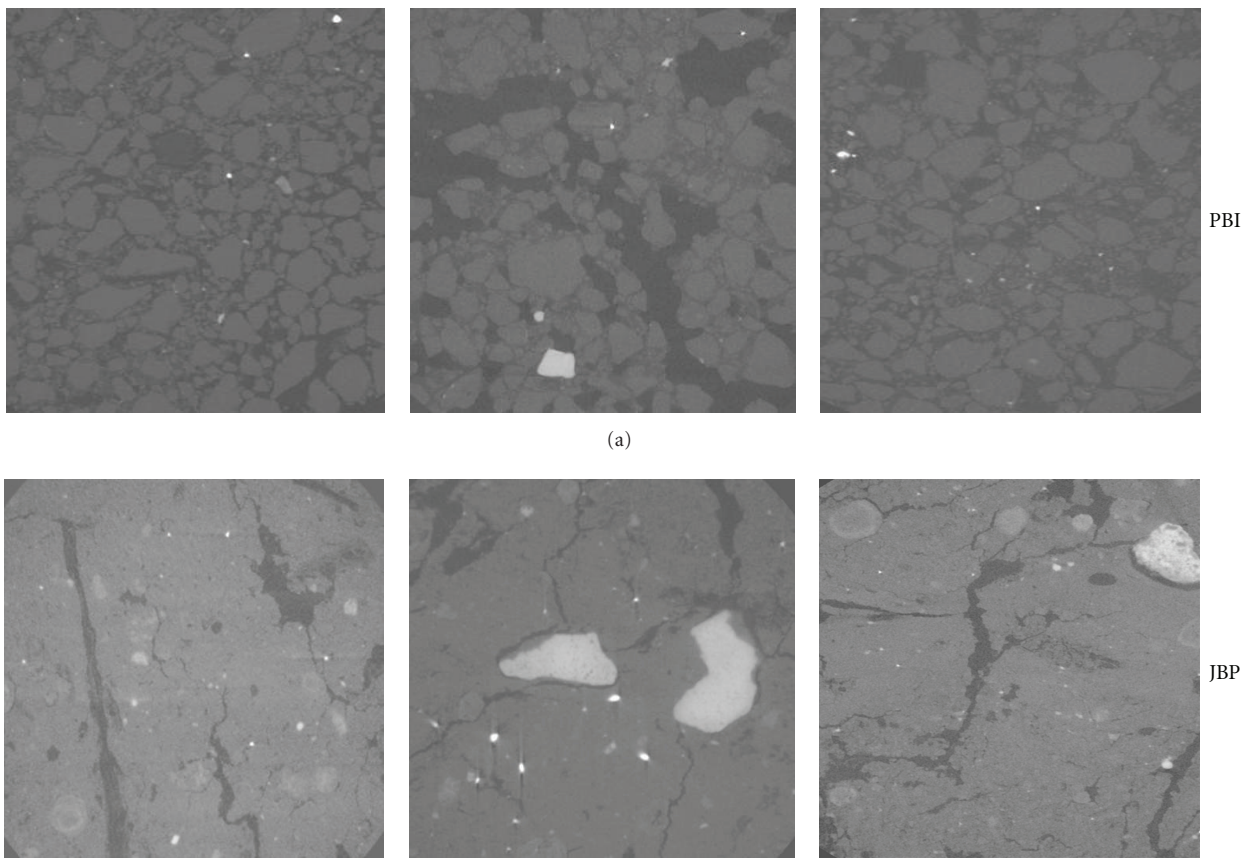
The observed differences in structures and SSR are related to the soil tillage practices and the existence of grass residues that protect the soil surface from erosion, enhancing the inner structure and well-organised soil interconnectivity of the JBP plots [20–22].

In comparison, the highest soil surface roughness among PBI plots is closely related to the lack of soil structure



(a) (b)

Figure 5: Scanned soil samples. PBI samples are on the bottom and JBP samples are on the top. A) complete scanned sample, B) internal view of scanned sample.



(a)

(b)

Figure 6: Internal distribution of porosity. PBI is shown at the top and JBP is shown at the bottom. For each site, samples 1, 2 and 3 are ordered L-R.

due to the high erosion rate caused by the lack of soil surface protection and the soil management practices, that is, the exposure of bare surface to water erosion, destroying the inner aggregation and interconnectivity of the soil [19].

5. Conclusion

Based on the results obtained in this study, SSR is closely related to soil aggregation, structural porosity, and the presence of macroaggregates. The high porosity of PBI

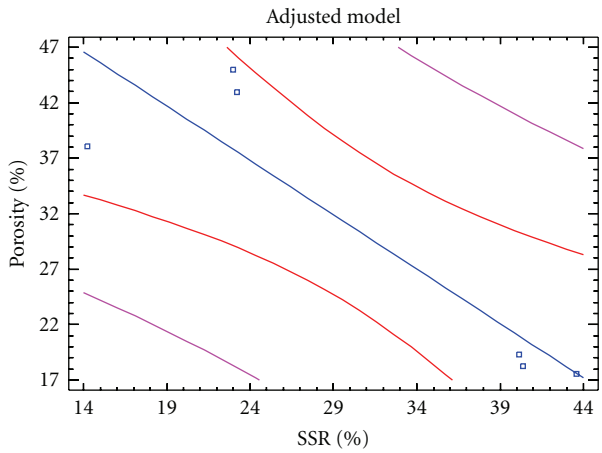


Figure 7: Regression analysis of porosity (%) versus SSR (%).

samples with low SSR percentages indicates high disturbance and the lack of macroaggregation of soil, whereas images of JBP samples with higher SSR values and lower porosities indicate a higher presence of macroaggregates and a higher degree of stabilisation.

Both parameters indicate that the implementation of management conservation practices, particularly conservation tillage practices and use of residual coverage on the soil surface, prevent erosion and enhance soil macroaggregation and stabilisation due to organic matter and organisms being less susceptible to soil surface erosion.

Therefore, the SSR values are increased as soil is tilled as a conservation practice, and the microstructures indicate aggregation. In this sense, the high percentage of SSR produces lower erosion rates in soil, promoting the formation of macroaggregates. Here, soil porosity and soil SSR have a negative correlation, with an R^2 value of 99.9%.

From these results, it can be concluded that the study of SSR and the observation of the inner structures of soil are complementary and can be used to evaluate the influence of soil management practices on soil erosion susceptibility.

However, further research on the comparison of soil types and different management practices must be performed to confirm the findings of this study relating soil microstructures and SSR as well as to complement other soil properties that are thought to be interrelated.

References

[1] R. G. Moreno, A. Saa Requejo, A. M. Tarquis Alonso, S. Barrington, and M. C. Díaz, "Shadow analysis: a method for measuring soil surface roughness," *Geoderma*, vol. 146, no. 1-2, pp. 201–208, 2008.

[2] R. G. Moreno, M. C. Díaz Álvarez, A. M. Tarquis, A. Paz González, and A. Saa Requejo, "Shadow analysis of soil surface roughness compared to the chain set method and direct measurement of micro-relief," *Biogeosciences*, vol. 7, no. 8, pp. 2477–2487, 2010.

[3] N. Nunan, K. Ritz, M. Rivers, D. S. Feeney, and I. M. Young, "Investigating microbial micro-habitat structure using X-ray computed tomography," *Geoderma*, vol. 133, no. 3-4, pp. 398–407, 2006.

[4] D. Or, B. F. Smets, J. M. Wraith, A. Dechesne, and S. P. Friedman, "Physical constraints affecting bacterial habitats and activity in unsaturated porous media-a review," *Advances in Water Resources*, vol. 30, no. 6-7, pp. 1505–1527, 2007.

[5] K. E. Saxton, "Wind erosion and its impact on off-site air quality in the Columbia Plateau—an integrated research plan," *Transactions of the American Society of Agricultural Engineers*, vol. 38, no. 4, pp. 1031–1038, 1995.

[6] L. J. Cihacek, M. D. Sweeney, and E. J. Deibert, "Characterization of wind erosion sediments in the red river valley of North Dakota," *Journal of Environmental Quality*, vol. 22, no. 2, pp. 305–310, 1993.

[7] R. Röhrig, M. Langmaack, S. Schrader, and O. Larink, "Tillage systems and soil compaction—their impact on abundance and vertical distribution of Enchytraeidae," *Soil and Tillage Research*, vol. 46, no. 1-2, pp. 117–127, 1998.

[8] M. Kutílek and L. Jendele, "The structural porosity in soil hydraulic functions—a review," *Soil and Water Research*, vol. 3, supplement 1, pp. S7–S20, 2008.

[9] A. R. Dexter, "Soil physical quality: part I. Theory, effects of soil texture, density, and organic matter, and effects on root growth," *Geoderma*, vol. 120, no. 3-4, pp. 201–214, 2004.

[10] A. R. Dexter, E. A. Czyz, G. Richard, and A. Reszkowska, "A user-friendly water retention function that takes account of the textural and structural pore spaces in soil," *Geoderma*, vol. 143, no. 3-4, pp. 243–253, 2008.

[11] A. Papadopoulos, N. R. A. Bird, A. P. Whitmore, and S. J. Mooney, "Investigating the effects of organic and conventional management on soil aggregate stability using X-ray computed tomography," *European Journal of Soil Science*, vol. 60, no. 3, pp. 360–368, 2009.

[12] I. M. Young, J. W. Crawford, and C. Rappoldt, "New methods and models for characterising structural heterogeneity of soil," *Soil and Tillage Research*, vol. 61, no. 1-2, pp. 33–45, 2001.

[13] J. W. Crawford, J. A. Harris, K. Ritz, and I. M. Young, "Towards an evolutionary ecology of life in soil," *Trends in Ecology and Evolution*, vol. 20, no. 2, pp. 81–87, 2005.

[14] J. M. Oades, "The role of biology in the formation, stabilization and degradation of soil structure," *Geoderma*, vol. 56, no. 1–4, pp. 377–400, 1993.

[15] F. J. Larney, A. J. Cessna, and M. S. Bullock, "Herbicide transport on wind-eroded sediment," *Journal of Environmental Quality*, vol. 28, no. 5, pp. 1412–1421, 1999.

[16] A. Hadas, "Long-term tillage practice effects on soil aggregation modes and strength," *Soil Science Society of America Journal*, vol. 51, no. 1, pp. 191–197, 1987.

[17] A. R. Dexter, "Advances in characterization of soil structure," *Soil and Tillage Research*, vol. 11, no. 3-4, pp. 199–238, 1988.

[18] C. M. Monreal, M. Schnitzer, H. R. Schulten, C. A. Campbell, and D. W. Anderson, "Soil organic structures in macro and microaggregates of a cultivated Brown Chernozem," *Soil Biology and Biochemistry*, vol. 27, no. 6, pp. 845–853, 1995.

[19] A. N. Kravchenko, G. P. Robertson, X. Hao, and D. G. Bullock, "Management practice effects on surface total carbon: differences in spatial variability patterns," *Agronomy Journal*, vol. 98, no. 6, pp. 1559–1568, 2006.

[20] I. M. Young and K. Ritz, "Tillage, habitat space and function of soil microbes," *Soil and Tillage Research*, vol. 53, no. 3-4, pp. 201–213, 2000.

[21] I. M. Young, J. W. Crawford, N. Nunan, W. Otten, and A. Spiers, "Microbial distribution in soils: physics and scaling," *Advances in Agronomy C*, vol. 100, pp. 81–121, 2009.

[22] K. M. Hati, A. Swarup, A. K. Dwivedi, A. K. Misra, and K. K. Bandyopadhyay, "Changes in soil physical properties and organic carbon status at the topsoil horizon of a vertisol of central India after 28 years of continuous cropping, fertilization and manuring," *Agriculture, Ecosystems and Environment*, vol. 119, no. 1-2, pp. 127–134, 2007.

[23] J. Six, E. T. Elliott, and K. Paustian, "Aggregate and soil organic matter dynamics under conventional and no-tillage systems," *Soil Science Society of America Journal*, vol. 63, no. 5, pp. 1350–1358, 1999.

[24] N. G. Juma, "Interrelationships between soil structure/texture, soil biota/soil organic matter and crop production," *Geoderma*, vol. 57, no. 1-2, pp. 3–30, 1993.

[25] M. Deurer, D. Grinev, I. Young, B. E. Clothier, and K. Müller, "The impact of soil carbon management on soil macropore structure: a comparison of two apple orchard systems in New Zealand," *European Journal of Soil Science*, vol. 60, no. 6, pp. 945–955, 2009.

[26] M. Feser, J. Gelb, H. Chang et al., "Sub-micron resolution CT for failure analysis and process development," *Measurement Science and Technology*, vol. 19, no. 9, Article ID 094001, 2008.

[27] J. F. Darbyshire, S. J. Chapman, M. V. Cheshire et al., "Methods for the study of interrelationships between micro-organisms and soil structure," *Geoderma*, vol. 56, no. 1–4, pp. 3–23, 1993.

[28] J. M. Blair, R. E. Falconer, A. C. Milne, I. M. Young, and J. W. Crawford, "Modeling three-dimensional microstructure in heterogeneous media," *Soil Science Society of America Journal*, vol. 71, no. 6, pp. 1807–1812, 2007.

[29] S. de Gryze, L. Jassogne, J. Six, H. Bossuyt, M. Wevers, and R. Merckx, "Pore structure changes during decomposition of fresh residue: X-ray tomography analyses," *Geoderma*, vol. 134, no. 1-2, pp. 82–96, 2006.

[30] B. W. Hapke, "A theoretical photometric function for the lunar surface," *Journal of Geophysical Research*, vol. 68, pp. 4571–4586, 1963.

[31] R. F. Isbell, *The Australian Soil Classification, Revised Edition*, CSIRO, Canberra, Australia, 2002.

[32] Bureau of Meteorology, Australian Government, http://www.bom.gov.au/, 2010.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 650206, 17 pages
doi:10.1155/2012/650206

Research Article

Nitrate-Nitrogen Leaching from Onion Bed under Furrow and Drip Irrigation Systems

Parmodh Sharma, Manoj K. Shukla, Theodore W. Sammis, and Pradip Adhikari

Department of Plant and Environmental Sciences, New Mexico State University, MSC 3Q, P.O. Box 30003, Las Cruces, NM 88003, USA

Correspondence should be addressed to Parmodh Sharma, sharmap@nmsu.edu

Received 4 December 2011; Accepted 24 February 2012

Academic Editor: María Cruz Díaz Álvarez

Water is a limited resource for crop production in arid areas of Southern New Mexico. The objectives of this study were to estimate the amount and depth of water and nitrate-nitrogen (NO_3-N) fronts, water and NO_3-N balances, and irrigation efficiencies for two onion (*Allium cepa* L.) fields under furrow and drip irrigation systems. Monthly soil samples were analyzed for NO_3-N and chloride concentration for two onion growing seasons starting September 2006 to August 2009. The average amount of NO_3-N in the soil water estimated by chloride tracer technique varied from 97.4 to 105.2 mg L^{-1} for furrow and 65.2 to 66.8 mg L^{-1} for drip-irrigated fields for the 60- to 200-cm depth. The NO_3-N loadings below the rooting zone ranged from 145 to 150 kg ha^{-1} for furrow- and 76 to 79 kg ha^{-1} for drip-irrigated fields. The irrigation efficiencies varied from 78 to 80% for furrow- and 83% for drip- and N application efficiencies (NAEs) were 35 to 36% for furrow- and 38 to 39% for drip-irrigated fields. Small N fertilizer applications, delayed until onion bulbing starts, and water applications, preferably through drip irrigation, are recommended to reduce deep percolation and increase nitrogen and water efficiencies.

1. Introduction

Among all the elements needed for plant growth, nitrogen (N) is considered the most important fertilizer element applied to soils because crop requirements for N are high compared with requirements for phosphorous (P), potassium (K), and other essential plant nutrients [1]. However, solubility of nitrate (NO_3) sources in water can cause rapid movement through soils, and among the various sources of N loss in agricultural fields, leaching is considered a major source of NO_3-N loss under normal agricultural practices [1].

Crops differ in rooting depths, rooting densities, N and water requirements, and plant uptake efficiencies [2], and the percolation of NO_3-N to deeper soil layers depends on the cropping systems. In addition to N fertilizers and water applied by irrigation or received through precipitation, type of irrigation system and soil physical properties also play important roles in NO_3-N leaching to groundwater [3, 4]. In arid regions like New Mexico, excess irrigation is also applied to flush salts out of the rooting zone to control soil salinization [5], leading to high N leaching. Nitrate loading to groundwater ranged from 165 kg ha^{-1} NO_3-N for irrigated sweet corn (*Zea mays*) to 366 kg ha^{-1} NO_3-N for irrigated potato (*Solanum tuberosum*) on sandy soils in Wisconsin [6]. In the Santa Maria, California, region, where crops such as potatoes, beans (*Phaseolus*), cauliflower (*Brassica oleracea*), celery (*Apium graveolens*), lettuce (*Lactuca sativa*), and broccoli (*Brassica oleracea*) are grown on different soils (loam, loamy sand, and sandy loam), mean NO_3-N concentrations below the crop rooting zones ranged from 60 to 204 kg ha^{-1} [7]. Similarly, for spring barley (*Hordeum distichum* L.) planted in sandy soils and fertilized with 100 kg N ha^{-1}, leaching losses of 65 kg N ha^{-1} were reported [8]. Nitrate-N loading to groundwater was higher for onion than alfalfa (*Medicago sativa* L.) and chile (*Capsicum annuum*) under a furrow irrigation system of arid New Mexico [3].

The N use efficiency of onion has been reported to range from 15% [9] to 30% [10] in furrow irrigation systems. Drip

irrigation systems have been reported to reduce percolation of NO_3-N below the vadose zone of bell pepper (*Capsicum annuum*) [11]. Drip irrigation systems have the potential to supply water and N directly to onion roots and reduce water and NO_3-N leaching to the deeper soil layers. Drip systems reported to apply 22% [12] to 30% [13] less water than furrow irrigation systems. Higher onion yields, larger bulb sizes, less NO_3-N leaching, higher water use efficiency, and higher N fertilizer use efficiency were reported under drip irrigation systems compared to furrow irrigation systems [14].

The amount of N leaching can be estimated by measuring the concentration of chloride in irrigation water, and N and chloride concentrations in the soil below the rooting zone of a crop [3, 15, 16]. In these studies, soil N and chloride analyses were made after the harvest of crops to determine the growing season leaching fraction and amount of N reaching to the ground water. Most soils can supply N by mineralization, and P and K by weathering of minerals, but cannot supply a considerable amount of chloride. Also, chloride is involved in few biological reactions other than plant uptake and is present in most irrigation water [17]. Thus, chloride is uniquely suited as a tracer element to estimate N leaching below plant rooting zones, although the assumption that chloride is a conservative tracer and is not adsorbed or released by soil may not be always valid [18, 19]. There may also be errors in the chloride balance unless all sources or sinks of chloride are determined [20]. Despite these limitations, the chloride tracer method has been used to determine NO_3-N loading to the groundwater because the method is fast and easy to use and is less expensive method than constructing a lysimeter. However, there is a need to expand this technique by including soil nitrate-N, chloride, and irrigation measurements throughout the growing season. Therefore, objectives of this study were to determine how the existing management practices of onion in New Mexico can be improved to reduce NO_3-N leaching and improve water application and N efficiencies. The hypothesis of the study was that the transport behavior of chloride in soils is similar to that of NO_3-N, and that chloride can be used as a tracer to determine the amount of NO_3-N reaching to the groundwater throughout the growing season of onion in New Mexico.

2. Materials and Methods

2.1. Site Description. This study was conducted during two onion growing seasons from 2006 to 2009 in two fields under two irrigation systems. In furrow-irrigated field, onion was planted on 2 November 2006 and harvested on 14 July 2007; this period was considered as growing season 1 whereas onion planted on 25 September 2008 and harvested on 18 June 2009 was considered as growing season 2. Similarly, onion planted on 27 September 2006 and harvested on 8 June 2007 was considered as growing season 1 and onion planted on 23 February 2008 and harvested on 10 August 2008 was considered as growing season 2 in drip-irrigated field. For the previous eight years, both fields were planted with sudan grass (*Sorghum sudanense*) after the onion harvest of growing season 1 until onion planted for growing season 2. No N fertilizer was applied to the sudan grass and it was harvested three times during the growing season in the furrow-irrigated field, and one time in the drip-irrigated field where the biomass was left on the soil surface at the end of each cut and incorporated into the soil at the end of the growing season. The furrow-irrigated onion field was located on the Leyendecker Plant Science Research Centre (PSRC) at 32°11′N and 106°44′W and the drip-irrigated onion field was located on the Fabian Garcia Research Center (FGRC) at 32°16′N and 106°46′W of New Mexico State University (NMSU) near Las Cruces, New Mexico. Soils at both sites were classified as Glendale (fine-silty, mixed, calcareous, thermic typic Torrifluvents)-Harkey (coarse-silty, mixed, calcareous, thermic typic Torrifluvents) series [21]. The average annual precipitation for the experimental sites is 25.3 cm, and the average annual temperature is 17.7°C. The groundwater table was below 2 m in depth at both experimental sites, and both fields were irrigated with groundwater [22].

2.2. Seedbed Preparation. The fields at PSRC and FGRC were prepared under conventional tillage that included disking, chiseling, plowing, leveling, listing, and bed shaping. Disking was done to alleviate surface soil compaction and to incorporate sudan grass stubble into the soil in both the fields. Triple superphosphate was broadcasted at a rate of 200 kg P_2O_5 ha^{-1} on both onion fields before moldboard plowing. Both fields were laser leveled, bed shaped with 56 cm beds and 46 cm furrows, and two rows of onions transplanted into each bed 28 cm apart on 2 November 2006 and 25 September 2008 in the field located at PSRC and on 27 September 2006 and 23 February 2008 in the field located at FGRC. The length of row was 210 m at PSRC and 132 m at FGRC. The field at the PSRC was furrow irrigated while the field at the FGRC was drip-irrigated using T-tape (T-Tape: TSX-508-08-670, T-Systems, San Diego, CA) with an emitter spacing of 20 cm and a flow rate of 0.22 cm h^{-1} laid in the center of each bed between the two onion rows at a 10 cm depth. Onions were irrigated 19 times with total gross water application of 95 cm, and 21 times with total gross water application of 100 cm during growing seasons 1 and 2, respectively, in the furrow-irrigated field. Onion in drip-irrigated field received a total gross water application of 81 cm using 42 irrigation applications in growing season 1 and 72 cm using 40 irrigation applications in growing season 2. The amount of water applied during each irrigation event was measured with a flow meter (McCrometer, Inc., Hemet, CA) in each field. The precipitation data was obtained from weather stations located at each experimental site. The water application efficiency was calculated as the ratio of total water stored in the onion rooting zone during irrigation ($E_t + \Delta$ storage) and total water applied [23].

Urea ammonium nitrate (URAN) liquid fertilizer was the source of N applied in both the fields. Urea ammonium nitrate was applied at a rate of 49.2 kg N ha^{-1} per irrigation in the furrow-irrigated field. During growing season 1, six

irrigations with URAN fertilizer were applied during 2007 in the months of February, March, and April (total of 295 kg N ha^{-1}) and same number of fertilizer applications (total of 295 kg N ha^{-1}) were also made during growing season 2 in the furrow-irrigated field. In the drip-irrigated field, URAN was applied at an average rate of 36.5 kg N ha^{-1} through eight irrigations (total of 292 kg N ha^{-1}) during growing season 1 and 35.8 kg N ha^{-1} per irrigation through eight irrigations (total of 286 kg N ha^{-1}) during growing season 2 through the drip tape via injectors (H. E. Anderson company, Muskogee, OK). Nitrogen application efficiency was calculated as the ratio of total N uptake and total N applied. Nitrogen use efficiency was calculated as the ratio of total N uptake and total available N (initial soil N at a 0–50 cm depth + N fertilizer applied) [14].

2.3. Soil Sampling and Analysis. Twenty-four soil core samples were collected from three locations and four depths (0–10, 10–30, 30–40, and 40–60 cm) from both fields (2 fields × 3 locations × 4 depths = 24). Cores were trimmed in the laboratory, and soil bulk density (BD) was determined by the core method [24]. After determining soil BD, all soil cores were immediately placed in a water tray for 1 to 2 d at room temperature (24°C) to fully saturate by capillary rise, and saturated hydraulic conductivity (K_s) was determined by the constant head method [25].

Bulk soil samples were collected from each field at the end of each month from six depths (0–10, 10–30, 30–40, 40–60, 60–85, and 85–110 cm) and three locations (2 fields × 3 locations × 6 depths = 36 soil samples) from September 2006 to August 2009. Gravimetric soil moisture content for each bulk soil sample was determined immediately after sampling [26]. The gravimetric water content was multiplied by BD to calculate the volumetric soil water content (θ). The rest of each soil sample was stored in a cold room at 4°C until further analysis. Irrigation water samples collected during each month from both fields were analyzed for electrical conductivity (EC), pH, nitrate, and chloride from both fields. Soil samples were also collected from 150- and 200-cm depths during the last week of March and at the time of harvest from both fields to examine change in the concentration of NO_3-N and chloride.

Bulk soil samples were air dried for 48 hours, ground, and passed through a 2-mm diameter sieve. Fifty-one grams of sieved soil (<2 mm diameter) was used for particle size analysis by the hydrometer method [27]. Nitrate-N was determined by an automated spectrophotometric method using a Technicon autoanalyzer from soil-KCl extracts. The extracts were prepared by adding 25 mL of 2.0 M KCl to 2.5 g of soil, shaking the suspension for 1 hour, and filtering through filter paper (Whatman number 2) [28]. Chloride was determined with a 798 MPT Titrino titrator using silver nitrate solution (0.1 M). Electrical conductivity (EC) and pH were measured for a solution of (1 : 2 soil : water) using an EC electrode and 72 pH meter, respectively. Soil samples from six depths in both fields were also analyzed for organic matter (OM), exchangeable sodium percentage (ESP), sodium adsorption ratio (SAR), phosphorous (P), and potassium (K) at the New Mexico State University Soil and Water Testing (SWAT) Laboratory.

2.4. Onion Rooting Depth and Biomass. Onion rooting depth was determined by excavating two pits at each field just before each harvest. Two plants from each pit were excavated along with their roots at a depth increment of 20 cm from the top of the bed to a depth of 50 cm. The soil was gently washed in the lab and roots were separated, air dried, and weighed.

Onion samples were collected at monthly intervals from February until harvest during growing season 2 in furrow- and drip-irrigated fields for N uptake determination. For aboveground biomass determination, the crop was manually harvested before each harvest from four randomly selected plots (2.4 m × 1 m) at each field. Wet and dry plant biomass and onion yields were determined, separately, for each plot on a per-hectare basis. The plant samples were weighed fresh, then dried at 68°C for 72 h and reweighed to determine plant moisture content. The air-dried onion bulbs were analyzed for NO_3-N, total N and chloride at the SWAT lab of NMSU.

2.5. Crop Coefficient. The reference evapotranspiration (E_{t_o}) for grass using Penman's equation was obtained from NMSU weather station located at each experimental field. The crop coefficient (K_c) was calculated as follows:

$$K_c = B_0 + B_1 \sum_{i=1}^{n} \mathrm{GDD} + B_2 \left(\sum_{i=1}^{n} \mathrm{GDD} \right)^2 + B_3 \left(\sum_{i=1}^{n} \mathrm{GDD} \right)^3, \tag{1}$$

where i = day and n = total number of days, B_0 is the intercept, and B_1, B_2, and B_3 are regression coefficients for onion [29]. Growing-degree-days (GDD) were calculated as

$$\mathrm{GDD} = \frac{(T_{\max} + T_{\min})}{2} - T_b, \tag{2}$$

where $T_{\max}$ = daily maximum temperature (°C); $T_{\min}$ = daily minimum temperature (°C); T_b = base temperature (°C). The base temperature was set at 4°C [30]. K_c is defined as the ratio of crop evapotranspiration (E_t) and E_{t_o}; therefore, K_c was multiplied by E_{t_o} to calculate E_t.

2.6. Soil Water Content. Diurnal variations of θ in the onion beds were monitored by time domain reflectometry (TDR) sensors (Campbell Scientific, Inc., Logan, Utah). Each TDR system included one CR 10X datalogger, one SDMX50 multiplexer, one TDR 100, and eight CS-640 probes powered by a 12 V deep-cycle battery at each experimental site. A set of two probes was installed at depths of 20 and 50 cm from the top of the onion bed at four locations in each experimental field. Total of sixteen TDR sensors were installed and programmed to provide half hourly readings for the entire growing seasons at both experimental sites.

2.7. Chloride Tracer Technique. Chloride is present in almost every source of irrigation water and is either taken up by plants or remains in the water. Chloride is assumed

to be a conservative ion in this approach. As most soils do not adsorb or release chloride, the irrigation leaching fraction (LF) can be calculated by taking the ratio of chloride concentration in irrigation water to chloride concentration in the drainage water [15]. The LF is defined as the fraction of water moving below the rooting zone portion of the soil profile and is expressed as

$$\mathrm{LF} = \frac{(E_t \mathrm{Cl}_i) - \mathrm{Cl}_c}{\left(E_t \mathrm{Cl}_p\right) - \mathrm{Cl}_c}, \tag{3}$$

where E_t is the seasonal E_t (kg H_2O ha^{-1}), Cl_i is the chloride concentration (kg Cl^{-1} per kg H_2O) in the irrigation water, Cl_p is the chloride concentration (kg Cl^{-1} per kg H_2O) in the soil water below the rooting zone, and Cl_c is the amount of chloride taken up by the crop (kg Cl^{-1} ha^{-1}) [16]. Irrigation efficiency (IE) was calculated by subtracting the LF from 1. The amount of N (kg ha^{-1}) leaching in the groundwater (N_p) was calculated as

$$\mathrm{N}_p = \mathrm{Cl}_a \frac{(\mathrm{NO_3}^- - \mathrm{N})_s}{\mathrm{Cl}_s}, \tag{4}$$

where Cl_a is the amount of chloride (kg ha^{-1}) in the irrigation water and was calculated based on the concentration of chloride in the irrigation water, the volume of irrigation water, and the amount of water taken up by the crop, (NO_3-N)$_s$ is the concentration of NO_3-N in the soil (mg (NO_3-N) mg^{-1} soil), and Cl_s is the chloride ion concentration in the soil (mg Cl^{-1} mg^{-1} soil).

2.8. Water Balance. The following equation was used to estimate the water balance in both fields:

$$\Delta S = P + I - E_t - \mathrm{DP}, \tag{5}$$

where ΔS is change in soil water storage (cm), P is precipitation (cm), I is irrigation (cm), E_t is evapotranspiration (cm), and DP is deep percolation (cm).

2.9. Leaching Depth Calculations. The length of the roots and shoots of onion seedlings ranged from 6 to 10 cm during transplanting. Thus, actual rooting depth of onion seedlings was selected as 10 cm at the time of transplant. The increase in rooting zone depth (RD) during the growing season was calculated as follows:

$$\mathrm{RD} = \mathrm{RGC} \times \mathrm{GDD}, \tag{6}$$

where RGC is root growth coefficient equal to 0.0254 cm°C^{-1} [31]. Amount of available water content (AWC) stored in the rooting zone increased with an increase in depth of rooting zone and was calculated as

$$\mathrm{AWC} = \mathrm{RD} \times \partial\theta, \tag{7}$$

where $\partial\theta$ is change in water storage in the rooting zone during irrigation and can be calculated as

$$\partial\theta = \theta_f - \theta_i, \tag{8}$$

where θ_i is the TDR-measured initial soil volumetric water content obtained by averaging the previous three θ readings before the start of irrigation, and θ_f is the TDR-measured final volumetric water content of soil (average of three θ) 24 hours after the cessation of irrigation. The amount of leaching (LA) was determined as

$$\mathrm{LA} = \mathrm{TWA} - \mathrm{AWC}, \tag{9}$$

where TWA is the total water received by a crop as irrigation or rainfall. A constraint applied to (9) was that if TWA < AWC, then LA = 0. The TDR sensor at the 20 cm depth recorded changes in θ with irrigation, but almost no change in θ was recorded at 50 cm depth throughout the growing season. Hence, soil at or below 50 cm was considered at field capacity (FC) during the entire growing season in both fields. The θ at FC was determined by collecting gravimetric soil samples 24 h after the cessation of irrigation at both fields, from 0- to 110-cm depth. For the furrow-irrigated field, FC was 0.31 ± 0.02 cm^3 cm^{-3} for the 0- to 60-cm depth and 0.18 ± 0.03 cm^3 cm^{-3} for depths greater than 60 cm. For the drip-irrigated field, FC was 0.31 ± 0.04 cm^3 cm^{-3} for the 0- to 85-cm depth and 0.18 ± 0.05 cm^3 cm^{-3} for depths greater than 85 cm. The leaching depths in the upper 60 cm of the furrow-irrigated field and upper 85 cm of the drip-irrigated soil profile were calculated as

$$\text{Leaching Depth} = \frac{\mathrm{LA}}{0.31}. \tag{10}$$

Similarly, the leaching depths below 60 cm in the furrow-irrigated field and below 85 cm in the drip-irrigated field were calculated as

$$\text{Leaching Depth} = \frac{\mathrm{LA}}{0.18}. \tag{11}$$

The cumulative leaching depth below the rooting zone was obtained by adding leaching depths for each irrigation application. It was supposed that the first irrigation only saturated the upper 10 cm of the soil. A piston flow approach was adopted in both the fields to calculate the leaching depth of N fertilizer percolating to the deeper depths along with the irrigation water. The average NO_3-N and chloride concentrations for the months fertilizer applications were made and were used to calculate the LF, IE, and amount of NO_3-N loading below the rooting zone.

2.10. Statistical Analyses. As soil texture for the 0 to 50 cm soil profile also the rooting depth was similar in both fields, onion yield, leaching fractions, N use and application efficiencies in both fields were analyzed using the GLM (general linear model) procedure of SAS version 9.2 [32]. Statistical differences were evaluated at a probability level of $P \leq 0.05$.

3. Results and Discussion

3.1. Soil Properties. In the furrow-irrigated field, according to USDA soil classification, the top 60 cm of soil from

the surface was a sandy clay loam and was sand below 60 cm (Table 1). The average BD of 0 to 60 cm depths was 1.4 g cm^{-3}, and average K_s was 2.33 cm h^{-1}. The average soil pH and EC for the 0 to 60 cm depths were 7.4 and 2.0 dSm^{-1} and were 8.0 and 0.77 dSm^{-1} for depths greater than 60 cm, respectively. For the 0 to 60 cm depths, average soil OM, P, and K were 0.43%, 11.6 mg kg^{-1}, and 57.3 mg kg^{-1}and for depths greater than 60 cm, these were 0.02%, 2.2 mg kg^{-1}, and 29 mg kg^{-1}, respectively (Table 2).

In the drip-irrigated field, the top 60 cm soil profile was classified as a sandy clay loam, 60 to 85 cm depth as silt loam, and below 85 cm as sand (Table 1). The average BD for the 0 to 60 cm depth was 1.4 g cm^{-3}, and average K_s was 2.1 cm h^{-1}. The average soil pH and EC for the 0 to 60 cm depth were 7.6 and 2.0 dSm^{-1}, and for depths greater than 60 cm were 7.8 and 1.95 dSm^{-1}, respectively. The average soil OM, P, and K were 1.2%, 56.5 mg kg^{-1}, and 119.2 mg kg^{-1} for 0 to 60 cm depth and 0.34%, 4.9 mg kg^{-1}, and 62.5 mg kg^{-1} for depths greater than 60 cm, respectively (Table 2). Consequently, the mineralization rate below 60 cm could be considered as negligible.

The soil texture of both fields was similar except at the 60 to 85 cm depth, at which drip-irrigated field contained more fine-textured soil than the furrow-irrigated field. The average soil EC for the 0 to 60 cm depths was equal or below the threshold of 2 dS m^{-1} [33] in both the fields. Consequently, yield and evapotranspiration were not influenced by salinity stress. The OM contents of each field was low (OM < 1.8%); which is typical of the arid southern NM. The soil ESP and SAR values were well below the threshold levels of 13 and 15, respectively [33], and there were no sodicity problems in either field. Up to 60-cm depths, ESP, SAR, OM, P, and K contents were significantly higher ($P < 0.05$) in drip-irrigated field than in furrow-irrigated field.

3.2. Onion Rooting Zone Depth. The majority of the onion roots were found in the 0 to 20 cm soil depth, with a maximum rooting depth of 48 ± 2 cm during both the onion growing seasons. Hence, a maximum rooting depth of 50 cm was considered for both fields in estimating the leaching depth of NO_3-N and water. The maximum onion rooting depth reported in the literature ranges from 30 cm [31] to 45 cm [3].

3.3. Soil Nitrate-N Content. During growing season 1, average soil NO_3-N content of the plow layer (0–30 cm) increased from 18.0 mg kg^{-1} in December 2006 to 21.6 mg kg^{-1} in January 2007 (Figure 1), which was equivalent to 14.8 kg NO_3-N ha^{-1}. This increase could only be due to the mineralization of the sudan grass incorporated prior to planting of the onion crop, because no fertilizer was applied at this stage in the furrow-irrigated field. It has been reported that N mineralization sharply increases as the air temperature increases from 13°C to 22°C [34]. Air temperature in the experimental field sites was above 13°C for 21 days during December 2006, ranging from 13°C to 22°C, and for 15 days during January 2007, ranging from 13°C to 20°C. Thus, air temperature was favorable for mineralization.

The soil NO_3-N concentration increased to 26.9 mg kg^{-1} soil in the 0 to 50 cm layer during February 2007, after the first fertilizer application was made on February 23, 2007, four days before the soil sample was collected. The average soil NO_3-N concentration further increased to 43.1 mg kg^{-1} during March and 68.6 mg kg^{-1} during April in the 0 to 50 cm layer, as two fertilizer applications were made during March and three during April (Figure 1; Table 3). Bulb initiation started during early April 2007 and was completed by the end of June 2007. The NO_3-N concentration in the upper 0 to 50 cm soil layer (maximum root zone depth) then decreased in May and June, the months of high N uptake by onion plants [35].

Similarly, in growing season 2, average soil NO_3-N content (0–50 cm) increased from 8.8 mg kg^{-1} in September 2008 to 15.9 mg kg^{-1} in October 2008 that was due to the fertilizer application of 49.2 kg ha^{-1} in October 2008 (Figure 1). The average soil NO_3-N content decreased to 9.9 mg kg^{-1} in January 2009 due to winter irrigation leaching as no fertilizer was applied during November and December 2008 and January 2009. The fertilizer was applied at a rate of 49.2 kg N ha^{-1} per application during February (total of 49.2 kg N ha^{-1}), March (total of 98.4), and April, 2009 (total of 98.4 kg N ha^{-1}, Table 3). The average soil NO_3-N content increased until it reached 45.3 mg kg^{-1} during April, 2009 and then a decreasing trend was observed in soil NO_3-N content until harvesting of the crop. In general, soil NO_3-N concentration was higher in the rooting zone than below it throughout the sampling period during both the growing seasons (Figure 1). Theoretically, NO_3-N concentrations should be zero below the rooting zone, with all applied N taken up by the crop. The soil water NO_3-N concentration, calculated by dividing soil NO_3-N concentration by water content, was 105.2 mg L^{-1} during growing season 1 whereas it was 97.4 mg L^{-1} during growing season 2 below the crop rooting zone depth. This nitrogen can leach into the groundwater unless denitrification occurs within the capillary fringe zone just above the water table.

In the drip-irrigated field, the sequence of fertilizer applications was different and the number of fertilizer applications was also higher but the amount applied at each application was less for the drip- than the furrow-irrigated field. During growing season 1, average soil NO_3-N concentration increased from 11.8 mg kg^{-1} during December 2006 to 26.1 mg kg^{-1} during January 2007 in the plow layer (0–30 cm), amounting to 60.1 kg NO_3-N ha^{-1} (Figure 2). The 51.1 kg N ha^{-1} came from the fertilizer application in January, and the rest was likely from the mineralization of sudan grass. The NO_3-N concentration decreased for the 0 to 50 cm profile due to two rain events in February. The NO_3-N increased to 25.8 mg kg^{-1} in the 0 to 50 cm soil profile during March primarily due to two applications of fertilizer. No fertilizer was applied during April, and that resulted in an attendant decrease in the soil NO_3-N at almost all depths except 30 to 40 cm and 40 to 60 cm. Soil NO_3-N increased at all the depths within the 0 to 110 cm soil profile due to the

Table 1: Mean and standard errors for soil physical properties of onion fields under furrow- and drip-irrigation systems in NM in 2006.

Irrigation System	Depth cm	Sand %	Silt %	Clay %	Soil type	BD g/cm^3	K_s cm/hr
Furrow irrigated	0–10	56.5 ± 0.00	22.8 ± 0.35	20.7 ± 0.35	Sandy clay loam	1.3 ± 0.02	1.92 ± 0.20
	10–30	53.5 ± 0.35	25.6 ± 0.53	20.9 ± 0.18	Sandy clay loam	1.4 ± 0.04	2.02 ± 0.22
	30–40	52.3 ± 0.18	27.5 ± 0.18	20.2 ± 0.00	Sandy clay loam	1.4 ± 0.05	2.13 ± 0.34
	40–60	54.8 ± 0.71	25.2 ± 0.71	20.0 ± 0.00	Sandy clay loam	1.5 ± 0.13	3.23 ± 1.10
	60–85	90.3 ± 1.94	5.5 ± 1.59	4.2 ± 0.35	Sand	—	—
	85–110	95.8 ± 0.00	1.0 ± 0.00	3.2 ± 0.00	Sand	—	—
Drip irrigated	0–10	50.8 ± 0.35	27.0 ± 0.35	22.2 ± 0.00	Sandy clay loam	1.2 ± 0.03	1.52 ± 0.42
	10–30	49.8 ± 0.00	27.0 ± 0.00	23.2 ± 0.00	Sandy clay loam	1.4 ± 0.06	1.43 ± 0.20
	30–40	46.8 ± 1.77	34.0 ± 2.83	19.2 ± 1.06	Sandy clay loam	1.4 ± 0.13	1.34 ± 0.23
	40–60	55.3 ± 1.24	24.3 ± 0.35	20.4 ± 1.59	Sandy clay loam	1.4 ± 0.05	4.09 ± 0.9
	60–85	31.8 ± 4.24	55.0 ± 3.54	13.2 ± 0.71	Silt loam	—	—
	85–110	88.2 ± 1.13	7.8 ± 0.55	4.0 ± 0.22	Sand	—	—

BD: bulk density, K_s: saturated hydraulic conductivity, and —: value not determined.

Table 2: Soil chemical properties of two onion fields at the Plant Science Research Center (PSRC) and Fabian Garcia Research Center (FGRC) in NM in 2006.

Irrigation system	Depth cm	pH	EC_e dS/m	SAR	ESP %	OM %	P ppm	K ppm
Furrow irrigated	0–10	7.3	2.10	3.54	3.80	0.58	16.70	83.0
	10–30	7.3	1.81	1.97	1.60	0.61	17.40	61.0
	30–40	7.4	1.89	1.38	0.80	0.48	9.90	51.0
	40–60	7.5	2.19	1.56	1.00	0.04	2.30	34.0
	60–85	7.8	1.02	2.00	1.70	0.03	2.80	32.0
	85–110	8.2	0.51	2.52	2.40	0.00	1.50	26.0
Drip irrigated	0–10	7.7	2.31	3.52	3.80	1.44	62.50	137.0
	10–30	7.6	1.80	3.48	3.70	1.38	54.60	118.0
	30–40	7.5	2.33	3.88	4.30	1.67	88.90	138.0
	40–60	7.6	1.59	3.11	3.20	0.61	19.90	84.0
	60–85	7.7	1.85	3.14	3.30	0.43	7.20	79.00
	85–110	7.8	2.04	4.19	4.70	0.24	2.60	46.00

EC_e: electrical conductivity of saturated extract, SAR: sodium adsorption ratio, ESP: exchangeable sodium percentage, OM: organic matter, P: phosphorous, and K: potassium.

fertilizer application of May 2007, and not much change in soil NO_3-N was observed during June 2007.

A similar trend of soil NO_3-N content was also observed during growing season 2. The average soil NO_3-N content (0–50 cm) increased from 11.2 mg kg^{-1} in February 2008 to 19.8 mg kg^{-1} in March 2008 that was due to the two fertilizer applications (57.8 kg ha^{-1}) in March 2008. The average soil NO_3-N content further increased to 22.8 mg kg^{-1} during April 2008 that was in response to another two fertilizer applications (68 kg ha^{-1}) in April, 2008. The average soil NO_3-N content further increased to 27.3 mg kg^{-1} in May 2008 and 32 mg kg^{-1} in June 2008 in response to the two fertilizer applications in May (64.4 kg ha^{-1}) and another two in June (96.2 kg ha^{-1}, Table 3). The soil NO_3-N content decreased to 15.5 mg kg^{-1} in August 2008 due to plant uptake as well as N leaching as no fertilizer was applied during July and August 2008. An average soil water NO_3-N concentration of 132.5 mg L^{-1} and 130.3 mg L^{-1} was estimated in the rooting zone of the crop during growing seasons 1 and 2, respectively. Similarly, soil water NO_3-N concentration of 66.8 mg L^{-1} and 65.2 mg L^{-1} was estimated below rooting depth of the crop during growing seasons 1 and 2, respectively (Figure 2). Similar to the furrow-irrigated field, the NO_3-N concentrations below rooting depths were much higher than the NO_3-N levels of 10 mg L^{-1} recommended by the U.S. Environmental Protection Agency's drinking water standard [36].

3.4. Irrigation Water Front Depths. In the furrow-irrigated field, total water wetting front depth for the entire growing season was estimated at 213 cm during growing season 1 and 196 cm during growing season 2 (Table 4) from the

Table 3: Type, date, and amount of N fertilizer applied in furrow- and drip-irrigated onion fields in NM in 2006–2009.

Field	Irrigation system	Fertilizer type	Date of application	Amount of application (kg N ha^{-1})
PSRC	Furrow irrigated	URAN (26-0-0-6)	2/23/2007	49.2
			3/16/2007	49.2
			3/27/2007	49.2
			4/05/2007	49.2
			4/13/2007	49.2
			4/20/2007	49.2
			10/10/2008	49.2
			2/24/2009	49.2
			3/12/2009	49.2
			3/24/2009	49.2
			4/14/2009	49.2
			4/26/2009	49.2
FGRC	Drip irrigated	URAN (26-0-0-6)	11/9/2006	17.9
			11/18/2006	23.9
			12/16/2006	23.9
			1/13/2007	51.1
			2/10/2007	36.5
			3/27/2007	31.8
			3/30/2007	47.8
			5/05/2007	59.0
			3/17/2008	24.6
			3/25/2008	33.2
			4/14/2008	46.0
			4/28/2008	22.0
			5/07/2008	36.0
			5/22/2008	28.4
			6/18/2008	58.0
			6/30/2008	38.2

PSRC: Plant Science Research Center, FGRC: Fabian Garcia Research Center;
URAN: Urea ammonium nitrate.

soil surface, with average water wetting front velocity of 0.89 cm day^{-1} during growing season 1 and 0.78 cm day^{-1} in growing season 2. The NO_3-N front depth was estimated at 149 cm, with an average NO_3-N front velocity of 1.18 cm day^{-1} in growing season 1 whereas NO_3-N front depth was 196 cm, with average NO_3-N front velocity of 0.82 cm day^{-1} in growing season 2. Although NO_3-N moves with water, the water wetting front depth was greater than the N front depth during both the growing seasons because six water applications were made beginning November 2006, before the first N fertilizer was applied during February 2007 in growing season 1. Similarly, water application was started on September, 2008 and first N fertilizer application was made on October, 2008 in growing season 2 and piston flow was the only mechanism considered for solute transport (Table 4).

In the drip-irrigated field, the depth of the water wetting front was estimated at 147 cm (Tables 5(a) and 5(b)), with average water wetting front velocity of 0.58 cm day^{-1} in growing season 1 whereas the estimated depth of water wetting front was 105 cm (Tables 5(a) and 5(b)), with average water wetting front velocity of 0.67 cm day^{-1} in growing season 2. The NO_3-N front depth was estimated at 86 cm, with average NO_3-N front velocity of 0.41 cm day^{-1} in growing season 1 whereas estimated depth of NO_3-N front was 87 cm, with average NO_3-N front velocity of 0.63 cm day^{-1} in growing season 2. Similar to the furrow-irrigated field, the water wetting front was higher than the NO_3-N front depth because the water application was started earlier than N fertilizer application.

The assumption that wetting and NO_3-N fronts move at the same velocity may not be valid under field conditions. The NO_3-N flow velocity can be lower than the wetting front velocity if NO_3-N adsorption was taking place during the transport through the soil profile. In the case of anion exclusion, the NO_3-N flow velocity could be higher than the wetting front velocity, and deeper leaching of NO_3-N can be observed under such conditions [37].

Table 4: Date of irrigation, total amount of water applied (TWA), initial (θ_i) and final (θ_f) volumetric water content, rooting depth (RD), available water content (AWC), amount of leaching (LA), depth of leaching (LAD), cumulative leaching below the rooting zone depth (CLRBRZD), and depth of water front (DWF) during two onion growing seasons (2006–09) in a furrow-irrigated onion field in NM.

Date of Irrigation	TWA cm	θ_i cm^3 cm^{-3}	θ_f cm^3 cm^{-3}	RD cm	AWC cm	LA cm	LAD cm	CLRBRZD cm	DWF cm
11/2/2006	7.76	0.11	0.31	10.0	Bring root zone to field capacity				
11/17/2006	4.39	0.23	0.30	10.0	0.69	3.71	12.0	12.0	22.0
11/29/2006	4.39	0.24	0.30	10.0	0.67	3.72	12.0	24.0	34.0
12/15/2007	4.39	0.24	0.31	10.0	0.71	3.69	11.9	35.9	46
1/8/2007	4.09	0.23	0.31	10.0	0.77	3.32	10.7	46.6	57
2/9/2007	3.85	0.23	0.31	10.0	0.75	3.10	17.2	63.8	74
2/23/2007	3.55	0.24	0.30	10.0	0.57	2.98	16.5	80.4	90
3/6/2007	4.20	0.21	0.32	10.0	1.05	3.15	17.5	97.9	108
3/16/2007	4.14	0.22	0.32	12.7	1.28	2.86	15.9	113.7	124
3/27/2007	5.12	0.20	0.31	17.8	1.96	3.16	17.6	131.3	141
4/5/2007	4.70	0.22	0.32	22.9	2.29	2.41	13.4	144.7	155
4/13/2007	4.14	0.20	0.32	27.9	3.38	0.76	4.2	148.9	159
4/20/2007	4.91	0.21	0.31	30.5	3.05	1.86	10.4	159.3	169
5/12/2007	5.04	0.21	0.32	35.6	3.91	1.13	6.3	165.6	176
5/25/2007	4.45	0.21	0.31	38.1	3.81	0.64	3.5	169.1	179
6/1/2007	6.12	0.20	0.31	45.7	5.02	1.10	6.1	175.2	185
6/7/2007	6.10	0.21	0.31	45.7	4.63	1.47	8.2	183.3	193
6/19/2007	6.80	0.20	0.31	45.7	5.03	1.77	9.8	193.2	203
6/29/2007	6.78	0.19	0.30	45.7	4.97	1.81	10.1	203.2	213
2008-2009									
9/25/2008	5.8	0.12	0.33	10.0	2.10	3.66	11.8		10
10/10/2008	4.20	0.22	0.31	10.0	0.89	3.26	10.5	10.5	21
10/24/2008	3.24	0.24	0.29	10.0	0.57	2.67	8.6	19.1	29
11/14/2008	4.59	0.23	0.30	10.0	0.71	3.88	12.5	31.6	42
11/21/2008	4.15	0.24	0.32	10.0	0.77	3.38	10.9	42.5	53
12/5/2008	3.24	0.22	0.30	10.0	0.75	2.49	8.0	50.6	61
12/18/2008	3.31	0.23	0.30	10.0	0.67	2.64	8.5	59.1	69
1/9/2009	3.20	0.22	0.31	10.0	0.85	2.35	13.1	72.1	82
2/11/2009	3.12	0.22	0.30	12.7	1.03	2.09	11.6	83.8	94
3/6/2009	4.58	0.21	0.31	17.8	1.78	2.80	15.6	99.3	109
3/20/2009	4.66	0.21	0.31	22.9	2.29	2.29	12.7	112.1	122
3/27/2009	4.01	0.23	0.32	27.9	2.54	2.12	11.8	123.8	134
4/3/2009	4.81	0.22	0.32	30.5	3.05	0.96	5.3	129.2	139
4/9/2009	5.49	0.22	0.32	35.6	3.56	1.25	7.0	136.1	146
4/17/2009	5.91	0.20	0.30	38.1	3.81	1.68	9.3	145.5	155
4/24/2009	6.20	0.21	0.32	45.7	5.02	0.88	4.9	150.4	160
5/1/2009	6.78	0.22	0.33	45.7	5.08	1.12	6.2	156.6	167
5/8/2009	5.62	0.23	0.31	45.7	3.66	3.12	17.3	173.9	184
5/15/2009	6.53	0.22	0.33	45.7	5.03	0.59	3.3	177.2	187
5/27/2009	5.44	0.19	0.31	45.7	5.49	1.04	5.8	183.0	193
6/4/2009	5.3	0.20	0.31	45.7	4.97	0.47	2.6	185.6	196

3.5. Plant Nutrient Content and Onion Yield. In the furrow-irrigated field, total N concentration in plant tissue was 1.68 ± 0.003% of the total dry onion biomass in growing season 1 whereas it was 1.7 ± 0.004% in growing season 2. However, in the drip-irrigated field, total N concentration in plant tissue was 1.65 ± 0.003% and 1.63 ± 0.003% of the total biomass of dry onion in growing seasons 1 and 2, respectively. The chloride concentration in the plant tissue

Table 5: (a) Date of irrigation, total amount of water applied (TWA), initial (θ_i) and final (θ_f) volumetric water content, rooting depth (RD), available water content (AWC), amount of leaching (LA), depth of leaching (LAD), cumulative leaching below root zone depth (CLRBRZD), and depth of water front (DWF) during onion growing season 1 (2006-07) in a drip-irrigated onion field in NM. (b) Date of irrigation, total amount of water applied (TWA), initial (θ_i) and final (θ_f) volumetric water content, rooting depth (RD), available water content (AWC), amount of leaching (LA), depth of leaching (LAD), cumulative leaching below root zone depth (CLRBRZD), and depth of water front (DWF) during onion growing season 2 (2008) in a drip-irrigated onion field in NM.

(a)

Date of irrigation	TWA cm	θ_i cm^3 cm^{-3}	θ_f cm^3 cm^{-3}	RD cm	AWC cm	LA cm	LAD cm	CLRBRZD cm	DWF cm
9/29/2006	3.43	0.16	0.31	10.0	Bring root zone to field capacity				10
9/30/2006	4.20	0.29	0.32	10.0	0.35	3.85	12.4	12.4	22
10/3/2006	4.69	0.25	0.32	10.0	0.70	3.99	12.9	25.3	35
10/5/2006	3.80	0.28	0.31	10.0	0.35	3.45	11.1	36.4	46
10/6/2006	2.09	0.29	0.32	10.0	0.34	1.76	5.7	42.1	52
10/22/2006	4.61	0.23	0.32	10.0	0.87	3.74	12.1	54.2	64
10/30/2006	2.09	0.25	0.31	10.0	0.61	1.48	4.8	58.9	69
10/31/2006	0.83	0.29	0.30	10.0	0.17	0.66	2.1	61.0	71
11/9/2006	1.95	0.23	0.31	10.0	0.78	1.17	3.8	64.8	75
11/18/2006	1.95	0.24	0.32	10.0	0.78	1.17	3.8	68.6	79
11/25/2006	2.04	0.25	0.32	10.0	0.70	1.34	4.3	72.9	83
12/3/2006	1.02	0.26	0.30	10.0	0.36	0.66	2.1	75.0	85
12/8/2006	1.01	0.26	0.30	10.0	0.36	0.65	3.6	78.6	89
12/16/2006	1.77	0.27	0.30	10.0	0.26	1.51	8.4	87.0	97
12/24/2006	0.80	0.27	0.30	10.0	0.35	0.45	2.5	89.5	100
1/13/2007	1.13	0.26	0.32	10.0	0.61	0.52	2.9	92.4	103
2/4/2007	1.01	0.26	0.31	10.0	0.52	0.49	2.7	95.1	105
2/10/2007	1.65	0.25	0.32	10.0	0.70	0.95	5.3	100.4	111
2/20/2007	0.80	0.26	0.30	10.0	0.35	0.45	2.5	102.9	113
3/2/2007	1.64	0.25	0.32	12.7	1.01	0.63	3.5	106.4	117
3/10/2007	1.43	0.25	0.29	17.8	0.62	0.81	4.5	110.9	121
3/14/2007	1.29	0.26	0.30	22.9	0.80	0.49	2.7	113.7	124
3/19/2007	1.32	0.27	0.31	27.9	1.15	0.17	1.0	114.7	125
3/23/2007	1.53	0.27	0.30	30.5	0.80	0.73	4.1	118.7	129
3/27/2007	1.87	0.27	0.30	35.6	1.24	0.63	3.5	122.2	132
3/30/2007	1.35	0.28	0.31	38.1	1.50	0.00	0.0	122.2	132
4/4/2007	2.28	0.27	0.31	45.7	2.24	0.04	0.2	122.4	133
4/8/2007	0.99	0.27	0.29	45.7	1.12	0.00	0.0	122.4	133
4/11/2007	2.02	0.27	0.31	45.7	1.99	0.03	0.2	122.6	133
4/16/2007	2.54	0.27	0.31	45.7	2.25	0.29	1.6	124.2	134
4/23/2007	1.89	0.26	0.30	45.7	1.86	0.03	0.2	124.4	134
4/24/2007	1.35	0.30	0.32	45.7	1.14	0.21	1.2	125.6	136
4/26/2007	1.94	0.29	0.32	45.7	1.81	0.13	0.7	126.3	136
5/5/2007	3.22	0.25	0.31	45.7	3.09	0.13	0.7	127.1	137
5/11/2007	1.31	0.27	0.30	45.7	1.19	0.12	0.7	127.7	138
5/14/2007	1.49	0.26	0.29	45.7	1.19	0.30	1.7	129.4	139
5/21/2007	2.70	0.25	0.30	45.7	2.39	0.31	1.7	131.1	141
5/24/2007	0.30	0.27	0.30	45.7	1.25	0.00	0.0	131.1	141
5/25/2007	2.05	0.28	0.31	45.7	1.59	0.46	2.5	133.7	144
5/31/2007	2.02	0.25	0.30	45.7	1.99	0.03	0.2	133.8	144

(a) Continued.

Date of irrigation	TWA cm	θ_i cm^3 cm^{-3}	θ_f cm^3 cm^{-3}	RD cm	AWC cm	LA cm	LAD cm	CLRBRZD cm	DWF cm
6/1/2007	2.20	0.28	0.31	45.7	1.59	0.61	3.4	137.2	147
6/7/2007	1.16	0.25	0.30	45.7	2.11	0.00	0.0	137.2	147

(b)

Date of irrigation	TWA cm	θ_i cm^3 cm^{-3}	θ_f cm^3 cm^{-3}	RD cm	AWC cm	LA cm	LAD cm	CLRBRZD cm	DWF cm
3/1/2008	3.43	0.16	0.31	10.0	Bring root zone to field capacity				10
3/3/2008	2.15	0.27	0.32	10.0	0.55	1.60	5.2	5.2	15
3/6/2008	3.06	0.26	0.33	10.0	0.70	2.36	7.6	12.8	23
3/12/2008	1.90	0.26	0.31	10.0	0.55	1.35	4.4	17.2	27
3/17/2008	1.79	0.28	0.32	10.0	0.44	1.36	4.4	21.6	32
3/20/2008	1.54	0.25	0.31	10.0	0.57	0.97	3.1	24.7	35
3/25/2008	2.19	0.24	0.30	10.0	0.61	1.58	5.1	29.8	40
3/31/2008	1.02	0.27	0.31	10.0	0.47	0.55	1.8	31.6	42
4/3/2008	1.72	0.24	0.31	10.0	0.68	1.04	3.3	34.9	45
4/7/2008	2.05	0.25	0.33	10.0	0.78	1.27	4.1	39.0	49
4/9/2008	1.62	0.26	0.32	10.0	0.60	1.02	3.3	42.3	52
4/14/2008	1.13	0.24	0.30	10.0	0.56	0.57	1.8	44.1	54
4/17/2008	0.91	0.27	0.29	10.0	0.23	0.68	2.2	46.3	56
4/22/2008	1.57	0.26	0.31	10.0	0.46	1.11	3.6	49.9	60
4/25/2008	1.15	0.26	0.30	10.0	0.45	0.70	2.3	52.2	62
4/28/2008	1.33	0.27	0.31	10.0	0.41	0.92	3.0	55.1	65
5/1/2008	1.21	0.27	0.32	10.0	0.52	0.69	2.2	57.3	67
5/7/2008	1.25	0.25	0.31	10.0	0.60	0.65	2.1	59.5	69
5/12/2008	1.20	0.25	0.31	10.0	0.55	0.65	2.1	61.6	72
5/15/2008	1.21	0.27	0.32	12.7	0.76	0.45	1.5	63.0	73
5/20/2008	1.23	0.25	0.30	17.8	0.80	0.43	1.4	64.4	74
5/22/2008	1.37	0.26	0.31	22.9	1.02	0.34	1.1	65.5	76
5/26/2008	1.41	0.26	0.31	27.9	1.43	0.00	0.0	65.5	76
5/30/2008	1.13	0.27	0.30	30.5	0.80	0.33	1.1	66.6	77
6/3/2008	1.32	0.26	0.30	35.6	1.59	0.00	0.0	66.6	77
6/6/2008	2.21	0.27	0.32	38.1	2.27	0.00	0.0	66.6	77
6/10/2008	2.34	0.26	0.33	45.7	3.61	0.00	0.0	66.6	77
6/14/2008	2.43	0.28	0.31	45.7	1.57	0.86	2.8	69.4	79
6/18/2008	2.06	0.26	0.32	45.7	2.90	0.00	0.0	69.4	79
6/23/2008	3.14	0.27	0.31	45.7	2.25	0.89	4.9	74.3	84
6/27/2008	1.71	0.26	0.31	45.7	2.32	0.00	0.0	74.3	84
6/30/2008	1.22	0.29	0.32	45.7	1.60	0.00	0.0	74.3	84
7/3/2008	2.45	0.28	0.32	45.7	2.26	0.19	1.0	75.4	85
7/8/2008	2.92	0.25	0.31	45.7	3.09	0.00	0.0	75.4	85
7/14/2008	1.71	0.25	0.30	45.7	2.11	0.00	0.0	75.4	85
7/18/2008	1.63	0.26	0.29	45.7	1.19	0.44	2.4	77.8	88
7/22/2008	2.52	0.26	0.30	45.7	1.93	0.59	3.3	81.1	91
7/25/2008	1.72	0.27	0.29	45.7	1.12	0.60	3.4	84.4	94
7/29/2008	2.55	0.27	0.31	45.7	2.05	0.50	2.8	87.2	97
8/1/2008	2.41	0.27	0.30	45.7	1.07	1.34	7.4	94.6	105

Table 6: Mean and standard errors for NO_3-N, chloride, and water balance during two onion growing seasons (2006–09) in furrow-irrigated onion field in NM.

		Furrow-irrigated field (2006-07)			Furrow-irrigated field (2008-09)		
		Nitrate-N (kg ha^{-1})	Chloride (kg ha^{-1})	Water cm	Nitrate-N (kg ha^{-1})	Chloride (kg ha^{-1})	Water cm
1	Total applied	295	522	95	295	530	100
2	Deep percolated (>50 cm depth)	150 ± 2.2	331 ± 2.7	19 ± 0.85	145 ± 0.5	306 ± 3.07	22.3 ± 1.0
3	Crop uptake (*E_t)	104 ± 0.21	78 ± 0.20	*61 ± 0.0	106 ± 0.1	79 ± 0.52	*62 ± 0.0
4	Before planting (0–50 cm depth)	94 ± 1.7	105 ± 4.7	3 ± 0.24	83.7 ± 1.9	87.2 ± 1.9	3 ± 0.15
5	After harvest (0–50 cm depth)	134 ± 2.9	208 ± 4.7	10 ± 0.53	129.3 ± 1.2	224.2 ± 1.9	11 ± 0.29
6	Storage (0–50 cm depth)	40 ± 1.2	102 ± 8.2	7 ± 0.35	45.6 ± 2.6	137 ± 0.31	8 ± 0.43
7	Output (total loss, uptake, and change; Rows 2 + 3 + 6)	294 ± 1.8	512 ± 6.9	87 ± 1.2	297 ± 3.1	522 ± 2.6	95 ± 0.62
8	Mass balance error (Row 1–Row 7)	1 ± 1.8	10 ± 6.9	8 ± 1.2	−2 ± 3.1	8 ± 2.6	8 ± 0.62

*: E_t.

Table 7: Mean and standard errors for NO_3-N, chloride, and water balance during two onion growing seasons (2006–08) in drip-irrigated onion field in NM.

		Drip-irrigated field (2006-07)			Drip-irrigated field (2008)		
		Nitrate-N (kg ha^{-1})	Chloride (kg ha^{-1})	Water cm	Nitrate-N (kg ha^{-1})	Chloride (kg ha^{-1})	Water cm
1	Total applied	292	486	81	286	396	72
2	Deep percolated (>50 cm depth)	79 ± 6.9	298 ± 11.3	14 ± 1.77	76 ± 0.27	245 ± 5.6	12.5 ± 0.8
3	Crop uptake (E_t*)	112 ± 0.20	86 ± 0.21	56 ± 0.0*	111 ± 0.27	84 ± 0.32	55 ± 0.0*
4	Before planting (0–50 cm depth)	69 ± 3.5	86 ± 12.7	3 ± 0.18	65 ± 2.42	78 ± 4.3	4 ± 0.3
5	After harvest (0-50 cm depth)	159 ± 5.5	180 ± 13.9	9 ± 0.29	156 ± 3.95	139 ± 7.3	8 ± 0.3
6	Storage (0–50 cm depth)	90 ± 2.0	94 ± 10.0	6 ± 0.37	91 ± 2.38	61 ± 3.1	4 ± 0.4
7	Output (total loss, uptake, and change; Rows 2 + 3 + 6)	281 ± 3.8	477 ± 9.7	76 ± 1.72	278 ± 2.12	390 ± 8.1	71.5 ± 0.5
8	Mass balance error (Row 1–Row 7)	11 ± 3.8	9 ± 9.7	5 ± 1.72	8 ± 2.12	6 ± 8.1	0.5 ± 0.5

*: E_t.

was 1.26 ± 0.002% and 1.25 ± 0.002% of the total biomass of dry onion for furrow-irrigated field in growing seasons 1 and 2, respectively, and 1.24 ± 0.002% in growing seasons 1 and 2 for drip-irrigated field. Since amount of chloride uptake by plants constitutes a very small proportion of the total chloride flux, chloride uptake makes a little difference (<3%) in estimating the irrigation efficiencies [16].

In the furrow-irrigated field, onion yield on a wet basis was 45,120 kg ha^{-1} in growing season 1 and 45,420 kg ha^{-1} in growing season 2 with a moisture content of 90%. The total dry onion biomass yield was 6210 kg ha^{-1} and 6251 kg ha^{-1} in growing seasons 1 and 2, respectively, in the furrow-irrigated field.

The onion yield on the wet basis was 50,980 kg ha^{-1} and 50,840 kg ha^{-1} in growing seasons 1 and 2, respectively, in drip-irrigated field. These yields were significantly higher ($P < 0.01$) than those from the furrow-irrigated field during both the seasons. The total dry onion biomass yield was 6840 kg ha^{-1} and 6800 kg ha^{-1} in growing seasons 1 and 2, respectively. The yields were not significantly different ($P > 0.05$) between the two growing seasons for either furrow- or drip-irrigated fields.

3.6. Nitrate-N and Chloride Ratio. In the furrow-irrigated field, within the rooting zone, the NO_3-N/Cl ratio was variable. This could be due to the mineralization of sudan grass and/or N uptake by the onion crop (Figure 3). Below the crop rooting zone, where no N uptake occurs, the average NO_3-N/Cl ratio was similar, and averages of 0.61, 0.62, 0.61, and 0.62 were estimated during growing season 1 and 0.54, 0.53, 0.54, and 0.54 during growing season 2 at 85-, 110-, 150- and 200-cm depths, respectively (Figure 3). These values showed that NO_3-N distribution and LF were uniform throughout the growing seasons. They also showed that excess N was applied to meet the plant N needs throughout the growing seasons.

In the drip-irrigated field, the NO_3-N/Cl ratio in the entire soil profile was variable (Figure 3). The average NO_3-N/Cl ratio below rooting zone was 0.25, 0.26, 0.24, and 0.25 during growing season 1 and 0.29, 0.29, 0.28, and 0.29 during growing season 2 at 85-, 110-, 150-, and 200-cm depths, respectively. These values again showed excess but uniform NO_3-N distribution and LF during both the growing seasons in drip-irrigated field.

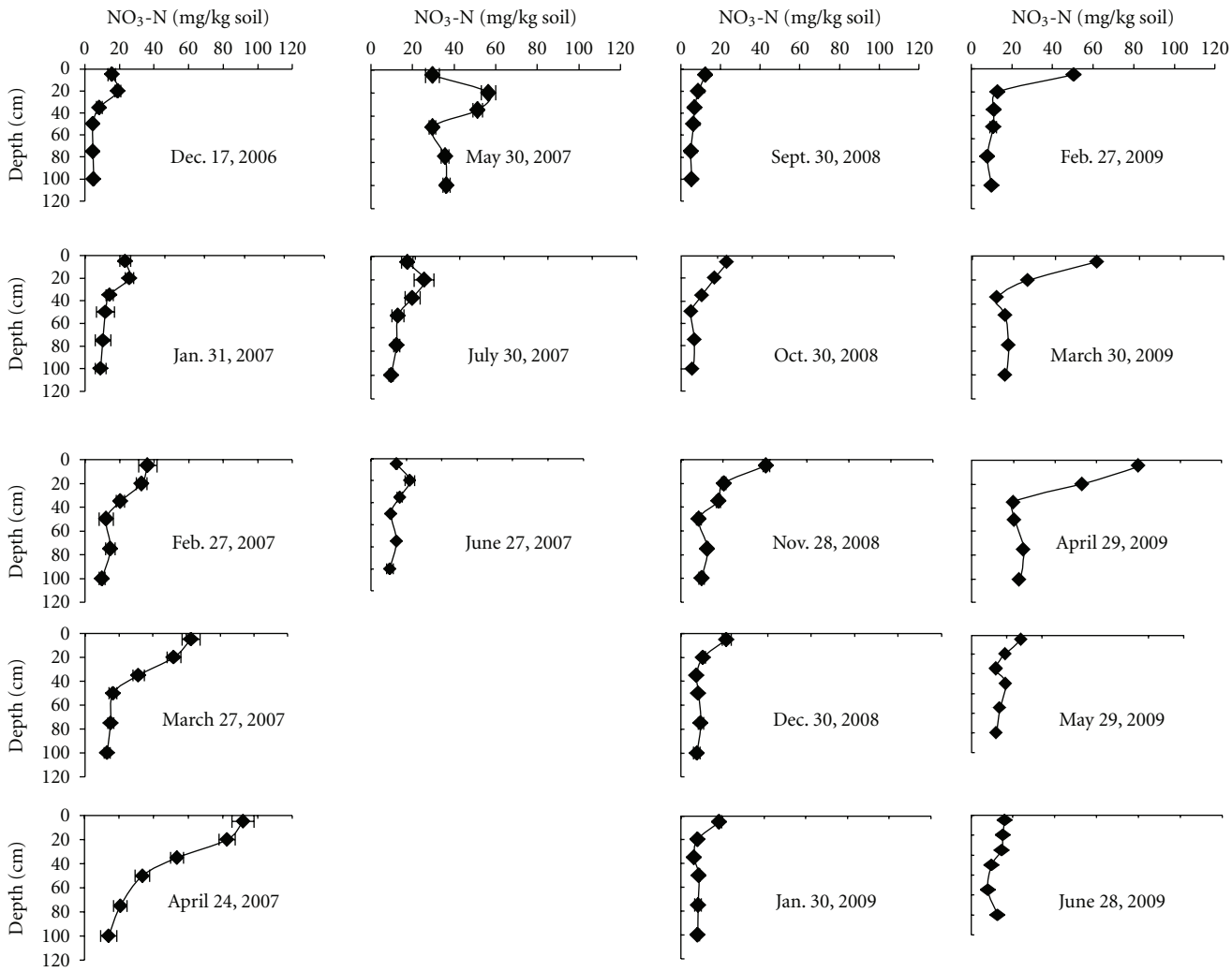


Figure 1: Monthly NO_3-N concentration (mg kg^{-1} soil) in the 0–110 cm soil profile during two onion growing seasons (2006–09) in furrow-irrigated onion field in NM. The horizontal bars represent standard errors of the mean. Each data point is the mean of three replicate soil samples.

3.7. Nitrate-N Loading. The Nitrate-N and chloride ratio was used in (4) to calculate the amount of NO_3-N percolated below the rooting zone of onion, and an averaged accumulated NO_3-N loading of 150 ± 2.2 kg ha^{-1} and 145 ± 0.5 kg ha^{-1} was obtained in growing seasons 1 and 2, respectively, for soil depths below the rooting zone of furrow-irrigated onion. The NO_3-N loading was significantly higher ($P < 0.01$) during growing season 1 than growing season 2 because all fertilizer applications were made during February, March, and April in the growing season 1 whereas fertilizer applications were spread over four months (October, February, March, and April) during growing season 2. The N front leached to a maximum depth of 149 cm in growing season 1 indicating that the NO_3-N below 149 cm depth was from the previous year. As same crop rotation was practiced in this field for many years and also the inputs such as water and N fertilizer were nearly the same during each growing season, therefore, this method can be used to estimate N leaching below root zone during these past years.

The average accumulated NO_3-N loading estimated below the rooting zone of onion was 79 ± 6.9 kg ha^{-1} and 76 ± 0.3 kg ha^{-1} during growing seasons 1 and 2, respectively, for the drip-irrigated field. Similar to the furrow-irrigation system, the NO_3-N concentration in the soil water below 86 cm depth represented the NO_3-N concentration from the previous year onion crop in the drip irrigation system. Almost similar amount of fertilizer was applied during both the growing seasons in furrow- and drip-irrigated fields but still 47% less NO_3-N loading during growing season 1 and 47.5% less during growing season 2 was recorded in drip- than in the furrow-irrigated field.

3.8. Nitrogen and Water Balance. The water balance presented an unaccounted amount of 8 ± 1.2 cm and 8 ± 0.6 cm of water during growing seasons 1 and 2, respectively, for of the total water received in the furrow-irrigated field (Table 6). Similarly, water balance in drip-irrigated field also showed

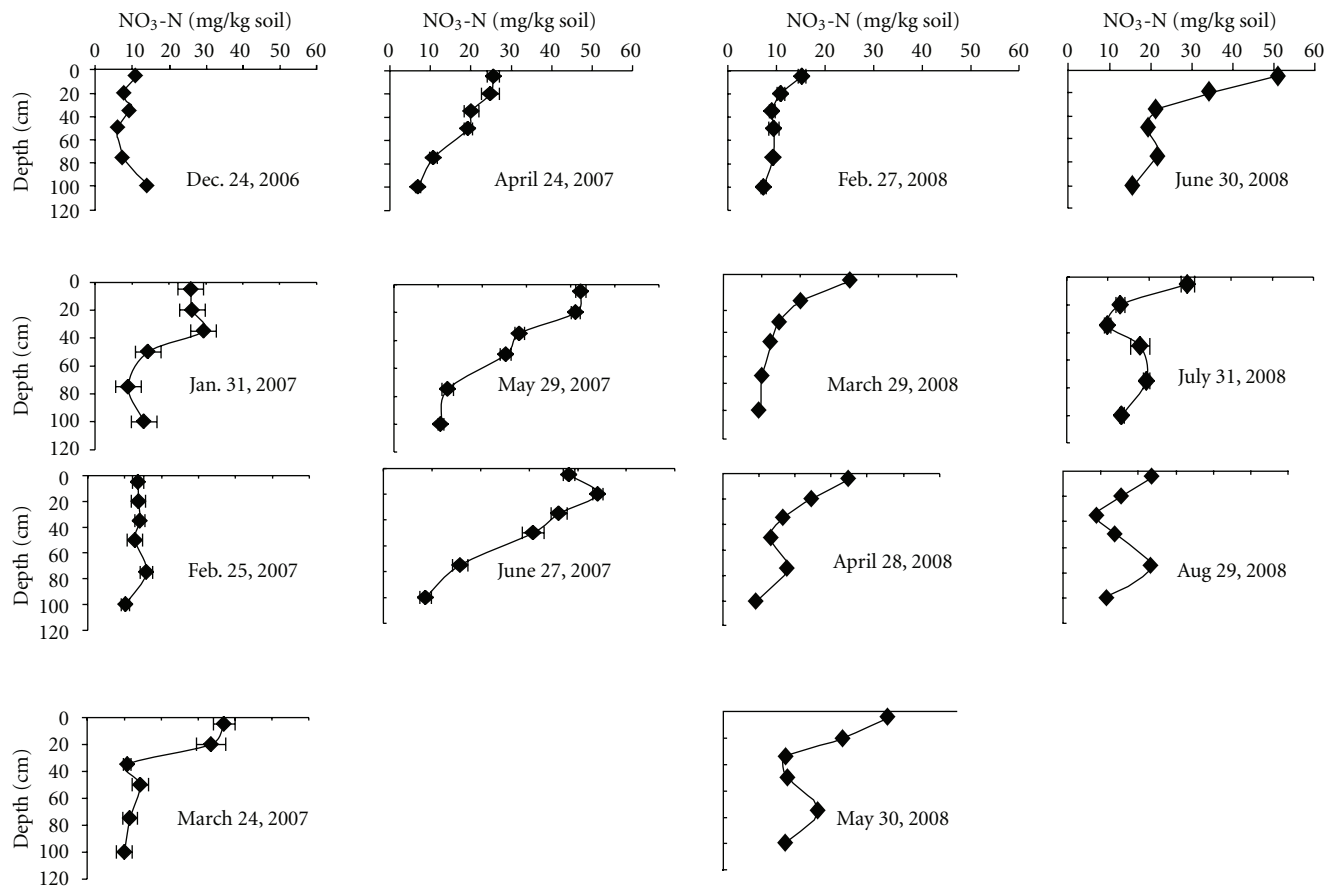


Figure 2: Monthly NO_3-N concentration (mg kg^{-1} soil) in the 0–110 cm soil profile during two onion growing seasons (2006–08) in drip-irrigated onion field in NM. The horizontal bars represent standard errors of the mean. Each data point is the mean of three replicate soil samples.

an unaccounted amount of 5 ± 1.72 cm and 0.5 ± 0.5 cm during growing seasons 1 and 2, respectively (Table 7). Water application efficiency was 72 ± 0.35% and 70 ± 0.43% during growing seasons 1 and 2, respectively, for the furrow-irrigated field and 77 ± 0.37% during growing season 1 and 82 ± 0.40% during growing season 2 for the drip-irrigated field. Water application efficiency was significantly higher ($P < 0.05$) in drip-irrigated field than in furrow-irrigated field during both the growing seasons. A higher amount of water and less frequent irrigations were applied to the furrow-irrigated field than the drip-irrigated field in this study. The chloride balance error in the drip-irrigated field could also be due to soil sampling errors, as the water flow is not one dimensional in drip irrigation system.

In the furrow-irrigated field, total N output—N loss plus N uptake plus storage of NO_3-N in soil profile—from the soil profile during the entire growing season 1 was 294 ± 1.8 kg N ha^{-1} and was 297 ± 3.1 kg N ha^{-1} during growing season 2 against the total input of 295 kg N ha^{-1} during both the growing seasons, in the form of URAN fertilizer (Table 6).

Nitrogen application efficiency was 35 ± 0.21% and 36 ± 0.1% during growing seasons 1 and 2, respectively, whereas N use efficiency was 26.7 ± 0.06% during growing season 1 and 28 ± 0.05% during growing seasons 2. Nitrogen application and use efficiencies were significantly higher ($P < 0.05$) in growing season 2 than in growing season 1, respectively. If the antecedent N level in the soil is sufficient for plant growth, N use efficiency can be increased by reducing the amount of fertilizer applied. N use efficiencies obtained in this study were greater (by about 15%) than those reported by [9] and less (30%) than those reported by [10] for onion under a furrow-irrigation system.

In the drip-irrigated field, total N output during the entire growing season 1 was 281 ± 3.8 kg N ha^{-1} and 278 ± 2.1 kg N ha^{-1} during growing season 2 against the total input of 292 kg N ha^{-1} and 286 kg N ha^{-1} during growing seasons 1 and 2, respectively, in the form of URAN fertilizer (Table 7). The total N output can be smaller due to mineralization of sudan grass that might have taken place in the rooting zone during the growing seasons. N use efficiency was 31 ± 0.25% and 32 ± 0.21% during growing seasons 1 and 2, respectively, whereas NAE was 38 ± 0.20% during growing season 1 and 39 ± 0.18% during growing season 2 in the drip-irrigated field. The unaccounted N in the NO_3-N balances might be due to denitrification taking place in the rooting zone or to the unaccounted N present in onion foliage at the time of harvesting. Nitrogen application and use efficiencies were

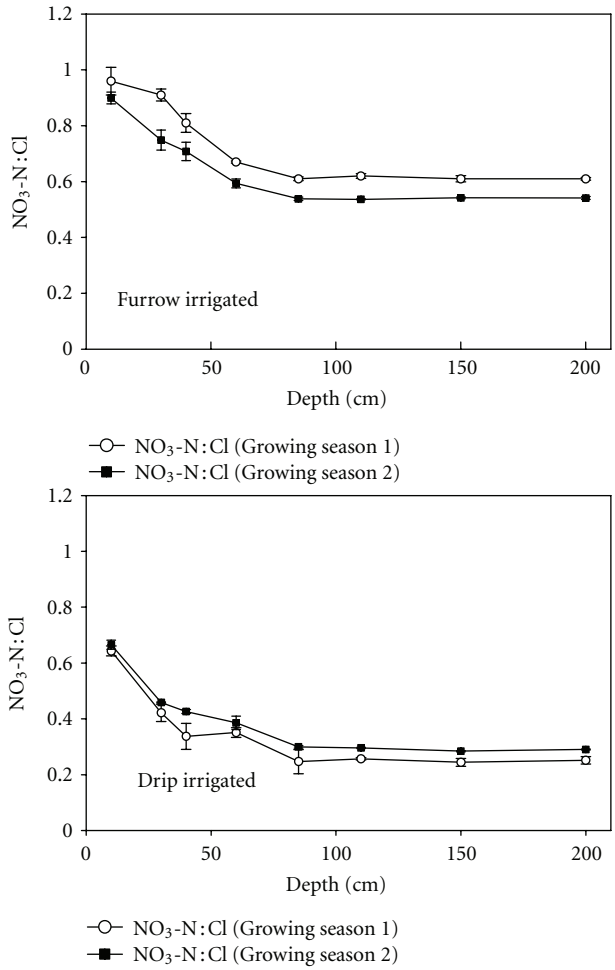


Figure 3: NO_3-N/Cl ratio at a 0–200 cm soil depth during two onion growing seasons in furrow- (2006–09) and drip-irrigated (2006–08) onion fields in NM. The vertical bars represent standard errors of the mean. Each data point is the mean of three replicate samples.

significantly higher ($P < 0.05$) in the drip-irrigated field than in the furrow-irrigated field during both the growing seasons.

In this study, N mineralization, denitrification N loss and N content of the onion foliage at harvesting were not determined and hence were not included in the N balance calculations for both fields. A search in the literature revealed that a total denitrification N loss of 51.2 kg N was reported for the agricultural fields, with a total fertilizer application of 335 kg N ha^{-1} [38], whereas it varied from 27 to 49 kg N ha^{-1} with the total fertilizer application ranging from 225 to 335 kg N ha^{-1} [39]. Literature searches did not yield any information on the N mineralization of sudan grass for southern New Mexico. Plow-down alfalfa was reported to contribute 35–125 kg N ha^{-1} yr^{-1} in the soil through mineralization [40, 41]. Sullivan [35] found 10 to 20% of the total onion N uptake to be present in onion foliage at the time of harvesting. Looking at the above numbers, it seemed that all the three factors of N—mineralized N, denitrified N, and N in the onion foliage—together could account for the N mass balance error obtained in the current study.

3.9. Leaching. Using (3), an LF of 0.20 ± 0.006 or IE (1 − LF) of 80 ± 0.60% during growing season 1 and an LF of 0.22 ± 0.004 or IE (1 − LF) of 78 ± 0.40% during growing season 2 were obtained for the furrow-irrigated field (Table 6). The LF during growing season 2 was significantly higher ($P < 0.05$) than the LF during growing season 1. An LF of 0.28 during growing season 1 and 0.30 during growing season 2 was estimated using water balance method. This indicated a low LF or high IE values using chloride tracer method as compared with the values obtained with water balance method.

Similarly, an LF of 0.17 ± 0.02 (IE = 83 ± 2.0%) during growing season 1 and an LF of 0.17 ± 0.007 (IE = 83 ± 0.7%) during growing season 2 were obtained for the drip-irrigated field using (3) (Table 7), whereas the LF obtained using the water balance method was 0.23 (or IE = 0.77%) during growing season 1 and 0.18 (or IE = 0.82%) during growing season 2. Similar to the furrow-irrigated field, the chloride tracer technique underestimated the LF and overestimated the IE for the drip-irrigated field. During both the growing seasons, LF was significantly higher ($P < 0.05$) for the furrow- than drip-irrigated field.

Average IEs ranging from 45 to 77% were reported for onion under drip irrigation systems, in which five different irrigation applications of 40, 60, 80, 100, and 120% of the nonstressed E_t were applied to onion crop at the FGRC in Las Cruces, NM [42]. In the present study, by contrast, irrigation applications of 79% and 83% during growing seasons 1 and 2, respectively, of the nonstressed E_t were applied to furrow-irrigated field whereas irrigation applications of 81% during growing season 1 and 72% during growing season 2 of the nonstressed E_t were applied to drip-irrigated field. The high IE obtained in this study under both irrigation systems is due to the deficit irrigation practiced in the study area to maximize yield from a unit of water rather than from a unit of land, since water is a limited resource in this region.

3.10. Best Management Approach. Plant N uptake was measured as 104 ± 0.21 kg ha^{-1} and 106 ± 0.1 kg ha^{-1} during growing seasons 1 and 2, respectively, in furrow- and 112 ± 0.20 kg ha^{-1} during growing season 1 and 111 ± 0.27 kg ha^{-1} during growing season 2 in drip-irrigated fields. Nitrogen application efficiency can theoretically be as high as water application efficiency; however, traditional best management practices result in NAEs of 50% [35]. Since NAEs in the present study were 35% and 36% in growing seasons 1 and 2, respectively, for furrow- and 38% in growing season 1 and 39% in growing season 2 for drip-irrigated fields, this indicated considerable potential for improvement in nitrogen management in onion crop of NM.

The average soil NO_3-N was high (84 kg ha^{-1} during growing season 1 and 94 kg ha^{-1} during growing season 2 in furrow- and 52.2 kg ha^{-1} during growing season 1 and 78 kg ha^{-1} during growing season 2 in drip-irrigated fields) in the early growing season when there was very low

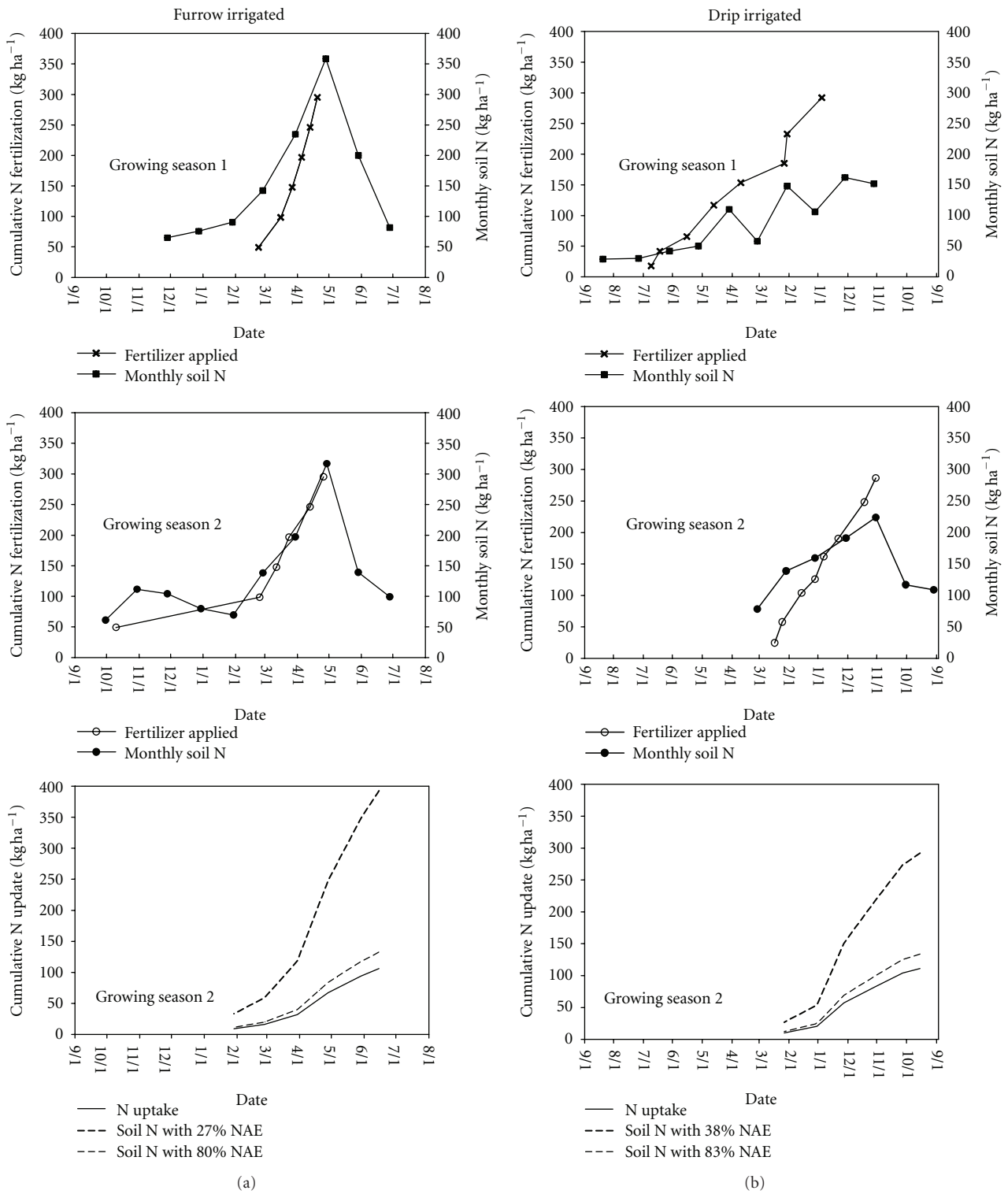


Figure 4: Cumulative N fertilization (kg ha^{-1}), cumulative N uptake (kg ha^{-1}) and monthly soil N (kg ha^{-1}) during two onion growing seasons each in furrow- (2006–09) and drip-irrigated (2006–08) onion fields in NM. NAE is the nitrogen application efficiency.

(<11 kg ha^{-1}) onion N uptake (Figure 4). Halvorson et al. [9] reported that onions need a maximum amount of N during bulbing, when rapid formation of bulb dry matter takes place. Therefore, better N management would be to start N application just before bulbing (early March) to provide N during the period of maximum need by onion plants and possibly reduce NO_3-N leaching, hence improving NAE in both fields.

In the furrow-irrigated field, soil NO_3-N was 235 kg ha^{-1} and 197 kg ha^{-1} during March and it increased to 359 kg ha^{-1} and 317 kg ha^{-1} during April in growing seasons 1 and 2, respectively, as fertilizer applications were made during March and April (Figure 4). Most of the excess soil NO_3-N probably leached with irrigation, as the soil NO_3-N during May decreased compared to soil NO_3-N during April. As the total onion N uptake was 104 kg ha^{-1} and 106 kg ha^{-1} during the entire growing seasons 1 and 2, respectively, therefore, a soil N of 150 kg ha^{-1} is sufficient throughout the growing season starting from March. This soil NO_3-N concentration can be maintained by reducing the amount of N to half during March and April, with a total N fertilizer application of 196 kg ha^{-1} through the entire growing season.

A single application of N before March might be sufficient for onion plants in drip-irrigated field, and N applications during December, January, and February during growing season 1 could be skipped for better N management (Figure 4). Most of the N fertilizer applied before March probably leached with irrigation water, as onion roots were only 10 to 15 cm deep until March. Curtailing excess fertilizer applications before March would reduce the total fertilizer application from 292 to 190 kg ha^{-1} during growing season 1. Similarly, by reducing the current N fertilizer applied during March, April, and June during the growing season 2 to half would reduce the total fertilizer application from 286 to 175 kg ha^{-1}. These results also showed that regular soil sampling is important to monitoring soil NO_3-N throughout the growing season.

The limitation of this study, in the drip-irrigated field, could be the difference in the planting dates of onion. However, this is a common practice in this area where farmers grow onion during fall followed by spring onion. Despite of this limitation, this study provided a detailed sketch of NO_3-N leaching and discussed the improvements that can be made to reduce the NO_3-N leaching in the farmers' fields of New Mexico.

4. Conclusions

Greater N concentrations were found in the onion crop rooting zone than below the rooting zone depth in both the furrow- and drip-irrigated onion in NM. The average NO_3-N loading flux below the rooting zone was 150 ± 2.2 kg ha^{-1} and 145 ± 0.5 kg ha^{-1} during growing seasons 1 and 2, respectively, at an average volumetric water content of 0.19 cm^3 cm^{-3} in furrow-irrigated field. Similarly, average NO_3-N loading flux below the rooting zone of drip-irrigated field was 79 ± 6.9 kg ha^{-1} and 76 ± 0.3 kg ha^{-1} during growing seasons 1 and 2, respectively, at an average volumetric water content of 0.32 cm^3 cm^{-3}. The ratio of NO_3-N and Cl decreased with increasing soil depth and was similar below the onion rooting zone (50–200 cm) in the furrow- and drip-irrigated fields. A leaching fraction of 0.20 ± 0.006 and 0.22 ± 0.004 during growing seasons 1 and 2, respectively, was obtained for the furrow-irrigated field and 0.17 ± 0.02 during growing season 1 and 0.17 ± 0.007 during growing season 2 for the drip-irrigated field using the chloride tracer technique. Therefore, irrigation efficiencies (1 − LF) under both systems were high: 80 ± 0.6% and 78 ± 0.004% during growing seasons 1 and 2, respectively, for the furrow-irrigated field and 83 ± 2.0% during growing season 1 and 83 ± 0.7% during growing season 2 for the drip-irrigated field. The chloride tracer technique underestimated the leaching fractions and, therefore, overestimated the IEs for both irrigation systems compared to the water-balance method. Nitrogen application and use efficiencies were low in both fields because of high levels of available N in the root zone due to application of excess N fertilizer compared with the total amount of N taken up by the onion plants. Reducing N application rates by half and delaying N applications until onion bulbing (early March) starts may improve N application and use efficiencies and potentially reduce the N loading in deeper soil layers. More frequent and smaller amounts of water and fertilizer applications can increase retention and reduce the leaching depth of water and fertilizer.

Acknowledgment

The authors thank the Agricultural Experimental Station of New Mexico State University for funding the project.

References

[1] F. E. Allison, "The fate of nitrogen applied to soils," *Advances in Agronomy*, vol. 18, pp. 219–258, 1966.

[2] G. A. Peterson and J. F. Power, "Soil, crop, and water management," in *Managing nitrogen for groundwater quality and farm profitability*, R. F. Follett, Ed., pp. 189–198, Soil Science Society of America, Madison, Wis, USA, 1991.

[3] M. S. Al-Jamal, T. W. Sammis, and T. Jones, "Nitrogen and chloride concentration in deep soil cores related to fertilization," *Agricultural Water Management*, vol. 34, no. 1, pp. 1–16, 1997.

[4] P. Cepuder and M. K. Shukla, "Groundwater nitrate in Austria: a case study inTullnerfeld," *Nutrient Cycling in Agroecosystems*, vol. 64, no. 3, pp. 301–315, 2002.

[5] "Diagnosis and Improvement of saline and alkali soils," in *Handbook No. 60*, L. A. Richards, Ed., United States Department of Agriculture, Washington, DC, USA, 1954.

[6] W. Stites and G. J. Kraft, "Nitrate and chloride loading to groundwater from an irrigated north-central U.S. sand-plain vegetable field," *Journal of Environmental Quality*, vol. 30, no. 4, pp. 1176–1184, 2001.

[7] L. J. Lund, "Variations in nitrate and chloride concentrations below selected agricultural fields," *Soil Science Society of America Journal*, vol. 46, no. 5, pp. 1062–1066, 1982.

[8] L. Bergstrom and R. Johansson, "Leaching of nitrate from monolith lysimeters of different types of agricultural soils," *Journal of Environmental Quality*, vol. 20, no. 4, pp. 801–807, 1991.

[9] A. D. Halvorson, R. F. Follett, M. E. Bartolo, and F. C. Schweissing, "Nitrogen fertilizer use efficiency of furrow-irrigated onion and corn," *Agronomy Journal*, vol. 94, no. 3, pp. 442–449, 2002.

[10] T. W. Sammis, "Nutrient management of onions: a New Mexico perspective," in *Proceedings of the Western Nutrient*

Management Conference, vol. 2, pp. 49–53, Salt Lake City, Utah, USA, March 1997.

[11] D. Romic, M. Romic, J. Borosic, and M. Poljak, "Mulching decreases nitrate leaching in bell pepper (*Capsicum annuum* L.) cultivation," *Agricultural Water Management*, vol. 60, no. 2, pp. 87–97, 2003.

[12] D. A. Bucks, L. J. Drie, and O. F. French, "Quality and frequency of trickle and furrow irrigation for efficient cabbage production," *Agronomy Journal*, vol. 66, no. 1, pp. 53–56, 1974.

[13] C. D. Gustafson, "Drip irrigation-worldwide 1975, present status and outlook for drip irrigation," Survey Report of University of California, University of California, San Diego, Calif, USA, 1975.

[14] A. D. Halvorson, M. E. Bartolo, C. A. Reule, and A. Berrada, "Nitrogen effects on onion yield under drip and furrow irrigation," *Agronomy Journal*, vol. 100, no. 4, pp. 1062–1069, 2008.

[15] P. F. Pratt, L. J. Lund, and J. M. Rible, "An approach to measuring leaching of nitrate from freely drained irrigated fields," in *Nitrogen Behavior in Field Soil*, D. R. Nielsen and J. G. MacDonald, Eds., vol. 1, pp. 223–265, Academic Press, New York, NY, USA, 1978.

[16] Z. Samani, T. Sammis, R. Skaggs, N. Alkhatiri, and J. Deras, "Measuring on-farm irrigation efficiency with chloride tracing under deficit irrigation," *Journal of Irrigation and Drainage Engineering*, vol. 131, no. 6, pp. 555–559, 2005.

[17] D. P. Genereux, S. J. Wood, and C. M. Pringle, "Chemical tracing of interbasin groundwater transfer in the lowlandrainforest of Costa Rica," *Journal of Hydrology*, vol. 258, no. 1–4, pp. 163–178, 2002.

[18] M. K. Shukla and P. Cepuder, "Anion exclusion during transport of chloride through soil columns," *Transactions of the American Society of Agricultural Engineers*, vol. 43, no. 6, pp. 1425–1430, 2000.

[19] M. K. Shukla, T. R. Ellsworth, R. J. Hudson, and D. R. Nielsen, "Effect of water flux on solute velocity and dispersion," *Soil Science Society of America Journal*, vol. 67, no. 2, pp. 449–457, 2003.

[20] B. A. Stewart, "Critique of an approach to measuring leaching of nitrate from freely drained irrigated fields," in *Nitrogen Behavior in Field Soil*, D. R. Nielson and J. G. MacDonald, Eds., vol. 1, pp. 267–273, Academic Press, New York, NY, USA, 1978.

[21] H. E. Bulloch and R. E. Neher, *Soil Survey of Dona Ana County Area New Mexico*, United States Department of Agriculture, Soil Conservation Service, 1980.

[22] L. H. Gile, J. W. Hawley, and R. B. Grossman, *Soils and Geomorphology in the Basin and Range Area of Southern New Mexico—Guidebook to the Desert Project*, New Mexico Bureau of Mines and Mineral Resources, New Mexico, NM, USA, 1981.

[23] V. E. Hansen, O. W. Israelsen, and G. E. Stringgham, *Irrigation Principles and Practices*, John Wiley and Sons, New York, NY, USA, 1979.

[24] G. R. Blake and K. H. Hartge, "Bulk density," in *Methods of Soil Analysis. Part 1*, A. Klute, Ed., Agronomy Monograph, 9, pp. 363–373, American Society of Agronomy, Soil Science Society of America, Madision, Wis, USA, 2nd edition, 1986.

[25] A. Klute and C. Dirksen, "Hydraulic conductivity and diffusivity: laboratory methods," in *Methods of Soil Analysis. Part 1*, A. Klute, Ed., Agronomy Monograph, 9, pp. 387–734, American Society of Agronomy, Soil Science Society of America, Madision, Wis, USA, 2nd edition, 1986.

[26] W. H. Gardner, "Water content," in *Methods of Soil Analysis. Part 1*, A. Klute, Ed., Agronomy Monograph, 9, pp. 493–544, American Society of Agronomy, Soil Science Society of America, Madision, Wis, USA, 2nd edition, 1986.

[27] G. W. Gee and J. W. Bauder, "Particle size analysis," in *Methods of Soil Analysis. Part 1*, A. Klute, Ed., Agronomy Monograph, 9, pp. 337–382, American Society of Agronomy, Soil Science Society of America, Madision, Wis, USA, 2nd edition, 1986.

[28] D. G. Maynard and Y. P. Kalra, "Nitrate and exchangeable ammonium nitrogen," in *Soil Sampling and Methods of Analysis*, M. R. Carter, Ed., pp. 25–38, Canadian Society of Soil Science, Lewis Publishers, 1993.

[29] M. S. Al-Jamal, T. W. Sammis, S. Ball, and D. Smeal, "Yield-based, irrigated onion crop coefficients," *Applied Engineering in Agriculture*, vol. 15, no. 6, pp. 659–668, 1999.

[30] T. W. Sammis, C. L. Mapel, D. G. Lugg, R. R. Lansford, and J. T. McGuckin, "Evapotranspiration crop coefficients predicted using growing-degree-days," *Transactions of the American Society of Agricultural Engineers*, vol. 28, no. 3, pp. 773–780, 1985.

[31] K. Thorup-Kristensen, "Root growth and nitrogen uptake of carrot, early cabbage, onion and lettuce following a range of green manures," *Soil Use and Management*, vol. 22, no. 1, pp. 29–38, 2006.

[32] SAS Institute Incorporation, *SAS 9.1 for Windows*, Version 9.1.3, Cary, NC, USA, 2002-2003.

[33] N. C. Brady and R. R. Weil, *Elements of the Nature and Properties of Soils*, Person Education, Upper Saddle River, NJ, USA, 2nd edition, 2004.

[34] Soil Improvement Committee and California Fertilizer Association, *Western Fertilizer Handbook*, The Interstate Printers and Publishers, Danville, Ill, USA, 6th edition, 1980.

[35] D. M. Sullivan, B. D. Brown, and C. C. Shock, *Nutrient Management for Onions in the Pacific Northwest*, PNW Extension Bulletin, Washigton, DC, USA; Oregon State University, Corvallis, Ore, USA, 2001.

[36] United States Public Health Service, *Drinking Water Standards*, vol. 956, United States Public Health Service, 1962.

[37] J. Mikołajków, "Laboratory methods of estimating the retardation factor of migrating mineral nitrogen compounds in shallow groundwater," *Geological Quarterly*, vol. 47, no. 1, pp. 91–96, 2003.

[38] J. C. Ryden, L. J. Lund, J. Letey, and D. D. Focht, "Direct measurement of denitrification loss from soils: II. Development and application of field methods," *Soil Science Society of America Journal*, vol. 43, pp. 110–118, 1979.

[39] N. Hofstra and A. F. Bouwman, "Denitrification in agricultural soils: summarizing published data and estimating global annual rates," *Nutrient Cycling in Agroecosystems*, vol. 72, no. 3, pp. 267–278, 2005.

[40] T. W. Bruulsema and B. R. Christie, "Nitrogen contribution to succeeding corn from alfalfa and red clover," *Agronomy Journal*, vol. 79, pp. 96–100, 1987.

[41] J. L. Gil and W. H. Fick, "Soil nitrogen mineralization in mixtures of eastern gamagrass with alfalfa and red clover," *Agronomy Journal*, vol. 93, no. 4, pp. 902–910, 2001.

[42] M. S. Al-Jamal, S. Ball, and T. W. Sammis, "Comparison of sprinkler, trickle and furrow irrigation efficiencies for onion production," *Agricultural Water Management*, vol. 46, no. 3, pp. 253–266, 2001.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 696174, 9 pages
doi:10.1155/2012/696174

Review Article

Soil Health Management under Hill Agroecosystem of North East India

R. Saha, R. S. Chaudhary, and J. Somasundaram

Indian Institute of Soil Science, Indian Council of Agricultural Research, Nabibagh, Berasia Road, Bhopal, Madhya Pradesh 462 038, India

Correspondence should be addressed to R. Saha, saharitesh74@rediffmail.com

Received 2 December 2011; Revised 2 February 2012; Accepted 3 February 2012

Academic Editor: María Cruz Díaz Álvarez

The deterioration of soil quality/health is the combined result of soil fertility, biological degradation (decline of organic matter, biomass C, decrease in activity and diversity of soil fauna), increase in erodibility, acidity, and salinity, and exposure of compact subsoil of poor physicochemical properties. Northeast India is characterized by high soil acidity/Al^{+3} toxicity, heavy soil, and carbon loss, severe water scarcity during most parts of year though it is known as high rainfall area. The extent of soil and nutrient transfer, causing environmental degradation in North eastern India, has been estimated to be about 601 million tones of soil, and 685.8, 99.8, 511.1, 22.6, 14.0, 57.1, and 43.0 thousand tones of N, P, K, Mn, Zn, Ca, and Mg, respectively. Excessive deforestation coupled with shifting cultivation practices have resulted in tremendous soil loss (200 t/ha/yr), poor soil physical health in this region. Studies on soil erodibility characteristics under various land use systems in Northeastern Hill (NEH) Region depicted that shifting cultivation had the highest erosion ratio (12.46) and soil loss (30.2–170.2 t/ha/yr), followed by conventional agriculture system (10.42 and 5.10–68.20 t/ha/yr, resp.). The challenge before us is to maintain equilibrium between resources and their use to have a stable ecosystem. Agroforestry systems like agri-horti-silvi-pastoral system performed better over shifting cultivation in terms of improvement in soil organic carbon; SOC (44.8%), mean weight diameter; MWD (29.4%), dispersion ratio (52.9%), soil loss (99.3%), soil erosion ratio (45.9%), and *in-situ* soil moisture conservation (20.6%) under the high rainfall, moderate to steep slopes, and shallow soil depth conditions. Multipurpose trees (MPTs) also played an important role on soil rejuvenation. *Michelia oblonga* is reported to be a better choice as bioameliorant for these soils as continuous leaf litter and root exudates improved soil physical behaviour and SOC considerably. Considering the present level of resource degradation, some resource conservation techniques like zero tillage/minimum tillage, hedge crop, mulching, cover crop need due attention for building up of organic matter status for sustaining soil health.

1. Introduction

Soil degradation has raised some serious debate, and it is an important issue in the modern era. It refers to the decline in soil's inherent capacity to produce economic goods and perform ecologic functions. It is the net result of dynamic soil degradative and restorative processes regulated by natural and anthropogenic factors. The degree of soil degradation depends on soil's susceptibility to degradative processes, land use, the duration of degradative land use, and the management. Soil and water degradation are also related to overall environmental quality, of which water pollution and the "greenhouse effect" are two major concerns of global significance. Recent global concerns over increased atmospheric CO_2, which can potentially alter the earth's climate systems, have resulted in raising interest in studying Soil organic matter (SOM) dynamics and carbon (SOC) sequestration capacity in various ecosystems [1]. Soils represent an important terrestrial stock of C and approximately two to three times as much as terrestrial vegetation and atmosphere, respectively, and the C in the SOM of agricultural land is composed of dominant terrestrial C stock. Soil quality is the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health and thus has a profound effect on the health and productivity of a given ecosystem and the environment related to it.

The North Eastern parts of India, comprising the states of Arunachal Pradesh, Assam, Manipur, Meghalaya, Mizoram,

Table 1: Trends of forest loss/gain (km^2) in NEH region.

States	1999 Assessment			2001 Assessment			Net difference
	Dense forest cover	Open forest cover	Total forest cover	Dense forest cover	Open forest cover	Total forest cover	
Arunachal Pradesh	57,756	11,091	68,847	53,932	14,113	68,045	(−) 802
*Assam	15,548	8,276	23,824	14,517	9,171	23,688	(−)136
Manipur	5,936	11,448	17,384	5,710	11,216	16,926	(−) 458
Meghalaya	5,925	9,708	15,633	5,681	9,903	15,584	(−) 49
Mizoram	3,786	14,552	18,338	8,936	8,558	17,494	(−) 844
Nagaland	5,137	9,027	14,164	5,393	7,952	13,345	(−) 819
Sikkim	2,363	755	3,118	2,391	802	3,193	75
Tripura	2,228	3,517	5,745	3,463	3,602	7,065	1320
**Total	83,131	60,098	1,43,229	85,506	56,146	1,41,652	(−) 1577

*Data for Assam is during the assessment year of 1997–1999 and **total reports only for NEH region.
Source: [4].

Nagaland, Sikkim, and Tripura, lies between 22°05′ and 29°30′ N latitudes and 87°55′ and 97°24′ E longitudes. The region is characterised by diverse agroclimatic and geographical situations. About 54.1 per cent of the total geographical area is under forests, 16.6 per cent under crops, and the rest either under nonagricultural uses or uncultivated land. The low area under agricultural crops is due to natural corollary of the physiographic features of the region, as major chunk of the land has more than 15 per cent slope, undulating topography, highly eroded and degraded soils, and inaccessible terrain. Continuous dilution of the forest cover in the region due to shifting cultivation, firewood, and timber collection is posing the most crucial problem resulting in poor soil health and environmental degradation in the hills.

2. Shifting Cultivation

Shifting cultivation, also known as *Jhum* cultivation, is the most traditional and dominant land use system in this region. On an average, 3,869 km^2 area is put under shifting cultivation every year. Shifting cultivation in its more traditional and cultural integrated form is an ecological and economically viable system of agriculture as long as population densities are low and *jhum* cycles are long enough to maintain soil fertility. The system involves cultivation of crops in steep slopes. Land is cleared by cutting of forests, bushes, and so forth up to the stump level, leaving the cut materials for drying and finally burning to make the land ready for sowing of seeds of different crops before the onset of rains. The cultivation is confined to a village boundary and often after two or three years, the cultivated area is abandoned and a new site is selected to repeat the process. The shifting cultivation became unsustainable today primarily due to the increase in population that led to increase in food demand. *Jhuming* cycle in the same land, which extended to 20–30 years in earlier days, has now been reduced to 3–6 years [2]. Land degradation in the region is 36.64% of the total geographical area, which is almost double than the national average of 20.17% [3]. The problem of land degradation is much serious in the states like Manipur, Nagaland, and Sikkim, where more than 50% of total geographical area is defined as wastelands. Of various degradation types, water erosion, reduced infiltration, acidification, nutrient leaching, burning of vegetation, decline in vegetative cover, and biodiversity are important in context to the NE region.

3. Effect of Shifting Cultivation

3.1. Change in Forest Cover. The total forest cover in the region is 1,41,652 km^2, which is about 54.1% of the geographic area as against the national average of 19.39% [4]. Manipur and Meghalaya have dense forest cover of 25.57 and 25.33%, respectively (Table 1). Similarly for Nagaland, Sikkim, Tripura, and Mizoram, the dense forest cover is 32.53, 33.70, 33.02 and 42.39%, respectively. Among seven sisters of NEH, Arunachal Pradesh is the only state, which has the dense forest cover of 64.0%. Since shifting cultivation is still practiced in the region, and every year dense forest is converted into *jhum* fields, there is drastic reduction in dense forest cover (canopy density > 40%) in most of the states.

3.2. Effect of Burning on Soil Fertility. The burning process related to shifting cultivation practices has tremendous effect on soil ecosystem. The impact of fire on ecosystem is profound and its consequences are dependent on intensity and frequency of fire, proportion of biomass burned, the time of monsoon setting, and total annual precipitation. The extent to which organic matter is transformed into ash depends on a number of factors *viz* intensity and duration of fire, fuel load, moisture content in the fuel, weather, and topography. Burning of above-ground vegetation showed an increase in pH and cations and a decrease in carbon and nitrogen contents in the surface soil [5]. Quick release of nutrients especially cations after burning has been reported by Kellman et al. [6]. The organic carbon content of soil decreased drastically after burning because of oxidation loss. Rise in pH, temperature, and bases of the soil might have increased the microbial activity after burning which in

Table 2: Effect of various MPTs on soil physical properties.

MPTs	Organic C (g kg^{-1})	Aggregate stability	Available water (m^3 m^{-3})	Infiltration rate (mm h^{-1})	Erosion ratio
Pinus kesiya	35.4	75.6	0.220	8.04	0.20
Alnus nepalensis	32.2	72.1	0.201	7.28	0.23
Parkia roxburghii	23.1	63.4	0.192	4.85	0.30
Michelia oblonga	33.6	73.2	0.210	6.10	0.22
Gmelina arboria	28.6	67.9	0.183	5.36	0.24
Control (No tree)	15.6	56.8	0.151	3.84	0.39

Source: [16].

turn resulted in accelerating mineralization of organic N to inorganic forms [7, 8].

3.3. Soil Erosion and Nutrient Loss. Soil erosion under shifting cultivation is highly erratic from year to year depending on rainfall characteristics. Studies on steep slopes (44–53%) have indicated the soil loss to the tune of 40.9 t/ha and the corresponding nutrient losses per hectare are 702.9 kg of organic carbon, 63.5 kg of P and 5.9 kg of K [9]. The soil loss from hill slopes (60–79%) under first year, second year, and abandoned *jhum* was estimated to be 147, 170, and 30 t/ha/yr [10]. In general, tolerable soil loss (T) value is 11.2 Mg/ha/yr (5.0 t/ac/yr) while it is between 5.0 and 12.5 Mg/ha/yr (2.2 and 5.6 t/ac/yr) in North West Himalayas [11]. During first few years of clearing, carbon and nitrogen levels decrease rapidly. According to one estimate annual loss of top soil, N, P and K due to shifting cultivation is 88346, 10669, 0.372, and 6051 thousand tones in the region [12]. Singh et al. [13] reported nutrient loss to the tune of 6.0 million tones of organic carbon, 9.7 tones of available P, and 5690 tones of K from the NEH region. Nutrient losses from the *jhum* field through runoff and percolation are rather heavy during cropping.

4. Long-Term Strategies for Resource Conservation and Improvement in Soil Health

Nearly 37.1% of the total geographical area in Northeast India is under the threat of land degradation, where erosion is a major land degradative process. With the great concern of poor soil health and severe land degradation, there is a need of viable option for ecorestoration and maintenance of soil resources which could sustain long-term soil productivity and improve food security of the poor tribal farmers of northeast India under the humid subtropical climate of the north-eastern Himalayan region. Three broad strategies, suitable for different land situation, elevation, and topography prevailing in this region, are discussed here.

4.1. Multipurpose Trees (MPTs). The multipurpose tree species (MPTs) form an integral component of different agroforestry interventions in crop sustainability. The MPTs, besides furnishing the multiple outputs like fuel, fodder, timber, and other miscellaneous products, help in improvement of soil health and other ecological conditions. Farmers of the region integrate various tree species in different land use in the region; however, priority species vary from state to state and even from place to place within a state based on ethnic diversity and food habits of the tribal communities. In the region, as many as 40 promising species are cultivated in tropical and subtropical region, and 30 in temperate zone of the region in different farming systems. Besides, 28 bamboo species and 2 genera of cane also find a place in various agroforestry programmes. Tree density ha^{-1} is also a crucial factor on sloppy lands. In general, optimum tree density in case of agri-horticulture system is 400 trees/ha, while in agri-silviculture, it is 200 plants/ha so as to minimize the effect of shade and biochemical interactions on growth and production of agricultural crops [14, 15].

Long-term effect of various multipurpose tree species on soil physical behaviour has been studied [16]. Multipurpose tree species with greater surface cover, constant leaf litter fall, and extensive root system increased soil organic C by 96.2%, porosity by 10.9%, aggregate stability by 24.0%, and available soil moisture by 33.2% and simultaneously reduced bulk density and erosion ratio by 15.9 and 39.5%, respectively (Table 2). Among the tree species tested, *P. kesiya, M. oblonga* and *Alnus nepalensis* were found suitable as bioameliorant in hilly terrain of northeast India in terms of organic matter buildup through presence of leaf litter, better soil aggregation, transmissivity, and infiltrability through extensive root system, improved soil conservation through constant surface cover with leaf biomass. Such improvement in soil hydrophysical properties in tree-based system has a direct bearing on long-term sustainability, productivity, and soil quality in hilly ecosystem.

4.2. Agroforestry Interventions in Degraded Lands. The region has a very high rate of land degradation. In this region, 7.85 million ha area is degraded which need rehabilitation through various agroforestry models [3]. Agroforestry system (AFS) has today become an established approach of integrated land management system not only for renewable resource production but also for ecological consideration. It represents the integration of agriculture and forestry to increase the productivity and sustainability of farming system.

4.2.1. Soil Fertility Buildup. Study revealed that organic carbon, available P, and exchangeable cations contents in surface soil ranged in between 2.0–2.5%, 10.4–13.2 ppm, and 5.9–8.4 cmol (p$^+$) kg^{-1}, respectively, under jackfruit-based AFS,

Table 3: Effect of agroforestry systems on soil properties.

Soil properties	Agroforestry systems					
	Agrisilviculture	Agrihorti culture (khasi mandarin + crops)	Agrihorti culture (Assam lemon + crops)	Silvihorti pastoral (Alder + pine apple + fodder grass)	Multistoried AFS (Alder + tea + black pepper + crops)	Natural forest
pH	4.65	4.62	4.80	4.25	4.61	4.62
Organic C (%)	1.62	1.55	2.02	2.19	1.91	1.92
Exchangeable Ca [cmol (p^+) kg^{-1}]	0.40	0.86	0.74	0.31	0.65	0.26
Exchangeable Mg [cmol (p^+) kg^{-1}]	0.75	0.51	0.33	0.48	0.71	0.16
Exchangeable K [cmol (p^+) kg^{-1}]	0.232	0.244	0.249	0.238	0.201	0.169
Exchangeable Na [cmol (p^+) kg^{-1}]	0.201	0.220	0.194	0.195	0.197	0.196
Exchangeable Al [cmol (p^+) kg^{-1}]	2.65	2.70	2.20	3.15	2.05	2.20
Available N (ppm)	190.1	180.8	203.6	199.4	216.9	167.2
Available P (ppm)	2.75	4.10	5.36	0.94	3.36	0.63
Available Fe (ppm)	8.9	10.4	12.8	10.9	13.9	7.3
Available Mn (ppm)	0.58	0.92	0.79	0.83	1.04	0.04
Available Zn (ppm)	0.08	0.05	0.07	0.006	0.08	0.025
Available Cu (ppm)	0.21	0.23	0.37	0.30	0.27	0.10

Source: [18].

while 1.5–1.8% organic carbon, 3.8–6.7 ppm available P, and 3.9–5.9 cmol (p^+) kg^{-1} total cations were found under arecanut/khasi mandarin-based AFS [17].

In an another study, long-term effect of agri-horticulture (comprising *Khasi* mandarin + agricultural crops, and Assam lemon + agricultural crops), agri-silviculture (multipurpose tree species + annual agricultural crops), silvi-horti-pastoral (alder + pine apple + fodder grasses), and multistoried AFS (alder + tea + black pepper + annual agricultural crops between the tree rows) on soil properties and fertility status was evaluated in acid Alfisol of Meghalaya compared with natural forest as a control. In all the AFS, significant (1.17–1.65 fold) increase in organic carbon was found as compared to initial status, the maximum contribution being by silvi-horti-pastoral AFS. The same system also registered 43.2% higher exchangeable Al compared to natural forest and consequently a maximum decrease of 0.50 units in pH (Table 3). The exchangeable Ca, Mg, Na, K, and Al and available N, P, and K content were higher in all the systems compared to natural forest and the content of these nutrients decreased with increasing soil depth [18].

In a study under Farming System Research Project (FSRP) carried out at ICAR Research Complex, Barapani, effect of various AFS like silvi-pastoral, silvi-horticulture, agri-horti-silvipastoral has been evaluated [19] after 17 years of their adoption on soil fertility indices (Table 4). The natural fallow and abandoned *jhum* land at Umiam were taken for comparison. Organic carbon content increased in all the AFS including natural fallow, however, the quantity largely depended on the nature of vegetation in different systems. Adoption of different cropping pattern in various AFS markedly influenced the exchangeable Ca, Mg, and K content in the soil. Maximum accumulation of these cations was recorded under agri-horti-silvipastoral and silvi-horticulture AFS followed by natural fallow and silvi-pastoral systems. Accumulation of exchangeable K was maximum in silvi-horticulture followed by agri-horti-silvipastoral. The available N, P, and S were higher in agri-horti-silvipastoral and silvi-horticulture compared to natural fallow and silvi-pastoral AFS.

4.2.2. Soil Physical Health. Effect of various land use systems on soil physical properties shown in Table 5 indicated that the maximum reduction in bulk density over shifting cultivation was recorded in forest (17.6%) followed by agri-horti-silvi-pastoral (14.3%), livestock based (13.4%), natural fallow, and agriculture system (12.6%). Higher percentage of macroaggregates (54.5%), organic C content (2.95%), and biotic activity were also observed in forest ecosystem. Soil biota influences soil properties through formation of stable aggregates, development of organomineral complexes by improving macroporosity and continuity of pores from surface to the subsoil which ultimately increase the water transmission and reduce run-off. Higher transmission and storage pore volume coupled with lower value of residual pores associated with modified land use systems as compared to shifting-cultivated plots was thus an indication of maintaining the pore geometry of the soil under these systems.

Table 4: Effect of different land use systems developed under FSRP, Meghalaya on soil properties.

Characteristics	Agroforestry systems				
	Natural fallow	Abandoned *jhum* land	Silvipastoral	Agri-horti-silvipastoral	Silvihorti-culture
pH	4.99 (4.90)	4.76 (5.20)	4.52 (5.10)	4.92 (4.90)	4.91 (4.90)
Organic C (%)	2.94 (1.85)	3.42 (1.90)	2.61 (1.80)	2.97 (1.82)	2.97 (1.80)
Exchangeable Ca [cmol (p^+) kg^{-1}]	1.96 (1.15)	1.57 (1.16)	1.25 (1.10)	2.11 (1.20)	2.00 (1.20)
Exchangeable Mg [cmol (p^+) kg^{-1}]	0.55 (1.15)	0.38 (1.16)	0.43 (1.20)	1.45 (1.20)	0.85 (0.60)
Exchangeable Al [cmol (p^+) kg^{-1}]	0.88	1.30	1.56	0.90	0.90
Available N (ppm)	179.2	251.1	214.5	220.3	210.9
Available P (ppm)	1.9	2.0	2.1	16.6	12.9
Available K (ppm)	175.6	130.8	98.0	162.7	265.0
Available S (ppm)	14.5	14.8	10.4	19.9	12.9

Figures in parentheses indicate the initial values at the start of the project.
Source: [19].

Table 5: Effect of different land use systems developed under FSRP, Meghalaya on soil physical properties.

Soil properties	Land use systems					
	Agriculture	Agri-horti-silvipastoral	Forestry	Livestock based	Natural fallow	Shifting cultivation
Bulk density (Mg m^{-3})	1.04	1.02	0.98	1.03	1.04	1.19
Total Porosity (%)	59.67	60.47	62.02	60.08	66.23	53.88
Macroaggregates (>0.25 mm)	21.72	54.19	54.47	50.02	50.53	18.17
Microaggregates (<0.25 mm)	47.85	23.23	23.81	22.80	21.90	42.34
MWD (mm)	2.76	2.99	3.16	2.85	2.93	2.31
Available water (m^3 m^{-3})	0.210	0.222	0.231	0.220	0.233	0.169
Hydraulic Conductivity (cm hr^{-1})	2.74	4.72	5.47	2.95	6.66	2.09

Source: [20].

The better soil aggregation under natural forest, multistoried AFS, and silvihortipastoral systems maintaining intensive vegetative cover throughout the year could be ascribed to the effect of higher percentage of organic matter, clay content, and high amount of Al and Fe oxides in soil.

4.2.3. Soil and Water Conservation. Some of the potential farming systems such as agriculture on bench terraces, horticulture, and agri-horti-silvipastoral systems have been evaluated [21] at the experimental watershed of ICAR Research Complex at Barapani for long-term runoff, soil and nutrient losses, production behaviour, biotic and abiotic changes, and so on. The data indicated that mixed land use systems with appropriate soil conservation measures, namely, bench terraces, contour trenches, and so forth, were the most effective in retaining 90–100% annual rainfall and simulated the effects of natural forest. The contributions to stream flow in the watersheds having substantial area under natural forest is primarily by subsurface flow (base flow). The watersheds having continuous stream flow characteristics generated base flow to the extent of 70–90% of its total water yields. As expected, the watershed treated with jhum (shifting) cultivation yielded the highest peak runoff while the one left undisturbed with natural vegetation gave the minimum peak runoff. The results revealed that agroforestry and other mixed land use systems most effectively conserved moisture and substantially reduced peak runoff (Tables 6(a) and 6(b)). The low erosion ratio values in silvi-horti-pastoral and multistoried AFS (3.07 and 3.06, resp.) showed that these systems were the most suitable for soil and water conservation in hilly ecosystem [22]. This could be ascribed to the effect of heavy litter fall, which might have increased the cohesiveness in the soil system after decomposition and also binds the soil tightly in lower horizons by their deep root systems.

4.2.4. Soil C Sequestration Potential. Assessment of soil quality is an invaluable tool in determining the sustainability and environmental impact of agricultural ecosystems. Soil quality under different agroecosystems using soil organic carbon (SOC) and soil microbial C (SMBC) as soil quality indicators suggests that the shifting cultivated areas had the lowest SMBC value of 192 mg/kg while soil under *Michelia oblonga* plantation had the significantly ($P < 0.05$) highest

Table 6: (a) Pretreatment (year) precipitation, storm flow, peak flow rate in different land use systems. (b) posttreatment water yield, base flow, and peak flow in different land use systems (average of nine years).

(a)

Land use system	Precipitation (mm)	Threshold rainfall (mm)	Total water yield (mm)	Total water yield (% of rainfall)	Surface runoff (mm)	Peak flow (mm hr^{-1})
Dairy based farming	2249.30	363.20	27.21	1.20	27.21	3909
Forestry block	2249.30	399.90	655.21	29.12	54.30	16.94
Agroforestry	2249.30	533.70	32.55	1.45	9.90	6.17
Agropastoral	2249.30	364.30	25.50	1.13	25.50	31.86
Agrohortisilvipastoral	2249.30	348.60	4.10	0.18	4.10	10.45
Natural fallow	2249.30	541.70	2.87	0.13	2.87	13.65
Shifting cultivation	2249.30	1634.5	15.88	0.70	15.88	35.30

(b)

Land use systems	Annual water yield range (mm)	Mean water yield (mm)	Mean water yield (% of annual rainfall)	Maximum peak flow (mm hr^{-1})
Dairy based farming	0–66.699	9.56	0.37	7.81
Forestry block	67.42–1013.88	371.90	4.73	13.54
Agroforestry	39.31–648.26	241.14	9.55	12.87
Agropastoral	0.60–62.49	12.47	0.69	20.71
Agrohortisilvipastoral	0.24–121.91	28.98	1.14	12.07
Natural fallow	0–51.39	11.77	0.46	4.49
Shifting cultivation	0–517.72	102.94	4.07	86.10

Source: [21].

Table 7: Growth, litter production, fine root biomass of promising MPTs in humid tropics, and their contribution on SOC content.

MPT	Annual litter production (g m^{-2})	Time required for decomposition (days)	Total fine root biomass (g m^{-2})	Organic C (g kg^{-1})
P. kesiya	621.5	718	496.75	35.4
A. nepalensis	473.75	350	435.50	32.2
P. roxburghii	341.75	385	415.50	23.1
M. oblonga	512.25	390	462.00	33.6
G. arboria	431.75	360	419.00	28.6

Source: [16].

value of 478 mg/kg. The proportion of SMBC to total soil organic carbon (SOC) was in the range of 0.76 to 1.96% across all the systems. Multipurpose tree species like *P. kesiya*, *A. nepalensis*, *P. roxburghii*, *M. oblonga,* and *G. arboria* with greater surface cover, constant leaf litter fall, and extensive root systems increased soil organic carbon by 96.2% (Table 7), helped with better aggregate stability by 24.0%, improved available soil moisture by 33.2%, and in turn reduced soil erosion by 39.5% [16, 23]. Similarly, a comparative study on the effect of various MPTs on soil organic carbon pool (Table 8) showed a concomitant rise in SOC in soils under MPTs and a subsequent decline in soils of open space over 4–16 years. Maximum rise in SOC was noticed in soils of *A. indica* (28.6 Mg/hm^2) followed by *A. Aurculiformisi* (21.9 Mg/hm^2), *G. arborea* (21.8 Mg/hm^2), *M. Champaca* (16.7 Mg/hm^2), and so forth. The minimum rise in SOC was noted in soils under *T. grandis*. So an increase of SOC was noted from 3.8 Mg/hm^2 in soils of open space to 19.5 Mg/hm^2 in that under MPTs after 16 years. The comparatively high humin carbon present in soils under *A. auriculiformis*, *L. leucocephala,* and *G. Arborea* indicated the enhanced storage of organic carbon pool in agroforestry systems [24]. Swamy et al. [25] estimated that a six-year-old *G. arborea,* based agri-sivicultural systems in India sequestered 31.4 Mg hm^{-2} carbon.

4.3. Resource Conservation Techniques

4.3.1. Conservation Tillage. Conservation tillage are system of managing crop residue on the soil surface with minimum disturbance. The stubble mulch or reduced tillage/minimum tillage, no tillage and direct drill are components of conservation tillage. The objectives are (i) to leave enough plant residue on the soil surface at all times for water, and wind erosion control, (ii) to conserve soil and water and (iii) to reduce energy use [26]. Some of the conservation tillage practices followed in hill ecosystems are discussed here.

4.3.2. In-Situ Residue Management. Low native soil nitrogen (N) and very low phosphorus (P) coupled with apathy of farmers towards use of fertilizer is the major constraints limiting the rice productivity in NEH Region of India. Productivity and nutrient recycling potential in rice (*Oryza*

Table 8: Changes in SOC (Mg hm^{-2}) over the years under various MPTs in humid tropics.

MPTs	Years			
	4	8	12	16
A. auriculiformis	11.1	11.9	17.9	21.9
M. alba	9.9	9.9	9.9	15.9
L. leucocephala	11.5	11.5	12.8	16.7
D. sissoo	13.1	12.5	13.1	13.9
G. maculate	13.1	13.1	13.9	14.9
A. indica	10.9	10.9	14.7	28.6
M. champaca	13.9	13.7	13.9	16.9
E. hybrid	9.9	9.9	14.9	16.1
T. grandis	11.5	11.3	11.5	12.9
G. arborea	12.2	12.2	12.8	21.8
S. saman	10.6	11.3	11.3	13.9
A. procera	13.5	13.1	13.5	14.7
Open space (Control)	11.9	11.9	11.1	9.1

Source: [24].

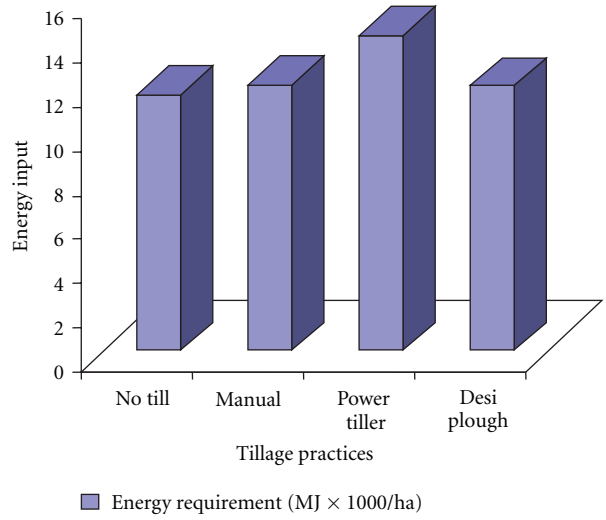


Figure 1: Energy requirement of different tillage practices. Source: [28].

sativa L.)—vegetables cropping sequences under low input *in-situ* residue management under rainfed condition was evaluated on lowland situation at ICAR Research Complex for NEH Region, Umiam, Meghalaya. After harvesting of rice, five vegetable crops, viz., tomato, potato, frenchbean, cabbage, and carrot, were grown. No external input including fertilizer, pesticides, and so forth was applied except one hand weeding at 30 days after transplanting in case of rice. In case of vegetables, only one earthing up and intercultural operations were done as per the requirement of the crops. Only the economic parts of crops were harvested and left-out portion including weed residues were chopped and incorporated into the soil. A considerable amount of nutrients were recycled through *in-situ* weed biomass incorporation. The weed biomass ranged from 37.5 q/ha with rice-tomato to 50.6 q/ha in rice-fallow. Highest amount of NPK recycling was recorded from rice-potato sequence. Soil fertility in terms of available NPK status analysed after 4 years was found stable in all the crop sequences except rice-cabbage, where it declined slightly. The soil biological properties like population of *Rhizobium*, bacteria, phosphorus solubilizing microorganisms, and earthworm activity all were found remarkably higher in experimental field compared to plots that are managed inorganically.

4.3.3. Incorporation of Jungle Grass. Long-term effects of different locally available grasses and weeds on soil hydrophysical properties and rice yield through a 5-year field experimentation under hilly ecosystem of Meghalaya depicted that incorporation of jungle grass (*Ambrosia spp.*), in puddled rice soil improved soil organic carbon (SOC) by 21.1%, the stability of microaggregates, moisture retention capacity, and infiltration rate of the soil by 82.5, 10, and 31.3%, respectively, and soil bulk density decreased by 12.6% [27]. Locally available jungle grasses are equally good as an organic amendment, which would also ease the problem of disposal of these grasses during peak monsoon. Therefore, these organic sources may serve as alternative to farm yard manure (FYM) and have a dramatic effect on long-term productivity of rice.

4.3.4. Zero Tillage. Zero tillage in rice-based system improves physical properties of soil like soil structure, increase the relative proportion of biochannels, macropores, and decrease the susceptibility of crusting. It has been observed that the bulk density of soil decreased about 25%, total porosity and soil aggregates increased by 29 and 32%, respectively, over the conventional tillage practices (2-3 passes of power-tiller/spade). It also increases the SOC content by 12.5%, available P by 14.3%, and K by 29.4% over conventional tillage. Zero tillage saved 20% energy (Figure 1) and fertilizer needs as compared other conventional tillage methods by conserving soil and water [28] without jeopardizing the crop production (rice yield of 37 q/ha). In other tillage practices like power-tilled, desiploughed, or manually weeding, the energy in terms of labour requirement was much higher.

Integrated Plant Nutrient Supply. Integrated use of balanced inorganic fertilizer in combination with lime and organic manure sustains a better soil health for achieving higher crop productivity under intensive cropping systems in hilly ecosystem of north eastern India. Study suggests that addition of NPK fertilizers along with organic manure, lime, and biofertilizers had increased SOC content, aggregate stability, moisture retention capacity, and infiltration rate of the soil while reducing bulk density. The SOC content under the treatment 100% NPK + lime + biofertilizer + FYM was significantly higher (68.6%) than control plots [29].

Pastural Development. Resource conserving and environmental friendly production strategies are desirable for agrarian economies. Grass cover is the key factor in improving soil physicochemical health by assuring regular addition of

organic matter, thus reducing surface runoff and soil erosion. Some promising perennial grasses like Setaria, Congosignal, Guinea, Napier, and Broom grass were tested for their effect on soil physicochemical properties. Study [30] revealed that continuous 15 years grass covers significantly increased the SOC, the highest SOC content with Setaria (2.24%). Similarly, Soil microbial biomass carbon, soil aggregation, and infiltration rate under various grass covers were also high as compared to plots without grass covers.

Hedgerow Intercropping. As the trees have long gestation period, farmers may be reluctant to cultivate the trees mainly due to prevailing land tenure system in the region. However, cultivating various hedgerow species even in *jhum* field could be better option for them as these species have short gestation period. Hedgerows alone reduced soil loss by 94% and run-off by 78%. When twigs and tender stem of hedge plants are used for mulch, it conserved 83% of the soil and 42% of rainfall. In a study conducted at Changki, Nagaland in NEH region, the soil loss was reduced by 22% with the incorporation of hedgerow species in the *jhum* fields compared to traditional *jhum* site (38.14 t/ha/yr). Thus contour hedgerow technology provides an option for farming on the hill slopes on a sustainable basis. Growing of nitrogen fixing hedge species on the field bunds helps in fixation of atmospheric nitrogen and reduces the leaching losses of mineral nitrogen. Their vigorous root system mobilizes phosphorus, potassium, and other trace elements.

ICAR Research Complex for NEH Region has also screened various hedgerow species for plantation, and *Cajanus cajan, Crotalaria tetragona, Desmodium rensonii, Flemingia macrophylla, Indigofera tinctoria*, *Tephrosia candida,* and *Gliricidia maculata* have been found suitable for farming in Eastern Himalayas. Survival percentage of these species ranged from 60.0 to 80.0 over degraded sites. The total N, P, and K concentration in the foliage of hedgerow species ranged from 3.23–3.86; 0.32–0.81; 1.26–1.67%, respectively. Total leaf biomass production on the dry weight basis after one year of growth was found to be highest in *C. tetragona* (22.98 q/ha) followed by *G. maculata* (20.75 q/ha), *I. tinctoria* (16.99 q/ha), and *T. candida* (15.30 q/ha). Among the hedgerow species, *C. tetragona* enriched the soil fertility more efficiently as it accumulated higher amount of total N, P, and K (79.74, 11.03, and 37.46 kg/ha) through its leaf incorporation. The recycling of bases in litter of hedgerow could potentially counteract the acidification [31]. The incorporation of leaf biomass of *T. candida* improved the pH in acid soil by increasing 0.49 units from the initial level at surface soil. Thus, the biomass produced from hedgerows showed a favorable influence on soil acidity.

4.4. Organic Farming. Organic farming is primarily in operation in areas under shifting cultivation and traditional land use systems in north east India. Nearly 57.1% of total geographical area (TGA) in India is under the threat of land degradation mainly by water erosion. On an average, 37.1% of TGA in NE India is in degraded state. Fertilizer use in most of the states of the region is far below the national average. The use of N, P, and K through fertilizer in the region is only 13.37, 11.12, and 11.0% of the crop removal thus necessasiating the organic source of nutrition in the domain of soil health management. Organic sources if pooled together can supply 13.07 kg N/ha, 7.18 kg phosphate/ha, and 7.34 kg potash/ha in NE India. The micronutrient supply from organic sources may be adequate. Substantial amount of potash can be obtained from crop residues if managed to add in soils. Biofertilizers in case of adequate supply can produce an increase (5–30%) in yield. Vermicomposting of rural wastes holds a great promise in mitigating nutrient hunger of soils in NE India considering supply of composting earthworms and need based training in compost technology. Soil amelioration with the use of limestone deposit available in north east can be brought in use. Finally, watershed based technology with proper soil and water conservation measures can be an effective avenue to nurture soil health for sustainable organic food production.

5. Epilogue

Even today, *Jhuming* is considered as a major source of rural economy in north eastern part of India and will remain as important one as it is associated with socioeconomic and cultural systems of the people of this region. Because of this, degradation will continue in the years to come and may reach to the extent of out of control, if proper care is not taken right now. Therefore, to reduce all types of degradation level, a comprehensive forest policy is required as a long-term strategy in the region for sustainability and augmentation of food, fuel, fodder, and timer requirements. In this direction, agroforestry coupled with some sound resource conservation techniques needs to be strengthened for long-term sustainable production and environmental conservations in fragile ecosystem which will contribute to improved food security and income generation for resource poor farmers and protect the environments.

Integrated farming system (IFS) has emerged as a well accepted, single window, and sound strategy for harmonizing simultaneously jointmanagement of land, water, vegetation, livestock, and human resources, The IFS developed for hill areas could reduce the risk of soil degradation, produce the soils productive potential, and reduce the risks of environmental degradation. Besides, these interventions having a tree crop with a high quality of leaf litter and root binding ability reduce erodibility of rainfall/runoff and improve the physicochemical conditions. Attempt should also be made to manage soil health through addition of organic inputs in this region.

References

[1] W. H. Schlesinger, "Carbon sequestration in soils," *Science*, vol. 284, no. 5423, p. 2095, 1999.

[2] D. N. Borthakur, *Agriculture of the North Eastern Region with Special Reference to Hill Agriculture*, Beecee Prakashan, Guwahati, India, 1992.

[3] Anonymous, *Wastelands Atlas of India*, Ministry of Rural Development, Government of India and National Remote Sensing Agency, Hyderabad, India, 2000.

[4] Anonymous, *State of Forest Report*, Forest Survey of India. Ministry of Environment and Forests, Government of India, 2001.

[5] S. C. Ram and P. S. Ramakrishnan, "Hydrology and soil fertility of degraded grasslands at Cherrapunji in North Eastern India," *Environment Conservation*, vol. 15, pp. 29–35, 1988.

[6] M. Kellman, K. Miyanishi, and P. Hiebert, "Nutrient retention by savanna ecosystems II. Retention after fire," *Journal of Ecology*, vol. 73, no. 3, pp. 953–962, 1985.

[7] I. F. Ahlgren and C. E. Ahlgren, "Effect of prescribed burning on soil microorganisms in a Minnesota jack pine forest," *Ecology*, vol. 46, no. 3, pp. 304–310, 1965.

[8] G. Griffith, "Fertility problems in Uganda," Technical Communication, Commonwealth Bureau of Soil Science, vol. 46, pp. 160–164, 1949.

[9] M. Ram and B. P. Singh, "Soil fertility management in farming systems," Lectures notes, off campus training on farming system, Aizawl, India, 1993.

[10] A. Singh and M. D. Singh, "Effect of various stages of shifting cultivation on soil erosion from steep hill slopes," *Indian Forester*, vol. 106, no. 2, pp. 115–121, 1981.

[11] D. Mandal, K. S. Dadhwal, O. P. S. Khola, and B. L. Dhyani, "Adjusted T values for conservation planning in Northwest Himalayas of India," *Journal of Soil and Water Conservation*, vol. 61, no. 6, pp. 391–397, 2006.

[12] U. C. Sharma, "Methods of selecting suitable land use system with reference to shifting cultivation in NEH region," *Indian Journal of Soil Conservation*, vol. 26, no. 3, pp. 234–238, 1998.

[13] N. P. Singh, O. P. Singh, and N. S. Jamir, *Sustainable Agriculture Development Strategy for North Eastern Hill Region of India*, Mittal, New Delhi, India, 1996, Edited by Shukla S. P. and Sharma N.

[14] B. P. Bhatt, "Agroforestry for sustainable mountain development in N.E.H. region," in *Central Himalaya Environment and Development: Potentials, Actions and Challenges*, M. S. S. Rawat, Ed., pp. 206–223, Transmedia Publisher, Uttaranchal, India, 2003.

[15] Umashankar, "Indigenous agroforestry tree species for conservation and rural livelihood," in *Agroforestry in North East India: Opportunities and Challenges*, B. P. Bhatt and K. M. Bujarbaruah, Eds., pp. 149–174, ICAR Research Complex for NEH Region, Umiam, Meghalaya, 2005.

[16] R. Saha, J. M. S. Tomar, and P. K. Ghosh, "Evaluation and selection of multipurpose tree for improving soil hydro-physical behaviour under hilly eco-system of north east India," *Agroforestry Systems*, vol. 69, no. 3, pp. 239–247, 2007.

[17] B. P. Singh and S. K. Dhyani, "Significance of jackfruit in restoration of soil fertility," Annual Report, ICAR Research Complex, Umiam, Meghalaya, 1995.

[18] B. Majumdar, K. Kumar, M. S. Venkatesh, Patiram, and Bhatt B. P., "Effect of different agroforestry systems on soil properties in acid Alfisols of Meghalaya," *Journal Hill Research*, vol. 17, no. 1, pp. 1–5, 2004.

[19] B. Majumdar, M. S. Venkatesh, K. K. Satapathy, K. Kumar, and Patiram, "Effect of alternative farming systems to shifting cultivation on soil fertility," *Indian Journal of Agricultural Sciences*, vol. 72, no. 2, pp. 122–124, 2002.

[20] R. Saha and V. K. Mishra, "Long-term effect of various land use systems on physical properties of silty clay loam soil of N-E hills," *Journal of the Indian Society of Soil Science*, vol. 55, no. 2, pp. 112–118, 2007.

[21] K. K. Satapathy, "Runoff production on hill slopes under different land use systems," in *Agroforestry in North East India: Opportunities and Challenges*, B. P. Bhatt and K. M. Bujarbaruah, Eds., pp. 451–459, ICAR Research Complex for NEH Region, Umiam, Meghalaya, 2005.

[22] R. Saha, V. K. Mishra, and J. M. S. Tomar, "Effect of agroforestry systems on erodibility and hydraulic properties of Alfisols in eastern Himalayan region," *Indian Journal of Soil Conservation*, vol. 33, pp. 251–253, 2005.

[23] R. Saha, P. K. Ghosh, V. K. Mishra, B. Majumdar, and J. M. S. Tomar, "Can agroforestry be a resource conservation tool to maintain soil health in the fragile ecosystem of north-east India?" *Outlook on Agriculture*, vol. 39, no. 3, pp. 191–196, 2010.

[24] M. Datta and N. P. Singh, "Growth characteristics of multipurpose tree species, crop productivity and soil properties in agroforestry systems under subtropical humid climate in India," *Journal of Forestry Research*, vol. 18, no. 4, pp. 261–270, 2007.

[25] S. L. Swamy, S. Puri, and A. K. Singh, "Growth, biomass, carbon storage and nutrient distribution in *Gmelina arborea* Roxb. Stands on red lateritic soils in central India," *Bioresource Technology*, vol. 90, no. 2, pp. 109–126, 2003.

[26] P. K. Ghosh, A. Das, R. Saha et al., "Conservation agriculture towards achieving food security in North East India," *Current Science*, vol. 99, no. 7, pp. 915–921, 2010.

[27] R. Saha and V. K. Mishra, "Effect of organic residue management on soil hydro-physical characteristics and rice yield in eastern Himalayan region, India," *Journal of Sustainable Agriculture*, vol. 33, no. 2, pp. 161–176, 2009.

[28] V. K. Mishra, R. Saha, and K. M. Bujarbaruah, "Zero tillage technique for transplanted rice in high rainfall eco-system," *Scientific leaflet.*, ICAR Research Complex for NEH Region, Umiam, Meghlaya, 2005.

[29] R. Saha, V. K. Mishra, B. Majumdar, K. Laxminarayana, and P. K. Ghosh, "Effect of integrated nutrient management on soil physical properties and crop productivity under a Maize (*Zea mays*)-mustard (*Brassica campestris*) cropping sequence in acidic soils of Northeast India," *Communications in Soil Science and Plant Analysis*, vol. 41, no. 18, pp. 2187–2200, 2010.

[30] P. K. Ghosh, R. Saha, J. J. Gupta et al., "Long-term effect of pastures on soil quality in acid soil of north-east India," *Australian Journal of Soil Research*, vol. 47, no. 4, pp. 372–379, 2009.

[31] K. Laxminarayana, B. P. Bhatt, and R. Tulsi, "Soil fertility buildup through hedgerow intercropping in integrated farming system: a case study," in *Agroforestry in North East India: Opportunities and Challenges*, B. P. Bhatt and K. M. Bujarbaruah, Eds., pp. 495–506, ICAR Research Complex for NEH Region, Umiam, Meghalaya, 2005.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 673926, 10 pages
doi:10.1155/2012/673926

Review Article

Soil Degradation-Induced Decline in Productivity of Sub-Saharan African Soils: The Prospects of Looking Downwards the Lowlands with the *Sawah* Ecotechnology

Sunday E. Obalum,[1,2] Mohammed M. Buri,[3] John C. Nwite,[4] Hermansah,[5] Yoshinori Watanabe,[1] Charles A. Igwe,[1,2] and Toshiyuki Wakatsuki[1]

[1] *School of Agriculture, Kinki University, Nara 631-8505, Japan*
[2] *Department of Soil Science, University of Nigeria, Nsukka 410001, Nigeria*
[3] *CSIR—Soil Research Institute, Academy Post Office, Kwadaso, Kumasi, Ghana*
[4] *Department of Crop Production Technology, Federal College of Agriculture, P.M.B. 7008, Ishiagu, Nigeria*
[5] *Department of Soil Science, Faculty of Agriculture, Andalas University, Limau Manis, Padang 25163, Indonesia*

Correspondence should be addressed to Sunday E. Obalum, ijewelle@yahoo.com

Received 4 November 2011; Revised 13 January 2012; Accepted 16 January 2012

Academic Editor: Rosario García Moreno

The paper provides an insight into the problem of land degradation in Sub-Saharan Africa, with emphasis on soil erosion and its effect on soil quality and productivity, and proposes a lowland-based rice-production technology for coping with the situation. Crop yields are, in addition to the degree of past and current erosion, determined by a number of interacting variables. This, coupled with the generally weak database on erosion-induced losses in crop yield in spite of the region's high vulnerability to erosion, makes it difficult to attain a reliable inference on the cause-effect relationship between soil loss and productivity. Available data suggest, however, that the region is at risk of not meeting up with the challenges of agriculture in this 21st century. Based on the few studies reviewed, methodology appears to have an overwhelming influence on the erosion-productivity response, whereas issues bordering on physical environment and soil affect the shape of the response curve. We argue that the *sawah* ecotechnology has the potential of countering the negative agronomic and environmental impacts of land degradation in Sub-Saharan Africa. This is a farmer-oriented, low-cost system of managing soil, water, and nutrient resources for enhancing lowland rice productivity and realizing Green Revolution in the region.

1. Introduction

Ever since mankind started agriculture, soil erosion has been the single largest threat to soil productivity and has remained so till date [1]. This is so because removal of the topsoil by any means has, through research and historical evidence, been severally shown to have many deleterious effects on the productive capacity of the soil as well as on ecological wellbeing. Doran and Parkin [2] captioned the impact of soil erosion in their popular maxim that "the thin layer of soil covering the earth's surface represents the difference between survival and extinction for most terrestrial life." Although fertile topsoils could be lost when scraped by heavy machineries [3], the key avenues of topsoil loss include water erosion and wind erosion. Sometimes erosion can be such gradual for so long a time as to elude detection in one's lifetime, thus making its adverse effects hard to detect. Eswaran et al. [4] propose an annual loss of 75 billion tons of soil on a global basis which costs the world about US $400 billion per year. A review of the global agronomic impact of soil erosion identifies two severity groups of continents and reveals that Africa belongs to the more vulnerable group [5].

Soil erosion by water seems to be the greatest factor limiting soil productivity and impeding agricultural enterprise in the entire humid tropical region [6]. This is evident in many regions of Africa [7], mainly in the humid and

subhumid zones of Sub-Saharan Africa (SSA) where population pressure and deforestation exacerbate the situation and the rains come as torrential downpours, with the annual soil loss put at over 50 tons ha^{-1} [8]. In SSA, the problem is not limited to water erosion as wind erosion prevails mainly in the semiarid and arid zones. For instance, soil loss to wind erosion of 58–80 tons ha^{-1} has recently been reported from the West African Sahel [9]. Both forms of erosion can thus aptly define land degradation in the region. Soil erosion selectively detaches the colloidal fractions of soils and carts them away in runoff [10, 11]. These soil colloidal fractions (clay and humus) are needed for soil fertility, aggregation, structural stability, and favourable pore size distribution. The concentration of humus is usually higher in topsoils while that of clay is usually higher in subsoils due to illuviation, and this is mostly true in Ultisols that are widespread in Africa. This implies that humus, which has much greater capacity to hold water and nutrient ions compared to clay, its inorganic counterpart [12], is the more easily eroded.

In spite of the fact that the problem of land degradation is particularly severe in SSA, only little reliable data were available by the end of the 20th century both on its extent [8, 13] and on the cause-effect relationship between soil erosion and soil productivity [4, 14]. Thereafter, no significant research progress has been made to beef up the data in the region. We review in this paper the little available data, with a focus on soil properties modified by erosion and the extent of erosion-induced decline in the yield of commonly grown crops, which is viewed as a proxy for soil productivity. The survey highlights the enormous rate of soil erosion and the attendant decline in the productivity of agricultural soils in SSA. It is therefore unsurprising that, in the face of the advances so far made in biotechnology, agricultural productivity in SSA stagnates and remains perennially low as evident in hunger and poverty levels in the entire region [15, 16].

All the adverse impacts on agronomic productivity and environmental quality are respectively due to a decline in land quality and deposition of sediments and have been designated on-site effect and off-site effect, respectively [4, 11]. It is widely believed that erosion-induced deposition of sediments occurs in response to topographic gradients and that, since water does not climb hills in agricultural watersheds, the process is hardly reversible. With this in view, we make a case for tackling the agroecological problem of soil erosion in the diverse watersheds of SSA offsite rather than onsite. This is a case for the *sawah* ecotechnology, an Asian-type system of rice (*Oryza sativa* L.) production that has been adapted in the abundant lowlands in the region. The system can compensate for the loss of upland soil productivity while counteracting the environmental degradation due to soil erosion. It is viewed as the promising option to boosting rice production on a sustainable basis for the realization of the much-awaited Green Revolution in SSA.

2. Soil Loss and Crops Yields in Sub-Saharan Africa: A Survey of the Literature

2.1. Indices of Soil Productivity Affected by Soil Loss. Soil productivity is the capacity of a soil to produce a certain yield of crops or other plants under a defined set of management practices [17]. Thus comparison of soil productivity losses to erosion should be done for similar soil and crop management scenarios. Soil productivity entails striking a balance among soil "physical," "chemical," and "biological" fertilities, as none is of much value without others. All these soil properties are affected by topsoil removal; crop yields are affected through the resulting changes in these soil properties. Some of the ways by which soil erosion reduces its productivity include removal of plant nutrients in the eroded sediments, exposure of root-toxic and poorly aerated subsoils, P tie-up in illuviated clay which makes it apparently the most deficient nutrient in eroded soils, soil structure deformation leading to surface sealing and crusting which reduce seedling emergence and infiltration, and nonuniform removal of soil within a field which complicates the task of managing the soil to maximize production [14, 18].

Soil erosion or simulation of topsoil loss has been severally reported to adversely influence such soil physical properties as root zone depth, gravel content, particle size distribution, strength, bulk density, porosity, aggregate stability, moisture retention capacity, moisture characteristics, saturated hydraulic conductivities, and infiltration rates in SSA [3, 19–29]. The presence of organic matter in the surface soil generally promotes aggregation and may engender a situation where moisture-retaining pores are preponderant in soil. Soil erosion reduces its productivity primarily through the loss of plant available water capacity. Three months after the artificial removal of the top (5 cm) soil at three locations in southern Nigeria, Mbagwu et al. [23] observed reductions in moisture retention capacity and saturated hydraulic conductivities of the exposed soil layer, which were greater in Ultisols than in Alfisols. Mbagwu and Lal [30] later reported that limited moisture more than increased compaction caused greater reduction in root growth and dry matter of maize (*Zea mays* L.) and cowpea (*Vigna unguiculata* L.) in those locations.

Soil chemical properties that are mostly adversely influenced by erosion or topsoil removal in SSA include pH, organic matter content, total N, available P, exchangeable bases, and cation exchange capacity [3, 21, 24–26, 28, 29, 31]. In an Alfisol in southwestern Nigeria, Lal [32] reported that the enrichment ratio (ER, the concentration of plant nutrients in eroded soil materials to that in residual soil) was 2.4 for organic matter, 1.6 for total N, 5.8 for available P, 1.7 for exchangeable K, 1.5 for exchangeable Ca, and 1.2 for exchangeable Mg. For another Alfisol in Central Kenya recording an annual soil loss of above 60 tons ha^{-1}, the corresponding values of the ER were 2.1, 1.2, 3.2, 1.5, 1.2, and 1.0, respectively [33].

2.2. The Nature and Magnitude of Erosion-Induced Yield Decline in Sub-Saharan Africa. Although topsoil loss generally has adverse effects on productivity of soils, there can sometimes be an artifact in which case the loss improves soil productivity or at least does not affect it adversely [34]. This is often as a result of exposure of the surface of a previously buried productive soil following erosion [35].

Such a situation can be found in some deep Andisols and Inceptisols [26], but hardly occurs in the relatively shallow Alfisols, Ultisols, and Oxisols predominant in the tropics, in which nutrients are concentrated in the surface layer [36]. We are thus primarily concerned with the negative impact of soil erosion on soil productivity, which is the more commonly reported observation in SSA. The adverse impacts of soil erosion on agronomic productivity might be of short term or long term (Figure 1).

Virtually all the short-term effects stem from a reduction in the thickness of surface layers and a selective reduction in the components of such layers that are essential for crop production. Long-term effects stem from the ensuing progressive reduction in the rooting zone depth.

As a first-hand appreciation of the peculiarity of erosion-induced degradation in SSA, no portion of only about 3% of the global land surface considered as prime or class 1 falls into the tropical region [4], to which belongs SSA and which accounts for about 39% of the world's land surface [37]. In the humid and subhumid zones of West Africa, deforestation proceeds at a rate of about 4 million ha per year, with deforestation to reforestation ratio of 30 : 1 on the average [8]. However, information on the extent and severity of natural and anthropogenic soil erosion and on the quantitative cause-effect relationships between soil loss and productivity of agricultural lands prone to erosion in SSA is generally lacking or, where available, is weak, subjective, and unreliable. This situation has been attributed to the difficulty in conducting the long-term, concentrated interdisciplinary research (including financial/time constraints) which is needed to overcome the complexity posed by annual and seasonal variations in number and magnitude of erosion, the multifactorial nature of yield factors, as well as the belief that inorganic fertilizers are all-ameliorating [4, 14, 19, 35, 38]. However, available data to date suggest a severity of erosion hazards in many agroecological zones of the SSA, with cases of advanced gullies in some of the zones (Figure 2) [39].

Dregne [7] reported that irreversible soil productivity losses from water erosion appeared to be serious on a national scale in Algeria, Morocco, and Tunisia in North Africa; in Ethiopia, Kenya, and Uganda in East Africa; in Nigeria and northern Ghana in West Africa; and in Lesotho, Swaziland, and Zimbabwe in southern Africa. He observed as much as 50% productivity loss to wind erosion in part of Tunisia, and delineated areas in Africa where about 20% permanent reduction on crop productivity have resulted from human-induced water and wind erosion. Lal [14] estimated that past erosion in Africa has caused yield reduction of 2–40%, and that if present trend continues, the yield reduction by 2020 may be 16.5%.

2.3. Selected Cases of Assessed Impact of Soil Loss in Sub-Saharan Africa

2.3.1. Desurfacing Experiments. In spite of the weak points of desurfacing experiments, most studies on erosion-induced decline in soil productivity in the tropics were done on artificially-desurfaced soils in order to close the information gap on soil loss and crop productivity relationship in the region [24]. The method is favoured in this region also because of the difficulty of separating the effect of past erosion from that of the present erosion vis-à-vis the rather few examples on the assessment of the impact of current rate of erosion on crop yield [11]. Selected trials based on topsoil desurfacing in SSA are summarized in Table 1. As a further hint to the data shown, it was reported in one of these trials that the relationship between the grain yield of maize, Y_a and Y_b (tons ha^{-1}) in the first and second year respectively, and the depth of topsoil desurfaced, x (cm), was of the exponential form [27]:

$$\begin{aligned} Y_a &= 3.2761e^{-0.1621x} \quad (R^2 = 0.998), \\ Y_b &= 1.6116e^{-0.1489x} \quad (R^2 = 0.985). \end{aligned} \tag{1}$$

2.3.2. Natural Soil Erosion. Studies on natural soil erosion are relatively few in SSA because such trials are conducted on runoff plots which are limited in number in the region. Moreover, such studies do not give rapid results since erosion is a gradual process such that noticeable differences in crop yield may take a long time to be established. The attraction for results emanating from this method, however, is that they reflect what happens in the field under natural conditions and so give the most realistic and reliable results. Few studies based on natural soil erosion are summarized in Table 2.

Lal [21] studied the effect of accumulative soil erosion for a 5-year period on the yields of maize and cowpea in Alfisols and reported that the reductions in their yields were, respectively, 9.0 and 0.7 kg ton^{-1} of soil loss. He also obtained the following linear relationships between yield, Y, in tons ha^{-1} and soil erosion, E, in tons ha^{-1}:

$$\begin{aligned} Y_{\text{maize}} &= 5.95 - 0.009E, \quad r = -0.87^*, \\ Y_{\text{cowpea}} &= 0.407 - 0.0007E, \quad r = -0.66^*. \end{aligned} \tag{2}$$

It was reported from Tanzania that reductions in maize yields due to severe past erosion of soils ranged from 15 to 48% [11]. From runoff plots located on a sandy loam Ultisol in Kumasi, Ghana, subjected to four different tillage practices, Adama and Quansah [41] reported that the grain yield of maize, Y, in kg ha^{-1} in the major season and cumulative soil loss, E, in tons ha^{-1} in the same season plus that in the previous year were related thus:

$$Y = 2686 - 13.92E, \quad r = -0.94^*. \tag{3}$$

2.3.3. Greenhouse Experiments. Under greenhouse conditions, the yield of maize was found to be 20–50% (with a mean of 40%) higher on surface soil than on subsurface soil, the latter of which showed to be deficient in N and P [42]. Mbagwu [24] reported that without any amendment, maize and cowpea yields were, respectively, reduced by 58 and 19% on soils from runoff plots established 12 years earlier on an Ultisol in southeastern Nigeria, with a soil loss rate of 55 tons yr^{-1}. With the addition of brewers' grains to the eroded soil under both crops, however, maize and cowpea showed lower yield reductions of 22 and 9%, respectively.

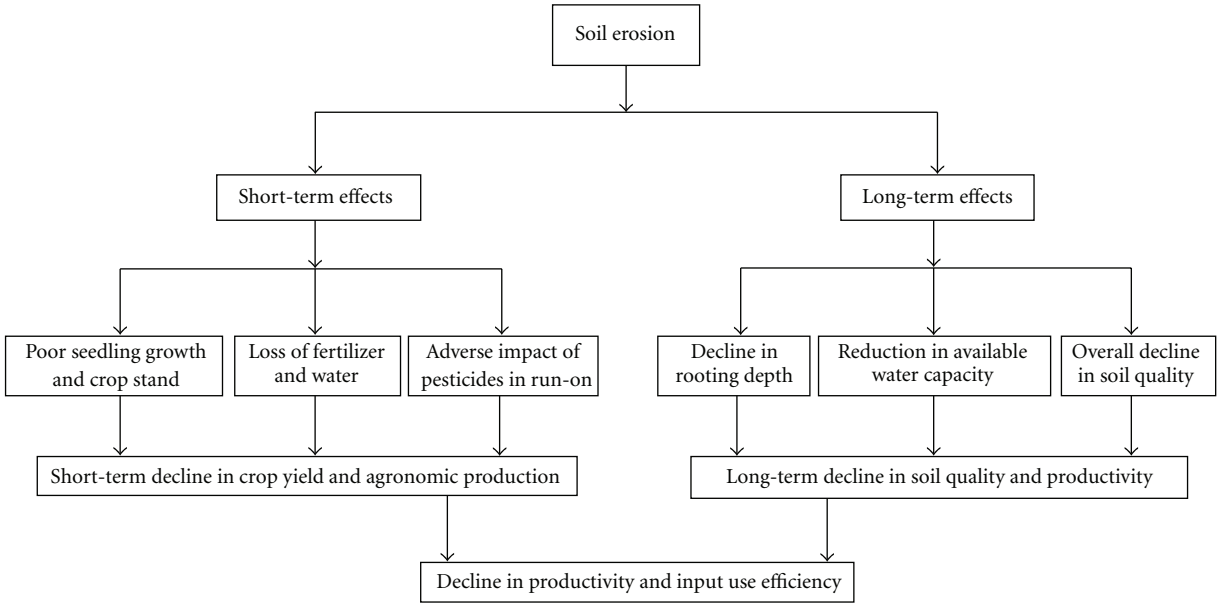


Figure 1: On-site effects of soil erosion on productivity decline (source: Lal et al. [11]).

Figure 2: A gullied farmland in southeastern Nigeria, after Igwe [39].

In a separate study, Mbagwu [36] reported that the topsoils outyielded the subsoils by a range of 18–40% on two Alfisols, two Ultisols, and one Inceptisol in southern Nigeria.

From the information for the desurfacing studies (Table 1), there appears to be a convex relationship between soil loss and productivity, that is, increasing productivity loss with increasing soil loss. The data also reveal that yield losses to soil erosion are more severe on Ultisols than Alfisols, thus implying that Ultisols have lower T values than Alfisols. This is attributed to the generally lower inherent fertility status of Ultisols than Alfisols [12, 40]. Yield reductions are also consistently lower for cowpea than for maize; irrespective of method of achieving soil loss, of soil order, and of location. This has been attributed to the ability of cowpea to nodulate, which maize could not do [40]. Notably, as the erosion severity increases, the percent reduction in the yield of cassava (*Manihot esculentum* C.) increases, which is not the case with the other crops. The explanation lies in the fact that cassava is a deep-feeder crop, unlike cereals and legumes which are relatively shallow feeders.

Furthermore, the comparison of the data in Table 1 with those in Table 2 reveals that yield reduction per centimetre of soil loss is always higher on naturally eroded soils than in soils from where equivalent soil depths have been desurfaced. This could be due partly to the fact that rains compact the soil whereas desurfacing does not. On two adjacent plots, Lal [14] reported that the decline in maize yield by natural erosion was about 16 times more than that by desurfacing. However, the topsoil is never uniformly removed in one growing season by natural erosion as does desurfacing. Therefore, within the same time scale, the sudden and total disappearance of topsoil due to desurfacing would be expected to result in much stronger changes in soil properties than with natural soil erosion, such that the negative effect of erosion on soil productivity may be exaggerated [43]. And that is the reason why den Biggelaar et al. [5] view studies on present erosion as mimicking inappropriate soil management practices and their adverse effects. The data in Tables 1 and 2 thus support the view that erosion-productivity relationships generated by different methods are hard to compare [4, 43].

3. Sustaining Soil Productivity against Land Degradation in Sub-Saharan Africa

Using the study by Oyedele and Aina [25] in southwestern Nigeria as a reference point, soil chemical properties can account for over 75% of the variation in the yield of cereals from eroded soils in SSA. Thus, erosion-induced short-term decline in productivity is more easily compensated by

Table 1: Erosion-productivity relationship for soils of Sub-Saharan Africa (desurfacing experiments).

Soil loss (cm)	Yield reduction (%)	Soil order	Climate/location	Country	Source
Maize (*Zea mays* L.) as a test crop					
2.5, 5, 7.5, 10, 12.5	23, 38, 49, 53, 56	Alfisol	Subhumid Ibadan	Nigeria	[19]
5, 10, 20	72.5, 82.6, 99.5	Alfisol	Subhumid Ilora	Nigeria	[40]
5, 10, 20	30.5, 73.6, 93.5	Alfisol	Subhumid Ikenne	Nigeria	[40]
5, 10, 20	95.4, 95.4, 100	Ultisol	Humid Onne	Nigeria	[40]
5	54.9	Alfisol	Subhumid Ilora	Nigeria	[36]
5	30	Alfisol	Subhumid Ikenne	Nigeria	[36]
5	15	Inceptisol	Subhumid Nsukka	Nigeria	[36]
5	69.7	Ultisol	Humid Onne	Nigeria	[36]
5	64.2	Ultisol	Subhumid Nsukka	Nigeria	[36]
10, 20	39.2, 81.7	Alfisol	Subhumid Ibadan	Nigeria	[14]
2.5, 7.5	50, $\gg$100	Ultisol	Humid Douala	Cameroon	[14]
5, 10, 20	47, 48, 63	Lateritic Alfisol	Semiarid Ouagadougou	Burkina Faso	[14]
3, 6	23, 55	Ultisol	Subhumid Nsukka (1)	Nigeria	[3]
3, 6	50, 95	Ultisol	Subhumid Nsukka (2)	Nigeria	[3]
5, 10, 15, 20	56.0, 82.5, 90.0, 95.5	Oxisol	Subhumid Ile-Ife	Nigeria	[27]
15, 25	17, 67 (upper slope); 65, 76 (lower slope)	Gravelly Alfisol	Subhumid Ibadan	Nigeria	[29]
Cowpea (*Vigna unguiculata* L.) as a test crop					
5, 10, 20	42.6, 33.1, 80.5	Alfisol	Subhumid Ilora	Nigeria	[40]
5, 10, 20	1.5, 59.1, 65.1	Alfisol	Subhumid Ikenne	Nigeria	[40]
5, 10, 20	62.0, 70.6, 68.3	Ultisol	Humid Onne	Nigeria	[40]
Cassava (*Manihot esculentus* C.) as a test crop					
10, 20	35.7, 53.7	Alfisol	Subhumid Ibadan	Nigeria	[40]

Quantification was achieved where both the depth of soil loss and the yield reduction were given by the authors or could be calculated from the information they presented.

Table 2: Erosion-productivity relationship for soils of Sub-Saharan Africa (natural erosion).

Soil loss (cm)	Yield reduction (%)	Soil order	Climate/location	Country	Source
Maize (*Zea mays* L.) as a test crop					
0.0024	26.9	Alfisol	Semiarid Harare	Zimbabwe	[14]
0.0080	0.1513	Alfisol	Subhumid Ibadan	Nigeria	[21]
0.0080	0.1720	Alfisol	Subhumid Ibadan	Nigeria	[21]
Pearl millet (*Pennisetum americanum* L.) as a test crop					
0.0928	51.6	Aridisol	Semiarid Niangoloko	Burkina Faso	[14]

All soil erosion rates were converted to equivalent depths of soil loss, assuming a bulk density of 1.25 mg m^{-3}.

inorganic and/or organic fertilization and supplemental irrigation, as opposed to long-term decline in productivity [11]. However, the efficiency of inorganic fertilizer in an eroded soil where the physical properties are degraded alongside chemical nutrients depletion depends, to a large extent, on the dynamic relationship between the level of harm done to the soil's physical condition and the level of progress made in the difficult task of improving it [35, 44, 45]. Such a situation needs a combination of carefully selected, suitable management practices depending on the shape of the yield reduction function. In Nigeria, for instance, research evidence from eroded Alfisols suggests that, rather than inorganic fertilization, application of poultry manure and fallowing to various grass and leguminous species for two years could improve the soil physicochemical properties and productivity [29, 46].

The situation in SSA calls for more sustainable farming systems and underscores the need to look beyond the use of inorganic fertilizers as a means of restoring the productivity of naturally eroded soils in the region. Except in the case of gullies where urgent intervention may be needed, incorporation of cover cropping into our agronomic systems can help to conserve "yet-to-be-degraded" soils against degradation while forestalling further erosion from already "degraded" upland soils [33]. Such a soil management practice allows eroded soils the chance to restore the loss in productivity at a rate commensurate with their resilience. For some time now, however, the question has been on

how to accommodate better the problem of soil erosion in SSA as part of livelihood strategies [13]. We propose in this paper that it would be more profitable to focus greater efforts on developing our huge lowland resources with the *sawah* ecotechnology. The *sawah* system is based on the concept of watershed development and, so, is an adaptation of the Japanese "Satoyama" system to African environments. Figure 3 is an example of African "Satoyama" concept, which is a watershed agroforestry applicable to cocoa belt region in West Africa.

Sawah refers to a lowland field that is demarcated using earthen bunds, puddled and leveled using a hand-operated power tiller, transplanted to a high-yielding rice variety in rows, and kept under regulated submergence throughout the growing season (Figure 4). Thus unlike the traditional lowland rice field that is a diverse and mixed-up environment, the lowland *sawah* system is a diverse and intensified rice-growing environment that is characterized by well-designed and well-demarcated field condition with clearly defined management of soil, water, and nutrient resources. The term *sawah* is of Malayo-Indonesian origin but has been adopted in SSA as corresponding to paddy fields in Asia. The adoption became necessary in order to differentiate the technology from unprocessed rice grain, upland rice field, or traditional lowland rice field (all of which are regularly referred to as paddy in SSA). It is hoped that the clearing of these terminological uncertainties would foster the sharing of ideas and strategies among all the stakeholders in rice production [16].

4. Why the Lowland *Sawah* Ecotechnology?

There is no gainsaying that food production in SSA needs to transit for its present level to the next level in terms of simultaneously increasing the output and conserving the natural environments. One of the ways of achieving this task is to work towards modifying the offsite effect of erosion, such that rather than compromising environmental quality, eroded sediments that eventually get deposited in the lowlands can be harnessed to contribute to agricultural production and environmental quality using such an appropriate technology as the lowland *sawah* systems. Because of the significant contribution of this sediment deposition process (otherwise known as geologic fertilization) to the fertility of lowland soils of SSA [48], the case for the *sawah* ecotechnology is clearly that of diverting attention from onsite to offsite as a means of coping with the problem of soil erosion.

In the first place, out of the about 2.4 billion ha of land in SSA, lowlands comprise about 250 million ha [49]. This implies that lowlands occupy above 10% of the region's land mass. The majority of the lowlands have huge potential for increasing agricultural production in SSA, yet many of them remain unexploited and most others grossly underutilized [50]. In his essay, "African Green Revolution needn't be a mirage," Ejeta [15] noted that in Africa where the culture of looking up to science for solutions to local problems is not well established, the people can realize Green Revolution with locally developed and locally relevant technologies. We can thus rhetorically "look downwards to a lowland technology" as an alternative to our quest for a sustainable agricultural production system in Africa. The people are increasingly conscious of this option. Consequently, gone are the days before the mid 1990s when there was a greater emphasis on growing rice in upland agricultural soils than in the lowland ecosystems under rainfed conditions [16, 51]. In West Africa that leads the rest of SSA in rice production, for instance, the ratio of uplands to lowlands in terms of area under rice is 10.00 : 6.13, and this ratio is rapidly decreasing [52].

Similar to their attitude of not looking up to science for solutions to local problems, African farmers tend to be alienated from any science-oriented agricultural production system that is not rooted in their farming culture and to which their indigenous knowledge does not make any contribution. To buttress this point, the peoples' shift of preference from upland to lowland farming has been identified as one of the reasons for the failure of agroforestry to achieve the success expected of it at the onset [51]. This may not be the case with the *sawah* ecotechnology in the lowlands where rice has been a traditional crop in Africa. Instead, the farmers in the region view the technology as that which is taking them from what they already know to how they can do it better. Apart from being agroecosystems that the farmers are familiar with, lowlands denote agroecologies of low elevation and so mostly offer favourable hydrological conditions for the rice crop. Particularly in the Equatorial Forest and the Guinea Savanna Zones, precipitation and lateral groundwater flow from the adjacent uplands cause the lower footslopes and valley bottoms to be saturated or flooded for a certain period, thereby ensuring a potentially long cropping period that permits either double rice cropping or cultivation of vegetables and root crops after rice [49].

Moreover, sediments from such runoffs can engender favourable soil hydrophysical status for *sawah*-managed rice, and this is usually most evident in the extreme valley bottoms [53]. There is thus more to the aforementioned geological fertilization. Such a natural mechanism of replenishment of soil "physical" and "chemical" fertility can be imagined from Figure 3. The aspect of enriching *sawah* system with plant nutrients is particularly cherished because of the inherently low-fertility status of the lowland soils in SSA [54] vis-à-vis the relatively low level of fertilizer use by SSA farmers [55]. Owing to the topographic position of the lowlands and to the ecological engineering works that go with *sawah* systems design, erosion is reduced to almost zero in these ecologies with the *sawah* ecotechnology. This, among other benefits, assures that the topsoil that is characterized by low bulk density especially early in the season (due to the puddling exercise) is not washed away, thus sparing the nutrient-rich sediments transported from the uplands. The technology is therefore very effective for conserving soil, water, nutrients, and the overall environment.

An earlier proposal for rice farming in West Africa is that uplands should be cultivated with short-to-long fallow periods, whereas large inland valleys, coastal plains,

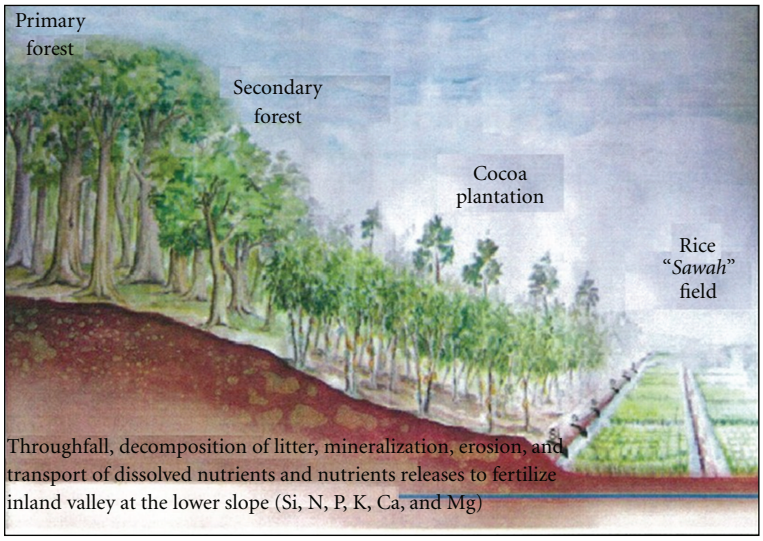


Figure 3: A typical example of African SATO-YAMA Concept developed by the Forest Research Institute of Ghana, after Owusu-Sekyere et al. [47].

Figure 4: A newly developed *sawah* field located in an inland valley in Jega, Kebbi State of Nigeria.

and floodplains should be cultivated more intensively [49]. However, the existing research concept to improve natural resource management in SSA may not bring about the desired results among the lowland rice farmers, unless there is a clearly defined research concept to improve soil and water conditions of the lowlands. Application of the three core Green Revolution technologies (high-yielding varieties, inorganic fertilizers, and irrigation facilities) outside the *sawah* system can even degrade the environment, such as that emanating from inefficient fertilizer use under situations of poor water management prevailing in non-*sawah* rice fields [48]. At the moment, the *sawah* ecotechnology appears to bring to an end the search for a farming system that addresses this issue in the region. So, for the advocacy for increased fertilizer use in Africa [55] to suitably apply to lowlands, *sawah* systems must first be put in place. The farmers themselves now know that the high-yielding varieties respond well to fertilizers only when they are grown under favourable soil and water conditions [16]. The *sawah* ecotechnology is therefore the only rice-farming system in the lowlands that can permit the proposed intensive cultivation of these rice ecologies on a sustainable basis, that is, without compromising high yields and environmental quality [48].

The *sawah* ecotechnology in the lowlands has a lot of prospects for coping with land degradation and ensuring sustainable agricultural production in SSA. Our 15-year and continuing trials in Ghana and Nigeria have demonstrated that the *sawah* system is the prerequisite for successfully applying the other Green Revolution technologies to realize lowland rice potential in SSA. The technology is farmer-friendly because the farmer is empowered to have absolute control and management of water in his field, which enables them to enjoy a flexible—and hence convenient—time table for the farming season. We hypothesize that if the farmer is placed at the centre of the creation of lowland *sawah* systems, field water control can be more effective and the struggle for a sustainable rice production system and a rice Green Revolution in SSA can be won. This is our *sawah* hypothesis I.

Furthermore, a properly managed *sawah* system has the potential of providing ecosystem services. This is mainly through enhanced C sequestration in forests and soils and the associated alleviatory effect on global warming problems [50]. The *sawah* system also neutralizes the soil pH thereby enhancing the availability of P and micronutrients in the soil. Such a condition of favourable soil nutrient status encourages the proliferation of a myriad of mostly anaerobic and photosynthetic microbes which, through a microbial nanowire collaborative network, constitute strong mechanisms for biological N fixation. In Asia, this phenomenon can result in annual values ranging from 20 to 200 kg N ha^{-1}, depending on the biophysical and the rice-growing environments [48]. The *sawah* system, thus, does not depend on only Azolla to sustain biological N in the soil.

Other benefits of the *sawah* system include favourable soil redox processes and suppression of weed growth due mainly to both the submerged soil condition and good tillering.

Above all, the mean grain yield of upland rice in West Africa is about 0.9 tons ha^{-1} [49]. To show that such low yields relate largely to the growing ecology and farming system, some scientists recently reported that the mean grain yield of the new rice cultivar for Africa (NERICA) from three locations in southern Benin was only 1.14 tons ha^{-1}, the fact that it was grown on previously fallowed uplands and with adequate fertilization notwithstanding [56]. On the other hand, rice grain yield under the *sawah* system ranges from 4.0 to 8.0 tons ha^{-1}, depending on the rice variety grown, external input level, water management, and other agronomic and management practices [57, 58]. On the average, therefore, the data just stated represent roughly between 4 and 8 times lower grain yield of rice under the upland growing systems than under the novel lowland *sawah* systems.

However, considering the fact that the upland system involves fallow periods which are not necessary under the *sawah* system, the yield gap between the two systems widens. At least 10 ha of upland is taken to be an equivalent of 1 ha of lowland *sawah* in terms of yield in a growing season. This is our *sawah* hypothesis II. In other words, each hectare of lowland *sawah* field enables the conservation of at least 10 ha of forest area. *Sawah* fields can thus foster both increased food production and forest conservation, which in turn enhances the sustainability of intensive lowland *sawah* systems by way of enhanced water conservation and supply of fertile topsoils through the geological fertilization. All this points to the sustainable nature of *sawah* systems compared to the upland rice culture which is mostly characterized by slash and burn, thereby degrading further our agroecological systems and environments.

5. Challenges of the *Sawah* Ecotechnology in Sub-Saharan Africa

Lowlands are particularly vulnerable to climate and environmental changes. For instance, the rise in sea level associated with contemporary global warming would, by modifying the coastal environments, ultimately affect the hydrological conditions of the lowlands. Hence, the lowlands are occasionally subject to such natural disasters as flooding. Multidisciplinary research is thus needed to reinforce the lowland *sawah* ecotechnology against such disasters. Closely related to this in the SSA environments is the need to empirically devise a means of coping with the possible adverse effect of the destabilization of soil structure by puddling. Granted that erosion is not a problem in lowland *sawah* soils, puddled soils may behave differently in the event of flood disasters if the soil structure does not regenerate properly. The off-season structural status of puddled lowland soils can also influence the performance of any crop grown after rice, thus stressing the need for a research on post-*sawah* crops that would maximize the use of the lowland soil resources in the region.

Furthermore, considering the importance of natural soil fertility replenishment as a way of minimizing inorganic fertilization and the associated reduction in economic returns, the extent of geological fertilization in different topographical and land-cover conditions needs to be quantified. Similarly, we only know of the extent of biological nitrogen fixation in Asian paddy fields, such is yet to be evaluated for the *sawah* systems in SSA with a different hydrophysical environment [50]. This is important, considering the low geological fertilization of the lowlands with respect to total N compared to available P [32, 33]. Finally, the *sawah* hypothesis II is yet to be validated in SSA environments. All this is needed to strengthen the case for the *sawah* systems as a means of simultaneously mitigating land degradation, ensuring sustainable rice production and promoting ecological wellbeing.

6. Perspectives

In most of the SSA, land degradation potentially undermines efforts towards sustainable agricultural production and so poses a major threat to the future of agriculture. Regrettably, available data to date on the quantitative relationship between soil loss and reductions in crop yield in the region are still fragmentary and grossly insufficient. The little available data, though characterized by a wide disparity, highlight the enormous loss of soil productivity to erosion in the region. The *sawah* ecotechnology for lowland rice production holds a lot of prospects. Although concentrated in the lowlands, well-managed *sawah* systems can help to conserve soil and water in the entire watershed. With the technology, SSA countries have the opportunity of achieving self-sufficiency in rice production while enhancing the quality of their environments. Although there are still areas needing long-term collaborative research in the adaptation of the *sawah* systems to SSA environments, we are so far convinced that proper application of the *sawah* ecotechnology at the rice farmer's field is a prerequisite for successfully applying other Green Revolution technologies.

Acknowledgments

The authors gratefully acknowledge the sponsorship of the Ministry of Education, Culture, Sports, Science and Technology (MEXT) of the Japanese Government through the Monbukagakusho Scholarship, the Japan Society for the Promotion of Science (JSPS), and New *Sawah* Project of the Kinki University of Japan.

References

[1] P. Sullivan, "Sustainable soil management: soil systems guide," Appropriate Technology Transfer for Rural Areas (ATTRA) Fayetteville AR 72702, National Center for Appropriate Technology (NCAT), 2004.

[2] J. W. Doran and T. B. Parkin, "Defining and assessing soil quality," in *Defining Soil Quality for a Sustainable Environment*, J. W. Doran et al., Ed., vol. 35, Soil Science Society of America Special Publication, Madison, Wis, USA, 1994.

[3] O. E. Ngwu, J. S. C. Mbagwu, and M. E. Obi, "Effect of desurfacing on soil properties and maize yield—research note," *Nigerian Journal of Soil Science*, vol. 15, no. 2, pp. 148–150, 2005.

[4] H. Eswaran, R. Lal, and P. F. Reich, "Land degradation: an overview," in *Proceedings of the 2nd International Conference on Land Degradation and Desertification*, pp. 1–5, Oxford Press, Khon Kaen, Thailand, 2001.

[5] C. den Biggelaar, R. Lal, K. Wiebe, and V. Breneman, "The global impact of soil erosion on productivity. I: absolute and relative erosion-induced yield losses," *Advances in Agronomy*, vol. 81, pp. 1–48, 2004.

[6] Board on Agriculture, *Sustainable Agriculture and the Environment in the Humid Tropics*, National Academic Press, 1993.

[7] H. E. Dregne, "Erosion and soil productivity in Africa," *Journal of Soil & Water Conservation*, vol. 45, no. 4, pp. 431–436, 1990.

[8] FAO, *Land and Environmental Degradation and Desertification in Africa*, FAO Corporate Document Repository, 1995.

[9] K. Ikazaki, H. Shinjo, U. Tanaka, S. Tobita, S. Funakawa, and T. Kosaki, "Field-scale aeolian sediment transport in the Sahel, West Africa," *Soil Science Society of America Journal*, vol. 75, pp. 1885–1897, 2011.

[10] USDA-NRCS, *Soil Quality Indicators: Organic Matter, Soil Quality Information Sheet*, The National Soil Survey Centre in Co-Operation with the Soil Quality Institute, NRCS and the National Soil Tilth Laboratory, ARS, 1996.

[11] R. Lal, C. den Biggelaar, and K. D. Wiebe, "Measuring on-site and off-site effects of erosion on productivity and environmental quality," in *Proceedings of the OECD Expert Meeting on Soil Erosion and Soil Biodiversity Indicators*, Rome, Italy, March 2003.

[12] I. E. Esu, *Fundamental of Pedology*, Stirling-Horden Publishers, Ibadan, Nigeria, 1999.

[13] A. Warren, S. Batterbury, and H. Osbahr, "Soil erosion in the West African Sahel: a review and an application of a "local political ecology" approach in South West Niger," *Global Environmental Change*, vol. 11, no. 1, pp. 79–95, 2001.

[14] R. Lal, "Erosion-crop productivity relationships for soils of Africa," *Soil Science Society of America Journal*, vol. 59, no. 3, pp. 661–667, 1995.

[15] G. Ejeta, "African green revolution needn't be a mirage," *Science*, vol. 327, no. 5967, pp. 831–832, 2010.

[16] S. S. Abe and T. Wakatsuki, "Sawah ecotechnology—a trigger for a rice green revolution in sub-Saharan Africa: basic concept and policy implications," *Outlook in Agriculture*, vol. 40, no. 3, pp. 221–227, 2011.

[17] Soil Science Society of America, *Glossary of Soil Science Terms*, American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Madison, Wis, USA, 2001.

[18] USDA SEA-AR, "Soil erosion effects on soil productivity: a research perspective," *Journal of Soil and Water Conservation*, vol. 36, no. 2, pp. 82–90, 1981.

[19] R. Lal, "Soil erosion on Alfisols in Western Nigeria. V. The changes in physical properties and the response of crops," *Geoderma*, vol. 16, no. 5, pp. 419–431, 1976.

[20] M. E. Obi and B. O. Asiegbu, "The physical properties of some eroded soils of southeastern Nigeria," *Soil Science*, vol. 130, no. 1, pp. 39–48, 1980.

[21] R. Lal, "Soil erosion problems on alfisols in Western Nigeria, VI. Effects of erosion on experimental plots," *Geoderma*, vol. 25, no. 3-4, pp. 215–230, 1981.

[22] M. E. Obi, "Runoff and soil loss from an oxisol in Southeastern Nigeria under various management practices," *Agricultural Water Management*, vol. 5, no. 3, pp. 193–203, 1982.

[23] J. S. C. Mbagwu, R. Lal, and T. W. Scott, "Effects of artificial desurfacing on Alfisols and Ultisols in southern Nigeria: II. Changes in soil physical properties," *Soil Science Society of America Journal*, vol. 48, no. 4, pp. 834–838, 1984.

[24] J. S. C. Mbagwu, "Physico-chemical properties and productivity of an Ultisol in Nigeria as affected by long-term erosion," *Pedologie*, vol. 38, no. 2, pp. 137–154, 1988.

[25] D. J. Oyedele and P. O. Aina, "A study of soil factors in relation to erosion and yield of maize on a Nigerian soil," *Soil and Tillage Research*, vol. 48, no. 1-2, pp. 115–125, 1998.

[26] P. I. Våje, B. R. Singh, and R. Lal, "Erosional effects on soil properties and maize yield on a volcanic ash soil in Kilimanjaro region, Tanzania," *Journal of Sustainable Agriculture*, vol. 12, no. 4, pp. 39–53, 1998.

[27] D. J. Oyedele and P. O. Aina, "Response of soil properties and maize yield to simulated erosion by artificial topsoil removal," *Plant and Soil*, vol. 284, no. 1-2, pp. 375–384, 2006.

[28] M. A. Anikwe, O. E. Ngwu, C. N. Mbah, C. E. Onoh, and E. E. Ude, "Effect of ground cover by different crops on soil loss and physicochemical properties of an Ultisol in South Eastern Nigeria," *Nigerian Journal of Soil Science*, vol. 17, pp. 94–97, 2007.

[29] F. K. Salako, P. O. Dada, C. O. Adejuyigbe et al., "Soil strength and maize yield after topsoil removal and application of nutrient amendments on a gravelly Alfisol toposequence," *Soil and Tillage Research*, vol. 94, no. 1, pp. 21–35, 2007.

[30] J. S. C. Mbagwu and R. Lal, "Effect of bulk density and irrigation frequency on root growth and dry matter yields of corn and cowpeas for three Nigerian topsoil and subsoil profiles," *Beitrage zur Tropischen Landwirtschaft und Veterinarmedizin*, vol. 23, pp. 277–285, 1985.

[31] F. B. S. Kaihura, I. K. Kullaya, M. Kilasara, J. B. Aune, B. R. Singh, and R. Lal, "Soil quality effects of accelerated erosion and management systems in three eco-regions of Tanzania," *Soil and Tillage Research*, vol. 53, no. 1, pp. 59–70, 1999.

[32] R. Lal, "Soil erosion on Alfisols in Western Nigeria. IV. Nutrient element losses in runoff and eroded sediments," *Geoderma*, vol. 16, no. 5, pp. 403–417, 1976.

[33] C. K. K. Gachene, N. Karanja, J. G. Mureithi, P. Khisa, and J. Maina, "Farmers' evaluation of legume cover crops for erosion control in Gathwariga catchment, Kenya," *International Journal of Agriculture and Rural Development*, vol. 5, pp. 176–186, 2004.

[34] M.G. Wolman, "Soil erosion and crop productivity: a worldwide perspective," in *Soil Erosion and Crop Productivity*, R. F. Follet and B. A. Stewarts, Eds., pp. 9–21, American Society of Agronomy, Madison, Wis, USA, 1985.

[35] L. D. Meyer, A. Bauer, and R. D. Heil, "Experimental approaches for quantifying the effect of soil erosion on productivity," in *Soil Erosion and Crop Productivity*, R. F. Follet and B. A. Stewarts, Eds., pp. 213–233, American Society of Agronomy, Madison, Wis, USA, 1985.

[36] J. S. C. Mbagwu, "Soil-loss tolerance of some Nigerian soils in relation to profile characteristics," *Turrialba*, vol. 41, no. 2, pp. 223–229, 1991.

[37] J. Bouma, N. H. Batjes, and J. J. R. Groot, "Exploring land quality effects on world food supply," *Geoderma*, vol. 86, no. 1-2, pp. 43–59, 1998.

[38] R. Lal, "Agronomic impact of soil degradation," in *Methodology for Assessment of Soil Degradation*, R. Lal, W. Blum, C. Valentine, and B. A. Stewart, Eds., pp. 459–473, CRC Press, Boca Raton, Fla, USA, 1997.

[39] C. A. Igwe, "Tropical soils, physical properties," in *Encyclopedia of Agrophysics*, J. Glinski, J. Horabik, and J. Lipiec, Eds., pp. 934–937, Springer, 1st edition, 2011.

[40] J. S. C. Mbagwu, R. Lal, and T. W. Scott, "Effects of desurfacing of Alfisols and Ultisols in southern Nigeria: I. Crop performance," *Soil Science Society of America Journal*, vol. 48, no. 4, pp. 828–833, 1984.

[41] A. I. Adama and C. Quansah, "Cumulative soil loss under different tillage practices and its effects on the growth and yield of maize in the semi-deciduous forest zone of Ghana," in *Proceedings of the African Crop Science Conference*, vol. 9, pp. 343–349, 2009.

[42] P. O. Aina and E. Egolum, "The effect of cattle feedlot manure and inorganic fertilizer on the improvement of subsoil productivity," *Soil Science*, vol. 129, no. 4, pp. 212–217, 1980.

[43] M. M. Bakker, G. Govers, and M. D. A. Rounsevell, "The crop productivity-erosion relationship: an analysis based on experimental work," *Catena*, vol. 57, no. 1, pp. 55–76, 2004.

[44] J. S. C. Mbagwu, "Effects of soil erosion on the productivity of agricultural lands in the humid tropics," *Beitrage zur Tropischen Landwirtschaft und Veterinarmedizin*, vol. 24, no. 2, pp. 161–175, 1986.

[45] O. E. Ngwu, J. S. C. Mbagwu, and M. E. Obi, "Effects of surface soil loss in southeastern Nigeria: 1. Crop performance," *Nigerian Journal of Soil Research*, vol. 6, pp. 1–8, 2005.

[46] G. F. Wilson, R. Lal, and B. N. Okigbo, "Effects of cover crops on soil structure and on yield of subsequent arable crops grown under strip tillage on an eroded alfisol," *Soil and Tillage Research*, vol. 2, no. 3, pp. 233–250, 1982.

[47] E. Owusu-Sekyere, E. K. Nakashima, and T. Wakatsuki, "Extending cocoa agroforestry into sawah ecosystem in Ghanaian inland valleys," *Ghana Journal of Agricultural Science*, vol. 43, pp. 37–44, 2010.

[48] S. Hirose and T. Wakatsuki, *Restoration of Inland Valley Ecosystems in West Africa*, Association of Agriculture and Forestry Statistics, Tokyo, Japan, 2002.

[49] P. N. Windmeijer and W. Andriesse, Eds., *Inland Valleys in West Africa. An Agro-ecological Characterization of Rice-growing Environments*, Institute for Land Reclamation and Improvement (ILRI), Wageningen, The Netherlands, 1993.

[50] T. Wakatsuki and T. Masunaga, "Ecological engineering for sustainable food production and the restoration of degraded watersheds in tropics of low pH soils: focus on West Africa," *Soil Science and Plant Nutrition*, vol. 51, no. 5, pp. 629–636, 2005.

[51] A. L. Kaswamila and J. A. Mkavidanda, "The neglect of traditional agro-forestry (TAF) and its effects on soil erosion and crop yield: the case of the West Usambara Mountains," in *Proceedings of the International Conference on Geo-Information for Sustainable Land Management (SLM '97)*, pp. 17–21, Enschede, The Netherlands, August 1997.

[52] West Africa Rice Development Association WARDA, *Africa Rice Trends, 2007*, The Africa Rice Center, Cotonou, Benin, 2008.

[53] S. E. Obalum, J. C. Nwite, J. Oppong, C. A. Igwe, and T. Wakatsuki, "Variations in selected soil physical properties with landforms and slope within an inland valley ecosystem in Ashanti Region of Ghana," *Soil and Water Research*, vol. 6, pp. 73–82, 2011.

[54] S. S. Abe, M. M. Buri, R. N. Issaka, P. Kiepe, and T. Wakatsuki, "Soil fertility potential for rice production in West African lowlands," *Japan Agricultural Research Quarterly*, vol. 44, no. 4, pp. 343–355, 2010.

[55] A. Bationo, A. Hartemink, O. Lungu et al., "African soils: their productivity and profitability of fertilizer use," in *Proceedings of the African Fertilizer Summit*, Abuja, Nigeria, June 2006.

[56] B. Koné, G. L. Amadji, S. Aliou, S. Diatta, and C. Akakpo, "Nutrient constraint and yield potential of rice on upland soil in the south of the Dahoumey gap of West Africa," *Archives of Agronomy and Soil Science*, vol. 57, pp. 763–774, 2011.

[57] J. C. Nwite, C. A. Igwe, and T. Wakatsuki, "Evaluation of sawah rice management system in an inland valley in southeastern Nigeria. I: soil chemical properties and rice yield," *Paddy and Water Environment*, vol. 6, no. 3, pp. 299–307, 2008.

[58] M. M. Buri, R. N. Issaka, and T. Wakatsuki, "Determining optimum rates of mineral fertilizers for economic rice grain yields under the "sawah" system in Ghana," *West African Journal of Applied Ecology*, vol. 12, 2008.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 687160, 6 pages
doi:10.1155/2012/687160

Research Article

Lifestyle Influence on the Content of Copper, Zinc and Rubidium in Wild Mushrooms

J. A. Campos,[1] J. A. De Toro,[2] C. Pérez de los Reyes,[1] J. A. Amorós,[1] and R. García-Moreno[3]

[1] *Departamento de Producción Vegetal y Tecnologia Agraria, UCLM, Ciudad Real, Spain*
[2] *Instituto Regional de Investigación Científica Aplicada (IRICA) and Departamento de Física Aplicada, Universidad de Castilla-La Mancha, 13071 Ciudad Real, Spain*
[3] *Departamento de ciencias de la Navegación y de la Tierra, Universidade da Coruña, A Coruña, Spain*

Correspondence should be addressed to J. A. Campos, juanantonio.campos@uclm.es

Received 2 December 2011; Revised 3 February 2012; Accepted 6 February 2012

Academic Editor: Philip White

The concentration of 18 trace elements in several species of fungi (arranged in three groups: ectomycorrhizae, saprobes, and epiphytes) has been determined. The measurements were made using the methodology of X-ray fluorescence. Higher contents of Cu and Rb (with statistical support) have been found in the ectomycorrhizal species. The Zn content reached higher concentrations in the saprophytic species. According to the normality test and the search for outliers, the species *Clitocybe maxima* and *Suillus bellini* accumulate large amounts of Cu and Rb, respectively, so that both can be named as "outliers." The leftwards displacement of the density curves and their nonnormality are attributed to the presence of these two species, which exhibit hyperaccumulation skills for Cu and Rb, respectively. Regarding Zn absorption, no particular species were classified as outlier; therefore it can be assumed that the observed differences between the different groups of fungi are due to differences in their nutritional physiology.

1. Introduction

Fungi are vital to ecosystem health as they play crucial roles in the geochemical cycles, element mobilization, and organic matter decomposition. As mycorrhizas they can improve plant growth by increasing uptake of nutrients, and as saprobes they are related to the recycling of biomass mineral constituents [1, 2]. Special mention deserves their role as wood decay agents since very little species of other groups of organisms are capable to attack recalcitrant substances such as cellulose or lignin. Because of the vital importance of fungi to the well-being of an entire ecosystem, the interaction of fungi with the organic and inorganic substrate should be traced.

Over the last few years many articles have been published on the subject of elemental content in sporocarps of wild fungi. Some of them were focused on the perspective of the nutritional skills, or toxicity, when consumed by humans [3–10] and others tried to settle differences between different species or places [11, 12]. There are also some recent papers on the subject of the weathering properties of wild mushrooms and their relation with the mineral particles of the soil [13–18]. Some studies have shown a correlation between fungal metal concentrations and point sources of metal pollution such as smelters or roadsides [19, 20].

The sporocarps of basidiomycetes have a collection of morphological features, by which we can identify and discriminate the species, and also a short lifetime, generally no more than 7-8 days, although the mycelium may live for many years. Thus, the fungal sporocarps become an advantageous material for the study of the concentrations of the different elements in a wide range of species and, therefore, to track the specific relations with the ecological niche where they live. The hypothesis of our work is that the differences in the absorption rates between fungal species of different lifestyles are a reflection of the substrate from which they feed. The aim of this study is threefold: (1) to compare trace element concentrations between different species and lifestyles; (2) to identify the elements that could be used as lifestyle indicators; (3) to identify the species with a special behaviour in the absorption of certain trace elements.

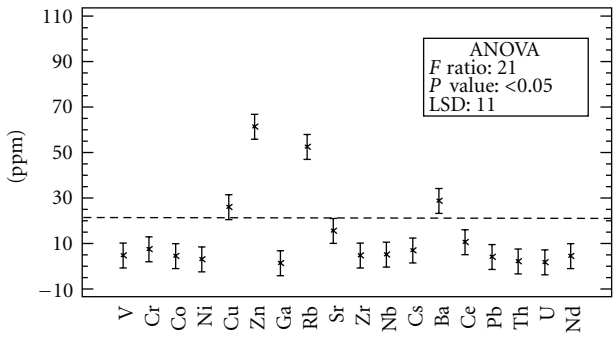


Figure 1: Mean value of trace element content averaged over the 18 fungi species studied. The dashed line separates the elements with statistically significant different values following an ANOVA test.

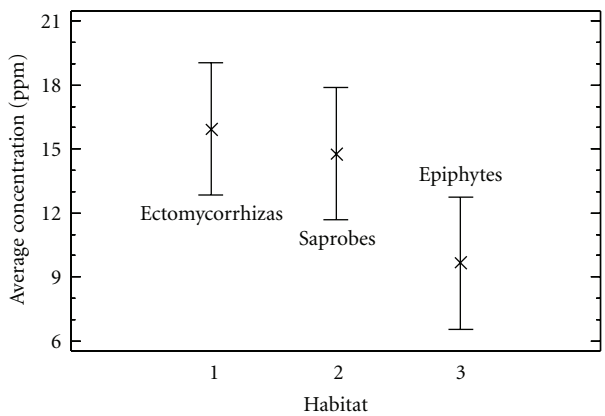


Figure 2: Mean value of trace element concentrations (averaged over the 18 elements measured and the seven species in each lifestyle) for each of the three lifestyles studied. The results of an ANOVA test are also shown.

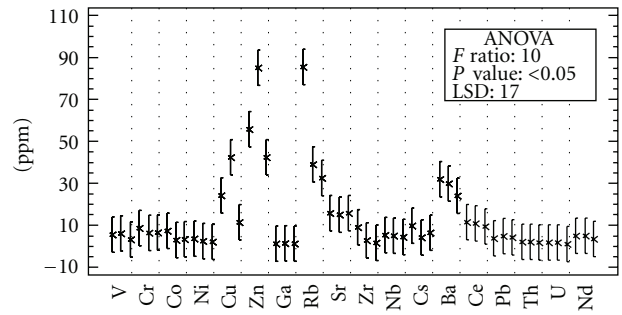


Figure 3: Mean value of each trace element for each of the three lifestyles studied (average over the seven species in each group) following the sequence ectomycorrhizae, saprobes, and epiphytic (from left to right in each element box).

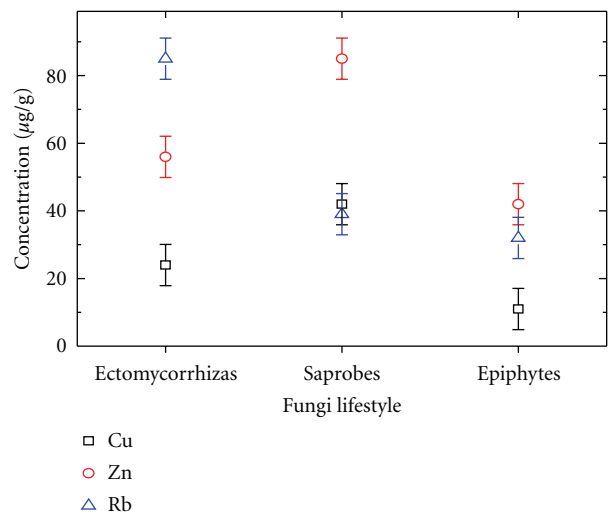


Figure 4: Mean content ($\mu g \cdot g^{-1}$ DM) of Cu, Zn, and Rb in each of the three lifestyles studied.

2. Material and Methods

2.1. Sampling and Sample Preparation. Sporocarps were collected from an area with large well-preserved mixed forest of pines and oaks on quartzite acidic soils in the province of Ciudad Real (Spain). We carried out a systematic sampling by which the complete fruiting bodies (cap and stalk) were carefully collected rejecting those very mature or rotten. All samples were brushed and washed with distilled water, then dried at 60°C for 48 h, powdered and sieved (100 μm mesh). The resulting powder was stored in hermetic plastic recipient until analysis.

2.2. Species Classification. The classification of the species of mushrooms tested in our study was made according to the systematic keys of Phylum Basidiomycota for European fungi and taking into account the chorological list of species cited in the region of Castilla La Mancha [21]. We have also used the Florule Evolutive des Basidiomycotina du Finisterre by Alain Gerault as a reference of most European species descriptions. These high-quality keys are accessible only via Internet (http://projet.aulnaies.free.fr/Florules/).

2.3. Metallic Elements Quantification (AC/AQ). The content of eight teen metals was measured by X-ray fluorescence spectrometry. The X-ray intensity was adjusted to obtain a LLD (low limit detection) of around 0.5 ppm for each element. For the use of this analytical method, each powdered sample (5.0 g) was mixed and homogenized with 0.5 mL methyl methacrylate (Vacite) and pressed into a pellet of 4.0 cm in diameter at a pressure of 150 kN. X-ray measurements were performed on the pellet samples using a wavelength dispersive X-ray fluorescence spectrometer (PHILIPS-PW2404 Pananalytical, Magix-Pro model) equipped with logging data software. The concentration of each metal is expressed in $\mu g \cdot g^{-1}$ (dry weight basis). In our work the spectrometer was subjected to a standardized calibration using 12 known elemental concentration standards supplied by the manufacturer. The exposition time to X-ray of the samples was calculated to provide errors below 2% in ten repeats of the same sample. Pure quartzite ground sand (SiO_2) was used as blank. Regarding the matrix effect, both

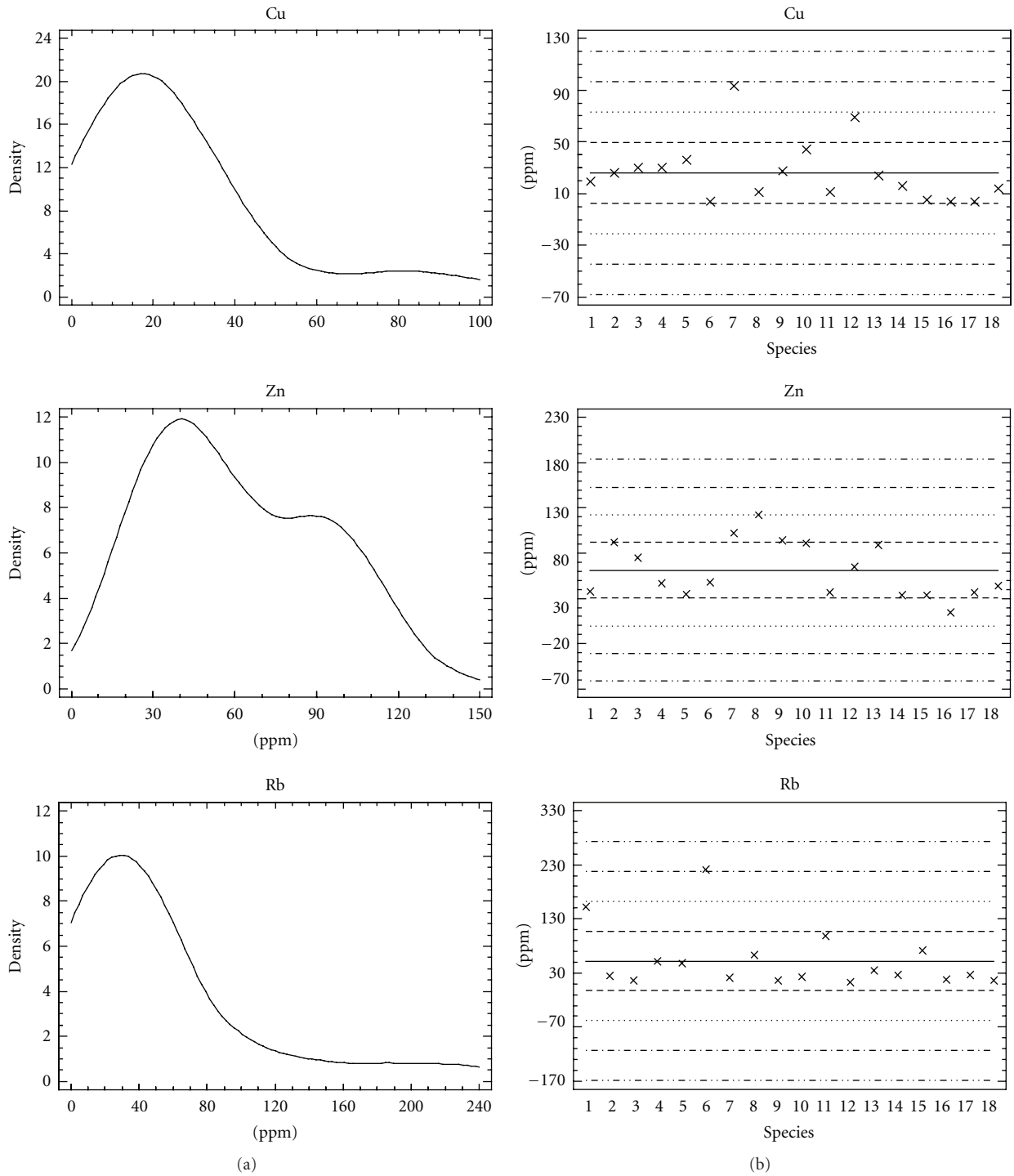


Figure 5: Nonparametrical density curves (a) and studentized values plots (b) for Cu, Zn, and Rb. The mean value is depicted by a solid line. The horizontal dashed lines are multiples of the standard deviation. The abscissa tick labels correspond to the species numbered in Table 2.

the standards provided for calibration by the manufacturer and the samples to be analyzed were homogenized and pressed in the same way so that all the (cylindrical) samples had the same physical characteristics. Also, for each sample measurement, the calcination percentage of each species, determined previously, was taken into account.

3. Statistics and Data Analysis

The concentration of each metal in each species as well as the total metallic content accumulated by each species were tested with a one-way analysis of variance (ANOVA) in order to establish statistically significant variations in the different

Table 1: Content ($\mu g \cdot g^{-1}$DM) of trace elements in fungi sporocarps for the 18 species studied. The species are sorted by their lifestyle (Ectomycorrhizae, saprobes, and epiphytic).

		V	Cr	Co	Ni	Cu	Zn	Ga	Rb	Sr	Zr	Nb	Cs	Ba	Ce	Pb	Th	U	Nd
Ectomycorrhizal	*1-Amanita phalloides* (Fr.)Link	9	14	3	6	19	38	2	152	16	26	6	8	34	10	3	3	2	6
	2- Hebeloma sinapizans (Paulet: Fr.) G.	4	6	3	2	26	92	1	24	15	3	5	3	30	8	3	2	1	4
	3- Lactarius zugazae	4	6	3	5	30	75	1	16	17	2	4	3	29	10	4	2	1	4
	4- Paxillus involutus (Batsch.:Fr.)Fr.	6	6	24	2	30	47	1	51	15	6	5	5	31	14	5	2	1	4
	5- Russula delica Fr.	5	10	3	4	36	35	1	48	15	9	5	8	32	13	5	2	2	5
	6- Suillus bellini (Inzenga)Kuntze	5	9	7	4	4	48	1	221	16	7	6	12	35	14	2	2	2	5
Saprobe	*7- Clitocybe maxima* Fr.:Fr.	15	7	3	3	93	102	1	21	15	2	5	5	30	11	4	2	4	3
	8- Entoloma lividum (Bull.)Quél.	5	6	2	2	11	122	1	64	14	1	5	3	26	8	3	2	1	3
	9- Lepista inversa (Scop).Pat	5	5	3	2	27	94	1	16	15	1	5	3	32	12	3	2	1	7
	10- Lepista nuda (Bull.:Fr.)Cooke	4	6	3	2	44	91	1	23	15	2	5	2	29	7	5	2	1	3
	11- Lycoperdon perlatum Pers.:Pers	4	11	3	4	11	37	1	98	15	8	4	7	35	14	7	2	2	9
	12- Macrolepiota procera (Scop.:Fr.)S.	3	5	3	1	69	65	1	12	15	2	4	4	26	13	4	2	1	4
Epiphytic	*13- Agrocybe aegerita* (Brig.)Fayod	2	6	3	2	24	89	1	35	15	2	5	5	27	9	4	2	1	4
	14- Armillaria mellea (Vahl.:Fr.)P. K.	3	6	3	2	16	34	1	27	14	2	5	9	29	8	4	2	1	3
	15- Gymnopilus spectabilis (Fr.)Singer	4	9	4	2	5	34	1	72	14	1	4	11	21	9	4	1	1	4
	16- Hericium erinaceus (Bull.:Fr:)Pers	3	7	3	2	4	15	1	18	15	1	4	5	27	6	4	2	1	1
	17- Inonotus hispidus (Bull.:Fr.)P.Karst	4	5	3	2	4	37	2	26	21	1	5	4	26	12	5	2	1	5
	18- Meripilus giganteus (Pers.:Fr.)P.K.	4	7	3	2	14	44	2	17	15	2	4	4	14	11	4	2	1	2

Table 2: Shapiro-Wilk normality test for Cu, Zn, and Rb.

Normality test Shapiro-Wilk-W	Statistic	*P* value	Skewness	Normality
Cu	0.8	0.002	3	Rejected
Zn	0.9	0.7	2	Assumed
Rb	0.7	0.001	4	Rejected

elements content between individual species and between different lifestyles. The same test was performed to analyze differences in the total content of the different elements. The density curves for those elements which showed statistical differences were depicted, and a test of normality [22] and a statistical analysis for the identification of outliers was carried out. All statistical analyses were performed using Statgraphics Centurion XV (Statistical Graphics Corp., Rockville, MD, USA).

4. Results and Discussion

Although most of the concentration values found by us fall between the thresholds presented by other authors using other analytical procedures [3–10], there are some cases where they do not. Particularly notable are the results of Borovička et al. [23], who give concentrations for some heavy metals several orders of magnitude lower than those found in our analysis [24, 25]. It must be noticed that, although the spectrometer was thoroughly calibrated, it seems that the methodology of X-ray fluorescence may not be adequate when the concentrations are close to the low detection limit (e.g., heavy metals).

The data of the trace elements measured in our work for each species are given in Table 1. The species have been ranked according to their lifestyle (ectomycorrhizal, saprobe, or epiphytic on wood) and alphabetically within each of these groups. They have also been numbered for a better understanding of the figures. Note that 18 different elements have been measured in 18 different species.

The content of each of the elements in the studied 18 species of mushrooms is around $6\,\mu g \cdot g^{-1}$ except for Cu, Zn, Rb, and Ba, where the concentrations are significantly higher. These elements are absorbed in greater quantity than the rest, with contents around $25\,\mu g \cdot g^{-1}$ for Cu and Ba and around $60\,\mu g \cdot g^{-1}$ for Zn and Rb (see Figure 1). The increased concentration of these elements in the fungal biomass is probably due to their presence in greater quantities in the substrate solution, although there might be other additional mechanisms for a selective absorption of these elements, especially for Cu and Zn, since it has been reported that the levels reached in fungal biomass are species dependent [26]. We have not found statistical differences between the total content of trace elements when the lifestyles are compared. Although the total content in epiphytic mushrooms is clearly lower, this difference does not reach statistical significance (see Figure 2).

When comparing the content of each of the elements between the three different lifestyles (see Figure 3), it can be observed that differences appear only for Cu, Zn, and Rb. In the case of Ba, the other element with an overall concentration in the fungal biomass statistically higher to the rest, no significant differences were found between lifestyles. Cu accumulates in saprobe species in higher quantities. Among the epiphytic and ectomycorrhizal species there were not found significant differences for this element. The same

holds for Zn, although with major differences (Figures 3 and 4). On the other hand, Rb is more efficiently absorbed by ectomycorrhizal species, without significant differences between the saprophytic and epiphytic groups of species.

Once established that there are clear differences in the absorption of these elements between the three groups of species (ectomycorrhiza, saprobes, and epiphytes), it is convenient to make a more detailed analysis trying to find out the causes of that differential absorption. To this purpose we have performed a test of normality [22] and a search for species with a particular behaviour in the absorption of these elements (statistical "outliers"). The final goal of this analysis is to verify whether that data lies within a normal distribution or not, and, if not, to examine the presence of species that may be considered as outliers. It is assumed that these species have a peculiar absorption for these elements and, therefore, are responsible for breaking the expected normal distribution of the data.

Figure 5 displays the nonparametric density curves for Cu, Zn, and Rb. The right- hand panels show the corresponding tests for outliers identification with the distribution of each species data (numbered according to Table 1) with respect to the mean (solid line) and standard deviation units (dashed lines). Normality is rejected for Cu and Rb (Table 2), with a leftwards shift of the density curve in both cases, which means that there are values high enough above the mean to break normality. The panels on the right-hand side show a different representation of the dispersion of the data in the different species. In the case of Cu, there is a datapoint with value almost three times the standard deviation, which can therefore be considered an outlier. This datapoint corresponds to *Clitocybe maxima*, a species located within the saprobe group. It seems that the presence of Cu in fungal biomass is more related to specific absorptive skills of the species than to the concentration of this element in the substrate solution, since we cannot expect a high spatial heterogeneity of the deposits of this element in a forest soil (e.g., from organic waste, excrements, etc.).

The density curve of the data shows that Zn is set to normal, according to the Shapiro-Wilk test (Table 2); however two modes are hinted, where the smaller one; centered at 95 ppm; corresponds to the saprobe group, consistently with the observed significant differences in Figures 3 and 4, and thus confirming that differences between saprobe species and the rest of the lifestyles do exist. It can therefore be concluded that the differences found between the saprobe species and the other groups are based on the physiological characteristics shared by the species of this particular trophic group and not by the presence of outliers. Alonso et al. [26] give values between 130 and 267 $\mu g \cdot g^{-1}$ in saprobe species of genus *Agaricus* (*A. campestris, A. Macrospores, and A. silvicola*). These authors suggest that saprophytic species have more efficient mechanisms to absorb trace elements than ectomycorrhiza ones.

In the case of Rb the test of normality (Table 2) indicates that the data were not normally distributed. In a more detailed analysis (Figure 5) it can be seen that the species *Suillus bellini* accumulates this element in a much greater amount than the other species (three times the standard deviation), shifting the density curve leftwards and breaking the expected normality. This species can be considered as a statistical outlier with a particular absorptive behaviour. We can assume that the absorption of Rb is species specific and that it is supported by a particular nutritional physiology not shared by the whole trophic group. The special skills of some ectomycorrhizal species (e.g., *Suillus* sp.), by attacking mineral particles of soil, may underlie this result.

5. Conclusions

The concentrations of the trace elements Cu, Zn, and Rb found in the biomass of eighteen species of mushrooms are related to the lifestyle (ectomycorrhiza, saprobe, or epiphytic on wood). Cu and Zn accumulate in greater amounts in saprobes species and Rb reaches higher concentrations in the ectomycorrhiza species group. Further analysis indicates that the species *Clitocybe maxima* and *Suillus bellini* act as hyperaccumulators of Cu and Rb, respectively, and this is crucial in establishing the statistical significance between the trophic groups. Hyperaccumulating species do not appear in the case of Zn and, therefore, we assume that the differences found between the three lifestyles stem from the peculiarities of the nutritional physiology shared by the species in each group.

References

[1] V. Wiemken, "Contributions of studies with in vitro culture systems to the understanding of the ectomycorrhizal symbiosis," in *Mycorrhiza: Structure, Function, Molecular Biology and Biotechnology*, A. Varma and B. Hock, Eds., Springer-Verlag, New York, NY, USA, 1995.

[2] P. E. Courty, M. Buée, A. G. Diedhiou et al., "The role of ectomycorrhizal communities in forest ecosystem processes: new perspectives and emerging concepts," *Soil Biology and Biochemistry*, vol. 42, no. 5, pp. 679–698, 2010.

[3] D. Mendil, O. D. Uluözlü, E. Hasdemir, and A. Çağlar, "Determination of trace elements on some wild edible mushroom samples from Kastamonu, Turkey," *Food Chemistry*, vol. 88, no. 2, pp. 281–285, 2004.

[4] N. Dursun, M. M. Özcan et al., "Mineral contents of 34 species of edible mushrooms growing wild in Turkey," *Journal of the Science of Food and Agriculture*, vol. 86, no. 7, pp. 1087–1094, 2006.

[5] L. Cocchi, L. Vescovi, L. E. Petrini, and O. Petrini, "Heavy metals in edible mushrooms in Italy," *Food Chemistry*, vol. 98, no. 2, pp. 277–284, 2006.

[6] P. K. Ouzouni, P. G. Veltsistas, E. K. Paleologos, and K. A. Riganakos, "Determination of metal content in wild edible mushroom species from regions of Greece," *Journal of Food Composition and Analysis*, vol. 20, no. 6, pp. 480–486, 2007.

[7] K. Chudzyński and J. Falandysz, "Multivariate analysis of elements content of Larch Bolete (*Suillus grevillei*) mushroom," *Chemosphere*, vol. 73, no. 8, pp. 1230–1239, 2008.

[8] J. J. Falandysz, T. Kunito, R. Kubota et al., "Some mineral constituents of Parasol mushroom (*Macrolepiota procera*)," *Journal of Environmental Science and Health, Part B*, vol. 43, pp. 187–192, 2008.

[9] P. Kalač, "Chemical composition and nutritional value of European species of wild growing mushrooms: a review," *Food Chemistry*, vol. 113, no. 1, pp. 9–16, 2009.

[10] P. Kalač, "Trace element contents in European species of wild growing edible mushrooms: a review for the period 2000–2009," *Food Chemistry*, vol. 122, no. 1, pp. 2–15, 2010.
[11] G. Tyler, "Metal accumulation by wood-decaying fungi," *Chemosphere*, vol. 11, no. 11, pp. 1141–1146, 1982.
[12] J. Vetter, "Mineral composition of basidiomes of *Amanita* species," *Mycological Research*, vol. 109, no. 6, pp. 746–750, 2005.
[13] R. Landeweert, E. Hoffland, R. D. Finlay, T. W. Kuyper, and N. Van Breemen, "Linking plants to rocks: ectomycorrhizal fungi mobilize nutrients from minerals," *Trends in Ecology and Evolution*, vol. 16, no. 5, pp. 248–254, 2001.
[14] E. Hoffland, T. W. Kuyper, H. Wallander et al., "The role of fungi in weathering," *Frontiers in Ecology and the Environment*, vol. 2, pp. 258–264, 2004.
[15] G. M. Gadd, *Fungi in Biogeochemical Cycles*, Cambridge University Press, Cambridge, UK, 2006.
[16] G. M. Gadd, "Global biogeochemical cycling: fungi and their role in the biosphere," in *Encyclopedia of Ecology*, Elsevier, Amsterdam, The Netherlands, 2007.
[17] R. Amundson, D. D. Richter, G. S. Humphreys, E. G. Jobbágy, and J. Gaillardet, "Coupling between biota and earth materials in the critical zone," *Elements*, vol. 3, no. 5, pp. 327–332, 2007.
[18] L. van Schöll, T. W. Kuyper, M. M. Smits, R. Landeweert, E. Hoffland, and N. V. Breemen, "Rock-eating mycorrhizas: their role in plant nutrition and biogeochemical cycles," *Plant and Soil*, vol. 303, no. 1-2, pp. 35–47, 2008.
[19] J. D. McCreight and D. B. Schroeder, "Cadmium, lead and nickel content of Lycoperdon perlatum Pers. in a roadside environment," *Environmental Pollution*, vol. 13, no. 4, pp. 265–268, 1977.
[20] R. Bargagli and F. Baldi, "Mercury and methyl mercury in higher fungi and their relation with the substrata in a cinnabar mining area," *Chemosphere*, vol. 13, no. 9, pp. 1059–1071, 1984.
[21] F. D. Calonge, G. Moreno et al., "Flora Micológica de Castilla La Mancha. Situación actual y conservación de los hongos del bosque," in *Memoria Final (2004–2007)*, Hernandez-Crespo, Ed., Real Jardín Botánico CSIC, Madrid, Spain, 2008.
[22] S. S. Shapiro and M. B. Wilk, "An analysis of variance test for normality (complete samples)," *Biometrika*, vol. 52, pp. 591–611, 1965.
[23] J. Borovička, J. Kubrová, J. Rohovec, Z. Řanda, and C. E. Dunn, "Uranium, thorium and rare earth elements in macrofungi: what are the genuine concentrations?" *BioMetals*, vol. 24, no. 5, pp. 837–845, 2011.
[24] J. A. Campos, N. A. Tejera, and C. J. Sánchez, "Substrate role in the accumulation of heavy metals in sporocarps of wild fungi," *BioMetals*, vol. 22, no. 5, pp. 835–841, 2009.
[25] J. A. Campos, "Nutrients and trace elements content of wood decay fungi isolated from oak (*Quercus ilex*)," *Biological Trace Element Research*, vol. 144, no. 1-3, pp. 1370–1380, 2011.
[26] J. Alonso, M. A. García, M. Pérez-López, and M. J. Melgar, "The concentrations and bioconcentration factors of copper and zinc in edible mushrooms," *Archives of Environmental Contamination and Toxicology*, vol. 44, no. 2, pp. 180–188, 2003.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 243815, 8 pages
doi:10.1155/2012/243815

Research Article

The Effect of Rainfall Characteristics and Tillage on Sheet Erosion and Maize Grain Yield in Semiarid Conditions and Granitic Sandy Soils of Zimbabwe

Adelaide Munodawafa

Department of Land and Water Resources Management, Midlands State University, P.O. Box 9055, Gweru, Zimbabwe

Correspondence should be addressed to Adelaide Munodawafa, munodawafaa@msu.ac.zw

Received 4 November 2011; Revised 15 January 2012; Accepted 16 January 2012

Academic Editor: María Cruz Díaz Álvarez

In semiarid regions, rainfall is one of the primary factors affecting soil erosion and crop production under rain-fed agriculture. The study sought to quantify the effect of rainfall characteristics on sheet erosion and maize grain yield under different tillage systems. It was carried out under semiarid conditions and infertile sandy soils of Zimbabwe. Rainfall amount and intensity were recorded every 24 hours, while sheet erosion was measured from four tillage systems (Conventional Tillage (CT), Mulch Ripping (MR), Tied Ridging (TR) and Bare Fallow (BF)). Maize (*Zea mays L.*) was grown on three tillage systems (CT, MR, and TR). Rainfall amount varied significantly ($P < 0.001$) between seasons (164–994 mm). CT recorded the highest average soil losses (15 t/ha), while MR and TR recorded 1.3 and 1.2 t/ha, respectively. Maize grain yields increased with increasing seasonal rainfall giving yield-responses of 0.9 t/ha (TR) to 1.3 t/ha (MR) for every 100 mm rainfall increment. Overall, treatments didnot differ significantly ($P < 0.497$), except during drier seasons ($P < 0.025$). Regression equations showed that yields can be confidently predicted using rainfall amount and time, with R^2 values of 0.82 to 0.94. Maize grain yields proved to be mostly dependent on rainfall amount than fertility. The productivity of the soils decreased with increased length of cultivation.

1. Introduction

Rill and gully erosion in the smallholder areas of Zimbabwe is largely under control through mechanical conservation structures such as contour ridges, grassed waterways, and storm drains [1]. However, sheet erosion is still a major threat to soil fertility and productivity. The sheet erosion process is selective and deprives the soil of its fine particles (clay and organic matter) [2]. These particles are easily splashed out and carried in suspension, while the heavier particles remain behind [3–5]. The soils are thus impoverished as these nutrient reservoirs are lost together with inherent and applied plant nutrients. The bulk density of the soils is increased and plant available water is decreased. According to Stocking and Peake [6], the changes in soil conditions, in many cases, may be describing the effect of erosion induced low soil productivity.

In soil erosion research, rainfall amount and intensity (erosive power of rainfall) have been found to be the fundamental factors affecting soil erosion [7, 8]. The impact of raindrops on the soil surface results in temporary capping of the soil and lowered infiltration rate, thus generating runoff [9–11]. Runoff is directly dependent on rainfall amount and intensity and soil loss, being a function of runoff also depends primarily on these factors. According to Morgan [12], sheet erosion occurs when, during a rainstorm, soil moisture storage and/or the infiltration capacity of the soil are exceeded.

Rainfall is also the primary factor affecting crop production in rain-fed agriculture [5]. Previous studies in semiarid regions have shown that the yield parameter is mainly dependent on the amount and distribution of rainfall. Elwell [13] found a linear relationship between rainfall amount and yield on granitic sands and high rainfall conditions of

Zimbabwe, where yield increased proportionally to rainfall amount. The soil type also influences crop production as determined by the fertility level as well as the soil physical characteristics. The crop production potential of granitic sandy soils is low, but if adequate fertilisers are applied, average yields can be achieved [14]. However, the fertiliser application is very much dependent on rainfall, so that rainfall becomes the most important factor influencing crop production. Mid-season droughts are common in the semiarid areas due to the erratic nature of rainfall and the soils' low water holding capacity [15]. Rainfall distribution becomes an important factor if the effect of these mid-season droughts is to be minimized.

While it is generally known and acceptable, that rainfall (amount, intensity, and distribution) affects soil erosion and productivity, the principle cannot be applied everywhere successfully. Kaihura et al. [16] stated that the rates and the effects of erosion are dependent on the soil type and agro-ecological conditions. Thus, it is important to define the factors that affect soil erosion in different regions by developing equations that estimate soil erosion and productivity to cover areas of the same climatic and ecological conditions. These equations should, however, be simple and straight forward enough to be of benefit to the farmers. The objective of these equations should be to use some important and easily measurable variables to predict parameters of agricultural production. Thus from either a crop production or soil erosion/conservation point of view, rainfall characteristics and distribution are of importance if farmers are to successfully manipulate the soil and reduce the destructive potential of tropical storms. The objective of this study is, therefore, to determine the rainfall characteristics that affect soil erosion and maize grain yield. This study, therefore highlights the conservation potentials of different tillage systems under the semi-arid conditions of Zimbabwe. Furthermore, simple soil loss, runoff, and yield equations showing the most important factors that affect these parameters are developed.

2. Materials and Methods

2.1. Experimental Site. Zimbabwe lies well within the tropics but its climate is subtropical, being moderated by altitude. Its climate is thus classified as temperate (mild mid-latitude), with dry winters and hot summers (Cwb) according to the Koeppen climate classification system [17]. The average temperatures rarely exceed 33°C in summer or drop beyond 7°C in winter [18]. The country has been classified into five agroecological regions, namely, Natural Regions I, II, III, IV, and V. Only Natural Regions I and II have relatively high effective rainfall and are suitable for intensive agricultural production. Natural Regions III, IV, and V constitute 83% of the total land area (92% of small-holder farming area) and are not suitable for intensive, high input agriculture [15]. Zimbabwe's soils are predominantly derived from granite and the clay content of these soils varies according to the degree of weathering (influenced by rainfall) and catenal position [19, 20]. From among all the soils derived from granite, the sandy soils, of the fersiallitic group, comprise the majority, about 70% of the land area [19] and are dominant in the small-holder farming areas [21]. The agricultural potential of these soils is fair [20], and their productivity is likely to decline under intensive continuous cropping.

The study was carried out at Makoholi Research Station situated 30 km north of Masvingo town, which is the regional agricultural research centre in the medium-to-low rainfall areas. The station lies at an altitude of about 1 200 m, within Natural Region IV with an average annual rainfall of 450–650 mm. Characteristic of this region is the erratic and unreliable rainfall both between and within seasons [22]. The soils are also inherently infertile, pale, coarse-grained, granite-derived sands, (Makoholi 5G) of the fersiallitic group, Ferralic Arenosols [19, 23]. Arable topsoil averages between 82 and 93% sand, 1 and 12% silt, and 4 and 6% clay [21, 24]. The small amount of clay present is in a highly dispersed form and contains a mixture of 2 : 1 lattice minerals and kaolinite [23]. The organic matter content is also very low, about 0.8%, while pH ($CaCl_2$) is as low as 4.5. The soils are generally well drained with no distinct structure [24], but some sites have a stone line between 50 and 80 cm depth. The high infiltration rate and low water holding capacity are due to the soil texture characteristics.

2.2. Experimental Design and Treatments. The treatments were laid out in a randomised block design replicated three times. Four tillage systems were considered, conventional tillage, mulch ripping, tied ridging, and bare fallow. Conventional tillage is the most widely used tillage practice in the small-holder farming areas of Zimbabwe constituting 73–90% of the cultivated area [25]. The remainder of the land is ploughed using hired tractor (5–25%) and less than 1% is under tillage systems that conserve soil, moisture, nutrients and/or energy inputs [25]. Mulch ripping and tied ridging systems have a great potential in conserving soil and water and are being promoted in a bid to effectively manage the natural resources and sustain productivity.

Tillage for the different systems was carried out as follows: *conventional tillage* (CT): ploughed to 23 cm using an ox-drawn mouldboard plough; *mulch ripping* (MR): crop residues were left to cover the ground and only rip lines were opened between the mulch rows, 25 cm deep, using a ripper tine; *tied ridging* (TR): 20 cm high crop ridges were laid out at 1% slope and were 90 cm apart. Ties were constructed in the furrows at 1–1.5 m intervals to create microdams. *Bare fallow* (BF): tractor ploughed and kept crop and weed free throughout the season. Maize (*Zea mays L.*) is the staple food in Zimbabwe and is planted on >70% of all cultivated land in the small-holder sector. Thus maize was planted on all plots, except BF, at a population of 36 000 plants/ha. Optimal recommended fertiliser rates were applied at recommended times. For yield assessment, two subplots of 3.6×6 m were marked out on CT and MR plots, while four subplots were marked out on TR. Rainfall was measured every 24 hours, using standard and autograghic rain gauges.

2.3. Collection of Runoff and Sediments. The standard soil erosion methodology for Zimbabwe was used [26, 27], where

the plots were laid out at 4.5% slope. Erosion plots measured 30 × 10 m for CT, BF, and MR and 150 × 4.5 m for TR. Surface runoff and soil loss from each plot were collected at the bottom of the plots in 1500 litre conical tanks. The tanks were emptied daily and soil loss and runoff quantified [28]. The collection tanks were calibrated and runoff was measured using a metrestick. Once the first tank was full its overflow passed through a divisor box with ten slots, which channelled only one-tenth of the overflow into the second tank. Nine-tenths of this overflow was allowed to drain away, thus increasing the capacity of the second tank. Due to the larger net plots of the tied ridging treatment, three tanks were installed, so as to capture the anticipated larger volume of sediments.

2.4. Sampling Eroded Material. Rainfall data was collected from 1st of October through April of each year, which corresponds to the seasonal rainfall for this region. Tanks were emptied at the end of each storm unless the interval between storms was too short to allow emptying. Sediments and runoff (including the suspended material) collected from runoff plots were treated as different entities. Suspension was pumped out and subsampled for the determination of soil concentration in runoff, using the Hach spectrophotometer DL/2000. Later the sludge was transferred into 50 litre milk churns, topped up with water to a volume of 50 litres and weighed. The mass of oven dry soil, M_o (kg), was calculated using the following equation [21, 26]:

$$M_o = 1.7 \times (M_s - M_w), \tag{1}$$

where M_s is mass of fixed volume of sludge (kg), M_w is mass of the same volume full of water (kg), and 1.7 is constant for the soil type.

2.5. Statistical Analysis. Data was analysed using Genstat 5 Release 1.3 for analysis of variance (ANOVA). Equations to estimate runoff, soil loss, and crop yield were developed for each tillage system using the climatic data and time factor, as climatic data is readily available and easily accessible to all and undoubtedly greatly influences agricultural production at any given area. Time also affects yield or soil degradation depending on the use of a particular piece of land. Therefore, while some other parameters, for example, soil moisture and crop cover are known to influence soil erosion and productivity of the soils, [10, 29] these have not been taken into consideration, yet this does not mean that their importance is not acknowledged.

Multiple regression analysis was carried out on the data collected over nine years to find factors that determine soil loss by sheet erosion from among rainfall amount, energy, and time. Four types of regression analysis were considered:

(i) standard regression with a forward selection of variables,

(ii) multiple regression on data after logarithmic and/or inverse transformation of the dependent variables,

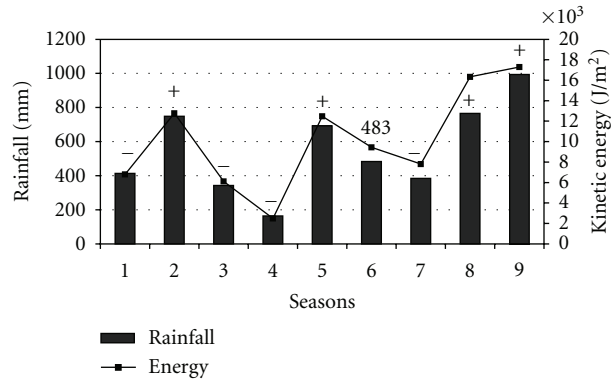


Figure 1: Seasonal rainfall energy and rainfall amount at Makoholi Contill site over nine seasons.

(iii) multiple regression after logarithmic transformation of the independent variables,

(iv) nonlinear regression analysis.

The transformations were done so as to fully explain the relationship of the dependent and independent variables as sometimes the relationship is not direct but logarithmic or exponential. Best-fit models were selected on the basis of the multiple regression coefficient (R^2) of the bare fallow (for runoff and soil loss) and conventional tillage (for yield). To enable comparison among the different tillage systems the same set of variables was used across all tillage systems.

3. Results

3.1. Rainfall Amount, Distribution, and Intensity. Over the nine years seasonal rainfall ranged from 164 mm during the 4th year, (drought) to 994 mm during 9th season (Figure 1). The average calculated over these years was 554 mm, which is well within the expected range for this natural region (450 to 650 mm). However, Figure 1 shows that although the 554 mm is within the expected range, individual seasons lie outside this range. Only one season (year 6) was within (483 mm) the range, while all the other seasons lay on either side (−; +) of the range (four seasons on each side). Apart from the fluctuations in the seasonal rainfall totals, monthly and daily rainfall distributions can also result in significant soil loss and runoff differences (Figure 2). Monthly rainfall totals, during the rainy seasons (October to April), ranged from 0 to 419 mm and daily rainfall from 0 to 182 mm. The rainfall data collected also clearly shows that the rainy season usually starts in October and extends to April, while the growing season starts in November. The wettest months are December, January, and February. During six out of nine seasons, planting was carried out in November, two seasons in December, and one season in October.

Rainfall energy can be expressed as the erosive power of rainfall and was found to be closely associated with the rainfall amount (Figure 3). Correlating the two parameters gave a correlation's coefficient of $r = 0.977$, indicating that the higher the rainfall amount, the higher the rainfall energy, that is, its erosive power.

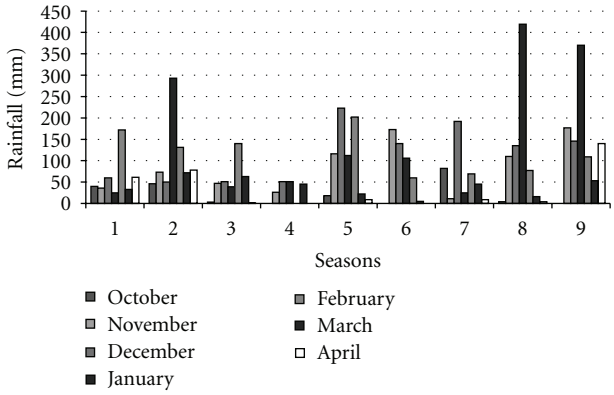


Figure 2: Monthly rainfall distribution over nine seasons at Makoholi Contill site.

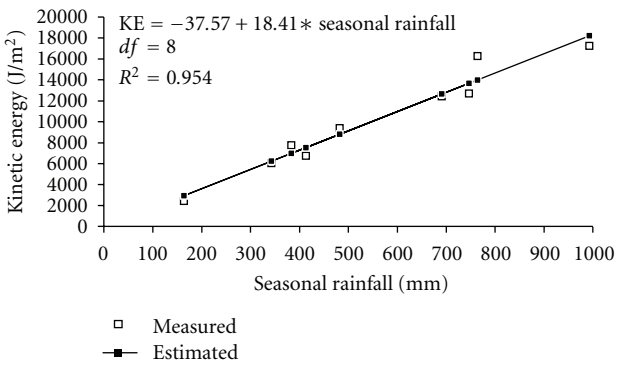


Figure 3: Correlation between seasonal rainfall amount and seasonal rainfall energy during nine seasons at Makoholi Contill site.

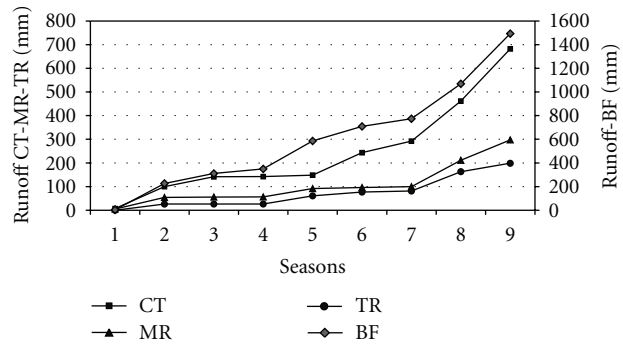


Figure 4: Cumulative runoff (mm) under different tillage systems over nine seasons at Makoholi Contill site.

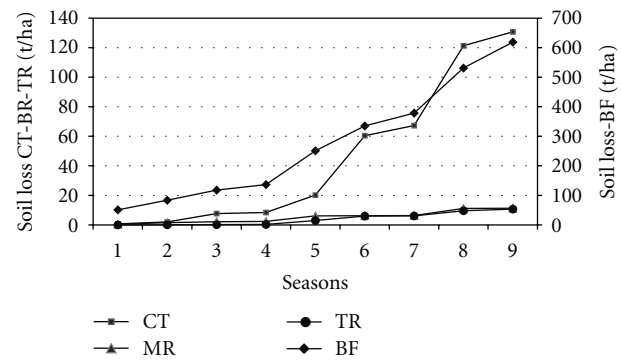


Figure 5: Soil losses under different tillage systems over nine seasons at Makoholi Contill site.

3.2. Runoff. There was a tendency for runoff to increase with the increase in the number of years of cultivation (Figure 4). The bare fallow had, as expected, the highest runoff average of 179 mm/ha over the nine years. On average, 32% of total rainfall received was lost as runoff, ranging between 17 and 43% over the nine-year period. Under this treatment-extreme conditions for accelerated erosion were created, giving the worst possible scenario under the given conditions. This treatment serves to show the erodibility of the soils under study. Among the cropped treatments, conventional tillage recorded the highest average runoff with a range of between 0.6 and 22% of total seasonal rainfall, while mulch ripping and tied ridging recorded the lowest runoff averages, which ranged from 0.3–15% and 0.0–11%, respectively. As can be seen from Figure 4, the two systems have a lower cumulative runoff compared to conventional tillage. Runoff generated from the different treatments differed significantly at $P < 0.001$. There was no significant difference between mulch ripping and tied ridging ($P = 0.385$). However, when the means of mulch ripping and tied ridging were compared with conventional tillage, the difference became significant at $P < 0.001$. Year, rainfall, and energy were also considered as sources of variance. For all the treatments, there was a significant difference ($P < 0.001$) between the different years, rainfall amount, and rainfall energy.

3.3. Soil Loss. Soil losses under the bare fallow ranged from 9 t/ha during the first year to 152 t/ha during the 8th year giving an average of 64 t/ha/yr over the nine-year period. Conventional tillage recorded the highest cumulative soil losses among the cropped treatments (Figure 5) and averaged about 15 t/ha over the nine seasons. Mulch ripping and tied ridging had, as expected, the lowest cumulative soil losses and proved to effectively conserve the soil. Analysis of variance showed the same trend as that of runoff, with differences between the treatments being significant at $P < 0.001$. There was no significant difference between mulch ripping and tied ridging ($P = 0.964$). Once again the difference between the mean of mulch ripping and tied ridging varied significantly ($P < 0.001$) when compared to conventional tillage. As in runoff, the effects of year, rainfall amount, and energy on soil loss gave significant differences at $P < 0.001$ for all the treatments except mulch ripping, where the variation was significant at $P < 0.01$.

3.4. Maize Grain Yield. Yield ranged from 0 t/ha during drier seasons to more than 7 t/ha during years with abundant rainfall. Mulch ripping had the highest cumulative yield and averaged (3.5 t/ha) over the nine years (Figure 6). Conventional tillage gave an average yield of 3.0 t/ha and tied

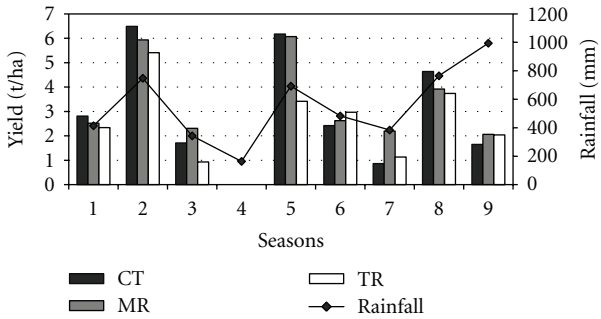


Figure 6: Maize yield under different tillage systems over nine seasons at Makoholi Contill site.

ridging recorded the lowest cumulative crop yield with an average of 2.6 t/ha. Maize grain yields under the different tillage systems did not differ significantly from one another ($P = 0.497$). Crop yields resulted in significant treatment differences only during somewhat drier seasons, or seasons characterised by poor rainfall distribution, for example, during the 7th season; yield gave a significant treatment difference at $P < 0.025$. Mulch ripping and tied ridging also differed significantly at $P < 0.025$. The different years with different rainfall amounts caused a variation in maize yield resulting in significant differences between years ($P < 0.001$) for all the treatments.

3.5. Developing Simple Equations. The high correlation between rainfall amount and rainfall energy ($r = 0.977$) left very little room from which to choose one parameter in place of the other, meaning that the two parameters can be interchanged. Therefore, if rainfall energy data is not available, rainfall amount can be used with negligible effect on the coefficient of determination (R^2).

The following parameters were used in the equations:

RO: total seasonal runoff (mm),

SL: total seasonal soil loss (kg/ha),

YI: seasonal maize grain yield (t/ha),

ENER: total seasonal rainfall energy (J/m^2),

nYEARS: number of years of cultivation,

RAIN: total seasonal rainfall amount (mm).

3.6. Runoff Equations. The following equation was used for the determination of runoff:

$$\mathrm{RO} = a + bX_1 + c\mathrm{EXP}(X_2), \tag{2}$$

where a is constant; b and c are coefficients; X_1 is ENER; X_2 is nYEARS.

Runoff under the bare fallow was estimated to increase directly with increasing rainfall energy (Table 1) at 19 mm/1000 kJ/m^2. Runoff also increased exponentially to the year, whereby the estimated runoff during the first year was 156 mm/10 000 kJ/m^2 and by the ninth year it was estimated to be 272 mm/10 000 kJ/m^2. With an R^2 value of 0.92, runoff

Table 1: The effect of rainfall energy and number of years of cultivation on runoff under different tillage systems at Makoholi Contill site.

Treatment	Constant	Energy	Exp. nYears	R^2
Bare fallow	−30.0	0.01856	0.01437	0.92
Conv. tillage	−26.9	0.00987	0.00960	0.80
Mulch ripping	−36.1	0.00661	0.00213	0.66
Tied ridging	−29.4	0.005377	−0.00223	0.67

from the bare fallow can be confidently predicted using rainfall energy and the number of years of cultivation.

Under cropped treatments, runoff was estimated to increase directly with the increase in rainfall energy at about 10 mm/1000 kJ/m^2 under conventional tillage, 6.6 mm/1000 kJ/m^2 under mulch ripping, and 5.4 mm/1000 kJ/m^2 under tied ridging. It was also predicted to increase from about 72 mm/10 000 kJ/m^2 during the first year of cultivation to about 150 mm/10 000 kJ/m^2 during the ninth year under conventional tillage, from 30 mm/10000 kJ/m^2 to 47 mm/10000 kJ/m^2 under mulch ripping and decrease from 25 mm/10000 kJ/m^2 to 6.3 mm/10000 kJ/m^2 under tied ridging. Thus runoff under conventional tillage was estimated to increase by 78 mm from the first year to the ninth year and by a mere 17 mm under mulch ripping for the same period. This increase under mulch ripping was generally a result of high runoff recorded during wet years, where the soil under the mulch was saturated (high infiltration and reduced evaporative losses). However, when rainfall amount was normal and well distributed, runoff tended to decrease with the number of years of cultivation, that is, the cumulative effect of mulch. Runoff under tied ridging was estimated to decrease by 16 mm over nine years. This is largely due to increased infiltration in the microdams which then reduces runoff. The R^2 value of 0.80 under conventional tillage is also high enough to allow for runoff to be confidently predicted using these two parameters (rainfall energy and the number of years of cultivation).

3.7. Soil Loss Equations. The following equation was used for the estimation of soil loss:

$$\mathrm{SL} = a + bX_1 + cX_2, \tag{3}$$

where a is constant, b and c are coefficients, X_1 is nYEARS, X_2 is ENER.

Under the bare fallow, the variables year and energy (Table 2) were the most descriptive ones. This was expected, as there was no ground cover to intercept rainfall energy. Soil loss was estimated to increase by 7.3 t/ha with the increase in the number of years of cultivation and by 5.1 t/ha/1000 kJ/m^2 rainfall energy. These variables explained 60% of the variation of soil loss. Using the same parameters as for the bare fallow, soil loss under the cropped treatments was estimated to increase by 2.9 t/ha under conventional tillage, decrease by 0.1 t/ha under mulch ripping, and increase by 0.2 t/ha under tied ridging with every increase in the number of years

Table 2: The effect of rainfall energy and number of years of cultivation on soil loss under different tillage systems at Makoholi Contill site.

Treatment	Constant	nYears	Energy	R^2
Bare fallow	−23.6	7.28	0.00505	0.60
Conv. tillage	−8.78	2.92	0.000869	0.25
Mulch ripping	−0.173	−0.113	0.000205	0.09
Tied ridging	−0.766	0.166	0.0001129	0.27

Table 3: The effect of rainfall amount and number of years of cultivation on maize yield under different tillage systems at Makoholi Contill site.

Treatment	Constant	Exp. nYear	Rainfall	R^2	Signif. level
Conv. tillage	−2.241	−0.0009493	0.011821	0.90	$P < 0.001$
Mulch ripping	−1.689	−0.0010092	0.011882	0.85	$P < 0.001$
Tied ridging	−1.382	−0.0006357	0.008736	0.82	$P < 0.025$

of cultivation. Soil loss also tended to increase by 0.9 t/ha under conventional tillage, 0.2 t/ha under mulch ripping, and 0.1 t/ha with every 1000 kJ/m^2 increase in rainfall energy.

The R^2 values were very low explaining only 25% of the variation under conventional tillage, 9% under mulch ripping, and 27% under tied ridging. While for all the treatments, increases in soil loss are given in relation to 1000 J/m^2; it should be noted that the average rainfall energy over the nine years is more than ten times this value, that is, 10 166 J/m^2.

3.8. Yield Equations. Yield was closely related to rainfall ($P < 0.001$) and was related exponentially to the year, for all treatments. The following equation was used for the determination of yield:

$$\mathrm{YI} = a + b\ \mathrm{EXP}(X_1) + cX_2, \tag{4}$$

where a is constant; b and c are coefficient; X_1 is nYEARS; X_2 is RAIN.

Under conventional tillage, yields were poorly and negatively correlated to year (−0.195), while better correlated to rainfall amount (0.551). The maize grain was estimated to increase by 1.2 t/ha for every 100 mm of rainfall received (Table 3). The highest yields were predicted under mulch ripping. A low and negative r value was also found between yield and year (−0.241), meaning that there is a decrease in yield with time. The correlation between yield and rainfall was lower than under conventional tillage (0.488). Crop yields were predicted to increase at 1.3 t/ha for every 100 mm of rainfall received. Yields under tied ridging were estimated to increase at 0.9 t/ha for every 100 mm of rainfall and also decrease exponentially to the year.

4. Discussion

The nine years data showed the erratic and unreliable nature of rainfall, both between and within the seasons, in the semi-arid region of southern Zimbabwe. Within the nine years of research, seasonal rainfall varied extensively from 164 mm to 994 mm. While the average rainfall amount (554 mm) still lay within the given range for this natural region (450–650 mm/yr), only one season recorded a seasonal total within this range and the other eight seasons recorded either higher or lower seasonal totals. The monthly and daily totals were equally variable, with some months recording much more than seasonal totals and some days recording more than monthly totals. This great variation in rainfall poses a high risk in agricultural production, as it becomes difficult to predict rainfall for any one season with certainty [15]. The erratic nature of rainfall also adds to the erosion problem [2]. Hudson [30] and Elwell and Stocking [31] reported that the rare and infrequent heavy storms cause severe erosion. The infiltration capacity of the soils, during such storms is exceeded and the high intensity causes crust formation [8], which leads to high runoff and soil losses.

The study results showed very high runoff and soil losses under the bare fallow and conventional tillage systems and negligible losses under mulch ripping and tied ridging. When the natural equilibrium of the soil is disturbed through cultivation—disruption of soil aggregates and increased aeration—the rate of organic matter mineralization is increased [32–34]. Organic matter is important in the soil aggregation and improves water infiltration and storage [35], thus its reduction results in higher rates of soil erosion. The very high topsoil losses with conventional tillage will eventually result in reduced plant available water and nutrients and thus productivity, as the soil depth is limited due to the presence of a stone line at around 50–80 cm depth [21]. Although plant nutrients can be compensated by additions of fertiliser or manure, in rain-fed agriculture, plant available water cannot be ameliorated. The physical properties, therefore, altered (e.g., water holding capacity) by soil erosion, are the most long term yield limiting factors [36]. Mulch ripping and tied ridging proved to be effective in reducing soil erosion. The mulch intercepts rainfall energy, thus increasing infiltration [37–39], while the rotting stover adds organic matter to the soil [40, 41]. The microdams under tied ridging enhance water ponding thus increasing water storage and reducing drainage [7]. The regression equations also support the dependence of runoff primarily on rainfall energy and the number of years of cultivation, $R^2 = 0.92$ for bare fallow and $R^2 = 0.80$ for conventional tillage. The R^2 values for the estimation of soil loss were generally lower than those found for the estimation of runoff, indicating that soil loss is also affected by other factors other than rainfall energy and time. Crop or ground cover and runoff volume and velocity have to be considered as well [40]. The ground cover effect is especially important under mulch ripping, while runoff volume and velocity are also drastically reduced under mulch ripping and tied ridging.

Mulch ripping had the highest yield average of 3.5 t/ha due to lower evaporative losses, especially during years with low rainfall or poor rainfall distribution. The soil moisture conserved ensured a better water supply to the crop during mid-season dry spells. Although runoff was drastically reduced under tied ridging, the water harvested in the microdams quickly drained away due to the very

high infiltration capacity, 40 cm/h according to Vogel [21], and low water holding capacity of these soils, which was found by Moyo and Hagmann [42] to be 10.3% by volume. Therefore, when rainfall events were widely spaced, the water harvested in micro-dams did not benefit the crop, as it would have drained away. The soil surface was also increased through ridging, thereby increasing evaporative losses [42]. The variables rainfall amount and time were found to adequately estimate yield; R^2 values ranged from 0.82 to 0.94 across all treatments. There was a direct linear relationship between yield and rainfall amount and maize grain yield increased by 0.9 (tied ridging), 1.2 (conventional tillage), and 1.3 t/ha (mulch ripping) with every 100 mm increment of rainfall. Yields decreased exponentially to the year under all the treatments indicating the reduction of productivity as soils are opened from virgin land.

Although the sandy soils are described as inherently infertile [43], the applied fertilisers seem to be adequate resulting in high yields during good rainfall seasons. Thus, rainfall, more than fertility, seems to be the most important yield-limiting factor. The study did not establish any conclusive yield variation among the treatments, except that under all tillage systems, there was a yield decline with the number of years of cultivation. Thus optimal fertiliser application and use of hybrid seed mask the effect of erosion on yield, as optimal crop growth can be achieved, if weather conditions are favourable. Thus, the fertiliser application is very much dependent on rainfall, so that rainfall becomes the most important factor influencing crop production. The effect of erosion on yield is of long term, while rainfall–thus soil moisture content—is the main short-term factor that influences yield.

5. Conclusions

The results of this study led to the following conclusions.

(i) Mulch ripping is the recommended tillage system for conserving soil and water and sustaining yields, while tied ridging can also be used satisfactorily to conserve soil and water but should be combined with mulch for better yields.

(ii) Conventional tillage practiced in the communal areas has to be replaced by conservation tillage techniques so as to reduce soil and water losses and maintain soil productivity.

(iii) Runoff and soil losses are a function of rainfall amount and intensity, number of years of cultivation and ground cover, that is, ploughing or minimum tillage; bare soil or soil covered with crops, weeds or mulch. The lower the intensity of tillage and the higher the ground cover, the better.

(iv) In semiarid regions where rainfall is limiting, yield is mostly dependent on the amount of rainfall and period of cultivation rather than fertility, if optimal fertilisers are used.

(v) Yield is a poor indicator of soil erosion when fertilisers and hybrid varieties are used as yield decline is masked. This is likely to be the case until such a time that yield declines even with the use of fertilisers and better cultivars, at which stage the damage might well be irreversible.

Acknowledgments

The author would like to express gratitude to GTZ for providing the much needed funding through CONTILL (Conservation Tillage), a collaborative project between GTZ and the Government of Zimbabwe (GoZ). Further acknowledgement goes to the GoZ, for providing the opportunity and research facilities. The author would like to thank all the CONTILL members from both Domboshawa and especially Makoholi site for their relentless support and input towards the success of this project.

References

[1] H. A. Elwell, "Sheet erosion from arable lands in Zimbabwe: prediction and control," in *Harare Symposium on Soil and Water Conservation*, pp. 429–438, Harare Zimbabwe, IAE, 1984.

[2] F. R. Troeh, J. A. Hobbs, and R. L. Donahue, *Soil and Water Conservation for Productivity and Environmental Protection*, Prentice-Hall, Englewood Cliffs, NJ, USA, 1980.

[3] J. Poesen and J. Savat, "Particle size separation during erosion by splash and runoff," in *Assessment of Erosion*, M. de Boodt and D. Gabriels, Eds., pp. 427–440, Wiley, Chichester, UK, 1980.

[4] R. Lal, "Monitoring soil erosion's impact on crop productivity," *Soil Erosion Research Methods*, pp. 187–201, 1988.

[5] R. J. Godwin, *Agricultural Engineering in Development: Tillage for Crop Production in Areas of Low Rainfall*, vol. 83, FAO Agricultural Services Bulletin, Rome, Italy, 1990.

[6] M. Stocking and L. Peake, "Soil conservation and productivity," in *Proceedings of the 4th International Conference on Soil Conservation*, pp. 399–438, University of Venezuale, Maracay, Venezuela, November 1985.

[7] N. W. Hudson, *Land Husbandry*, Batsford, London, UK, 1992.

[8] D. K. Cassel, C. W. Raczkowski, and H. P. Denton, "Tillage effects on corn production and soil physical conditions," *Soil Science Society of America Journal*, vol. 59, no. 5, pp. 1436–1443, 1995.

[9] G. Wrigley, *Tropical Agriculture: The Development and Production*, Macmillan Publishing, New York, NY, USA, 1992.

[10] Y. Le Bissonnais and M. J. Singer, "Crusting, runoff, and erosion response to soil water content and successive rainfalls," *Soil Science Society of America Bulletin*, vol. 56, no. 6, pp. 1898–1903, 1992.

[11] M. H. Beare, P. F. Hendrix, and D. C. Coleman, "Water-stable aggregates and organic matter fractions in conventional- and no-tillage soils," *Soil Science Society of America Journal*, vol. 58, no. 3, pp. 777–786, 1994.

[12] R. P. C. Morgan, *Soil Erosion and Conservation*, Longman Group UK Limited, Harlow, UK, 1986.

[13] H. A. Elwell, "Feasibility of modelling annual soil loss, runoff and maize yiled for the two research sites, Domboshawa and Makoholi. Projections to other natural regions in Zimbabwe. Testing of and contributions to SLEMSA," Consultancy Report, AGRITEX/ GTZ Conservation Tillage Project IAE, Harare, Zimbabwe, 1994.

[14] P.M. Grant, "Peasant farming on infertile sands," *Rhodesian Science News*, vol. 10, no. 10, pp. 252–254, 1976.

[15] S. Moyo, P. Robinson, Y. Katerere, S. Stevenson, and S. Gombo, *Zimbabwe's Environmental Dilemma. Balancing Resource Inequities*, ZERO, Harare, Zimbabwe, 1991.

[16] F. B. S. Kaihura, I. K. Kullaya, M. Kilasara, J. B. Aune, B. R. Singh, and R. Lal, "Impact of soil erosion on soil productivity and crop yield in Tanzania," *Advances in GeoEcology*, vol. 31, pp. 375–381, 1998.

[17] M. Rosenberg, "Koeppen climate classification," 2007, http://geography.about.com/od/physical geography/a/koeppen.htm.

[18] MNTR, *The National Conservation Strategy, Zimbabwe's Road to Survival*, The Government of Zimbabwe, Harare, Zimbabwe, 1987.

[19] J. G. Thompson and W. D. Purves, *A Guide to the Soils of Rhodesia*, vol. 1, Department of Research and Specialist Services, Harare, Zimbabwe, 1978.

[20] K. Nyamapfene, *Soils of Zimbabwe*, Nehanda Publishers, Harare, Zimbabwe, 1991.

[21] H. Vogel, "An evaluation of five tillage systems from smallholder agriculture in Zimbabwe," *Landwirtschaft der Tropen*, vol. 94, pp. 21–36, 1993.

[22] Anon, *Guide to Makoholi Experiment Station*, Department of Research and Specialist Services, Salisbury, UK, 1969.

[23] J. G. Thompson, *Report on the Soils of the Makoholi Experiment Station*, Department of Research and Specialist Services, Salisbury, UK, 1967.

[24] J. G. Thompson and W. D. Purves, *A Guide to the Soils of Zimbabwe*, vol. 3, Department of Research and Specialist Services, CHarare, Zimbabwe, 1981.

[25] CONTILL, "Conservation tillage for sustainable crop production systems," Working Document, IAE, Harare, Zimbabwe, 1990.

[26] F. E. Wendelaar and A. N. Purkis, "Recording soil loss and runoff from 300 m^2 erosion research field plots," Research Bulletin 24, Conex, Harare, Zimbabwe, 1979.

[27] A. Moyo, *Assessment of the effect of soil erosion on nutrient loss from granite-derived sandy soils under different tillage systems in Zimbabwe*, Ph.D. thesis, 2003.

[28] A. Munodawafa, "Assessing nutrient losses with soil erosion under different tillage systems and their implications on water quality," *Physics and Chemistry of the Earth*, vol. 32, no. 15–18, pp. 1135–1140, 2007.

[29] K. J. Olsen, "Modelling runoff and soil loss on coarse grained sandy soils at Domboshawa, Zimbabwe," Contill Project Research Report 12, IAE, Harare Zimbabwe, 1994.

[30] N. W. Hudson, *Erosion Research. Advisory Notes*, Conex, Salisbury, UK, 1958.

[31] H. A. Elwell and M. A. Stocking, "Rainfall parameters and a cover model to predict runoff and soil loss from grazing trials in the Rhodesian sandveld," *Grasslands Society for Southern Africa*, vol. 9, pp. 157–164, 1974.

[32] D. Schroeder, *Soils. Facts and Concepts*, FHAG, Berne, Switzerland, 1984.

[33] J. R. Salinas-Garcia, F. M. Hons, and J. E. Matocha, "Long-term effects of tillage and fertilization on soil organic matter dynamics," *Soil Science Society of America Journal*, vol. 61, no. 1, pp. 152–159, 1997.

[34] D. A. Angers, A. N'dayegamiye, and D. Cote, "Tillage-induced differences in organic matter of particle-size fractions and microbial biomass," *Soil Science Society of America Journal*, vol. 57, no. 2, pp. 512–516, 1993.

[35] R. H. Follett, S. C. Gupta, and P. G. Hunt, "Conservation practices: relation to the management of plant nutrients for crop production," in *Soil Fertility and Organic Matter as Critical Components of Production Systems*, pp. 19–51, Journal of Soil Science Society of America, Special Publication 19, Madison, Wis, USA, 1987.

[36] B. Lowery and W. E. Larson, "Erosion impact on soil productivity," *Journal of Soil Science Society of America*, vol. 59, no. 3, pp. 647–648, 1995.

[37] J. E. Adams, "Influence of mulches on runoff, erosion and soil moisture depletion," *Proceedings of Soil Science Society of America*, vol. 30, pp. 110–114, 1966.

[38] P. G. Braithwaite, "Conservation tillage- planting systems," *Rhodesian Farmer*, vol. 10, pp. 25–32, 1976.

[39] H. A. Elwell, "An assessment of soil erosion in Zimbabwe," *Zimbabwe Science News*, vol. 19, no. 3-4, pp. 27–31, 1986.

[40] W. L. Hargrove, "Crop residue management in the South-East," in *Proceedings of the National Conference on Crop Residue Management for Conservation*, Lexington, Ky, USA, 1991.

[41] D. C. Reicosky, W. D. Kemper, G. W. Langdale, C. L. Douglas, and P. E. Rasmussen, "Soil organic matter changes resulting from tillage and biomass production," *Journal of Soil and Water Conservation*, vol. 50, no. 3, pp. 253–261, 1995.

[42] A. Moyo and J. Hagmann, "Growth-effective rainfall in maize production under different tillage systems in semi arid conditions and shallow granitic sands of Southern Zimbabwe," in *Proceedings of the 13th International Soil Tillage Research Organisation*, Aalborg, Denmark, July 1994.

[43] P. Grant, "Peasant farming on infertile sands," in *The Rhodesia Science News*, vol. 10, pp. 252–254, Rhodesia Scientific Association, Harare, Zimbabwe, 10 edition, 1981.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 476821, 7 pages
doi:10.1155/2012/476821

Research Article

Organic Matter and Barium Absorption by Plant Species Grown in an Area Polluted with Scrap Metal Residue

Cleide Aparecida Abreu,[1] Mariana Cantoni,[2] Aline Renée Coscione,[1] and Jorge Paz-Ferreiro[3]

[1] *Centro de Solo e Recursos Ambientais, IAC, Avenida Barão de Itapura, 1481, 13020-902 Campinas, SP, Brazil*
[2] *Programa de Pós-Graduação em Agricultura Tropical e Subtropical, IAC, 13020-902 Campinas, SP, Brazil*
[3] *Departamento de Edafología, Universidad Politécnica de Madrid, 28004 Madrid, Spain*

Correspondence should be addressed to Jorge Paz-Ferreiro, jpaz@udc.es

Received 18 October 2011; Revised 27 December 2011; Accepted 8 January 2012

Academic Editor: Philip White

The effect of organic matter addition on Ba availability to *Helianthus annuus* L., *Raphanus sativus* L., and *Ricinus communis* L. grown on a Neossolo Litólico Chernossólico fragmentário (pH 7.5), contaminated with scrap residue was evaluated. Four rates (0, 20, 40, and 80 Mg ha^{-1}, organic carbon basis) of peat or sugar cane filter, with three replicates, were tested. Plant species were grown until the flowering stage. No effect of organic matter addition to soil on dry matter yield of oilseed radish shoots was observed, but there was an increase in sunflower and castor oil plant shoots when sugar cane filter cake was used. The average Ba transferred from roots to shoots was more than 89% for oilseed radish, 71% for castor oil plants, and 59% for sunflowers. Organic matter treatments were not efficient in reducing Ba availability due to soil liming.

1. Introduction

Accumulation of some chemical elements in the environment is of great concern because they can reach concentrations that may cause risks to human health and to the environment. Their concentration in soils depends on lithogenic and pedogenic processes, but also on anthropogenic activities. Soil pollution is a serious problem in many countries around the world. In São Paulo State, Brazil, since 2002, when the first survey was performed by the local environmental agency, more than 1600 contaminated areas have been identified [1].

The extensive industrial use of barium (Ba) adds up to the release of Ba in the environment and, as a result, Ba concentrations in air, water, and soil may be higher than naturally occurring concentrations on many locations [2–5]. Recently, it was observed that successive sewage sludge applications increased soil Ba concentration and accumulation in maize plants grown in the State of São Paulo [6]. Some research has shown probable Ba toxicity in plants, but such studies were short term and performed in nutrient solution [7, 8]. Ba is an alkaline earth element which occurs as a trace metal in igneous and sedimentary rocks. In nature it occurs mainly as low soluble minerals such as barite ($BaSO_4$) and witherite ($BaCO_3$). Ba solubilization and, consequently, the release of Ba^{2+} ions may occur under specific conditions. It has been shown to happen in acidic conditions [9], in the absence of oxygen, or even due to microbial action [10–13]. In contrast, Ba precipitates as a sulfate and/or carbonate salt in neutral or basic pH conditions. Therefore, the mobility of Ba is negligible in neutral or basic pH conditions, thus, reducing the risks of leaching and harmful health effects.

The application of lime and the addition of organic materials are considered the most efficient options to reduce heavy metal availability in soils [14–16]. The use of organic matter in chemically degraded areas can also be beneficial since plant development in such areas is frequently affected, exposing the soil to physical degradation.

Peat and humic materials concentrate reduced extractable Zn, Cu, Pb, and B in soil and mustard shoots [14] while liming reduced the available concentrations of Cd, Pb, Cu, and Zn in soils as well as its content in velvetbean shoots

Table 1: Chemical composition of the organic *materials**.

Source	pH	E.C.	O.C.	C/N	P	K	Ca	Mg	B	Cu	Fe	Mn	Zn
		$dS\,m^{-1}$	$g\,kg^{-1}$				$g\,kg^{-1}$					$Mg\,kg^{-1}$	
Sugar cane filter cake	7.5	0.9	263.7	12	10.3	2.3	16.2	3.7	21	60	5900	557	141
Peat	5.5	0.2	163.1	24	0.8	1.4	1.7	1.7	16	45	6300	47	36

E.C.: electrical conductivity; O.C.: organic carbon.
*Total elements concentration obtained by extraction with a mix of nitric and perchloric acids [17], results presented are the average of six replicates.

[16].The bioavailability of heavy metals to soybean and black-oat cultures was close to zero, when 8 $Mg\,ha^{-1}$ sewage sludge, flue dust, and aqueous lime was applied to soil surface in no-till system [15]. However, lime and organic matter addition to the Ba contaminated soil and its availability to soil and its plants absorption have not received a great deal of attention and few information on the topic has been reported. The organic matter complexation of Ba ions can lead to insoluble species, decreasing the availability of Ba and enabling the growth of vegetation in highly contaminated areas [3]. Consequently, Ba effects on plant grown in soils containing Ba still needs to be further investigated.

The aim of the present work was to evaluate the effect of application rates of peat and sugar cane filter cake on Ba concentration in soil and its potential availability to sunflowers (*Helianthus annuus* L.), oilseed radish (*Raphanus sativus* L.), and castor oil plants (*Ricinus communis* L.) grown in a soil (pH 7.5) contaminated with scrap metal residue.

2. Material and Methods

In 2005, automobile scrap "shredder residue" was applied to the soil of an agricultural area of approximately 3 ha located in Piracicaba (22°42′30″ S, 47°38′01″ W), São Paulo State, Brazil. The residue's metal content, obtained by the SW-846 3051 method [18] was, in $mg\,kg^{-1}$: 170 of B, 7.4 of Cd, 2497 of Cu, 775 of Pb, 178 of Cr, 153 of Ni, 8157 of Zn, and 920 of Ba. Residue addition was performed based on the supposition that it may provide Zn and Cu to sugar cane crops, and the residue was incorporated into the soil at a depth of 30 cm. The local environmental agency (CETESB) later verified that the area was contaminated by heavy metals (copper and zinc) and boron. Lime (10 $Mg\,ha^{-1}$) was added to the soil in order to reduce heavy metals mobility and potential leaching. The soil in this area is classified as Lithic Udorthent [19].

Soil samples were taken from the 0–20 cm depth layer, dried at room temperature, and sieved to 2.0 mm. The soil fertility attributes were measured as follows: pH_{CaCl2} = 7.5; MO = 30.5 $g\,dm^{-3}$; P_{resin} = 43.3 $mg\,dm^{-3}$; K_{resin} = 2.6 $mmol_c\,dm^{-3}$; Ca_{resin} = 294 $mmol_c\,dm^{-3}$; Mg_{resin} = 59 $mmol_c\,dm^{-3}$; CEC = 364 $mmol_c\,dm^{-3}$; H + Al = 9.0 $mmol_c\,dm^{-3}$; V = 98% according to [20]. For the determination P, K, Ca a mixed (cationic and anionic) ion exchange resin method (Amberlite IRA 120 and Amberlite IRA 400) was used to simulate elements soil availability. It employs a ratio of 2.5 of soil per 2.5 cm^3 of resin, which is kept in contact for 16 hours. The elements adsorbed by resin are washed away with 50 mL of a 0.8 $mol\,L^{-1}$ NH_4Cl + 0.2 $mol\,L^{-1}$ HCl, producing an extract where the elements are determined. Some of total elements content in the soil were measured by SW-846 3051 method [20] as follows, in $mg\,kg^{-1}$: 241 of Ba, 62 of B, 4.3 of Cd, 335 of Cu, 332 of Pb, 88.2 of Cr, 53.6 of Ni, and 2998 of Zn. This procedure consists of adding 10 mL of HNO_3 to 500 mg soil in a teflon capped vessel in a laboratory microwave system (CEM, Mars 5 model, Xpress vessels). The extraction is performed by raising the temperature to 170°C for 5 min and keeping it at this temperature during 10 minutes.

The experiment was carried out in a greenhouse at Campinas (São Paulo State, Brazil) in plastic pots (5 dm^{-3}). The following plant species: sunflowers (*Helianthus annuus* L.), oilseed radish (*Raphanus sativus* L.), and castor oil plants (*Ricinus communis* L.) were selected for the experiment due to previous works showing them to be tolerant to high concentration of heavy metals and boron in soil [21–24].

The experimental design was in randomized complete blocks with four rates (0, 20, 40, and 80 $Mg\,ha^{-1}$, organic carbon basis) of two organic matter sources (peat and sugar cane filter cake), with three replicates. The treatments were applied at ($g\,pot^{-1}$): 0.0, 37.9, 75.8, and 151.6 g of sugar cane filter cake per pot, respectively and 0.0, 61.3, 122.6, and 245.2 g of peat per pot. The chemical compositions of the peat and sugar cane filter cake (Table 1) were obtained by determination of elements in a 0.5 mg of sample extractred with nitric perchloric acids (3 : 1 ratio) [17].

The soil/organic materials were carefully homogenized and incubated at room temperature for 20 days with soil moisture maintained at 60% water holding capacity (WHC). The pots received 200 $mg\,kg^{-1}$ of P as triple superphosphate (41% P_2O_5) and the samples were homogenized and incubated for an additional 15 days after the sowing of seeds.

Three sunflowers and castor oil plants and ten oilseed radish were grown per pot. Deionized water was supplied by weighing the pots daily and adding the water needed to maintain 60% WHC. Nitrogen (30 $mg\,N\,kg^{-1}$ soil) was applied as ammonium nitrate (32% N) on emerging seedlings and again 15 days later.

Sunflower and oilseed radish were harvested 65 days after sowing, while castor oil plants were harvested 74 days after sieving. Shoots were separated from roots at harvest. The flowers were also separated when the oilseed radish and sunflowers were harvested. Roots were sieved, washed

and soaked for 90 min in a solution of 0.02 mmol L^{-1} EDTA (disodium salt). After soaking, the oilseed radish roots were washed again with distilled water. Plant samples were washed, dried, and weighed and then digested using HNO_3/H_2O_2 in a CEM Mars 5 microwave oven and analyzed for macro- and micronutrients, and barium, lead, cadmium, chromium, and nickel.

Soil samples collected after incubation were air-dried and sieved through a 2 mm mesh screen and then characterized for total and available metal contents. Available Ba content was analyzed using Mehlich-3 method (CH_3COOH 0.2 mol L^{-1} + NH_4NO_3 0.25 mol L^{-1} + NH_4F 0.015 mol L^{-1} + HNO_3 0.015 mol L^{-1} + EDTA 0.001 mol L^{-1} at pH 2.5) by agitation of five cm^3 of soil and 20 mL the Mehlich-3 solution for 5 min [25]. The availability of several nutrients (P, K, Ca, and Mg) was evaluated by the ion exchange method [20].

Ba transported from soil to shoots was evaluated using the transfer factor (TF) as follows: TF = PC (mg kg^{-1})/SC (mg kg^{-1}), where CP is the Ba concentration in the whole plant (root and shoot), and CT is the concentration of Ba in the soil [26]. The ability of each species to translocate Ba from the roots to the shoots was calculated using the following translocation index (TI): TI (%) = QPA (mg pot^{-1})/QAP (mg pot^{-1}) $\times$ 100, where QPA is the element accumulation in the shoots, and QAP is the element accumulation in the whole plant (shoots and roots) [26].

The plant efficiency for the removal of elements (removal factor, E) was calculated using the following equation: E(%) = QPA (mg pot^{-1})/QR (mg pot^{-1}) $\times$ 100, where QR is the amount of Ba to be removed from the soil (mg pot^{-1}) [27]. When considering a 75% reduction of Ba concentration in the soil as a target, the time (T, in years) needed for Ba removal was calculated as follows: $T = (R/E)/\mathrm{NC}$, where R is the percentage of Ba reduction in the soil, E is the removal factor, and NC is the number of crop cycles/year (considered as 1 cycle/year).

The data were submitted to analyses of variance (ANOVA), and the mean values were compared according to Tukey's test ($P \leq 0.05$). When significant, the results obtained with the different concentrations of organic material were also examined using regression analysis (linear and quadratic models tested).

3. Results and Discussion

The concentration of Ba found during the soil characterization (241 mg kg^{-1}) was close to the intervention levels (300 mg kg^{-1}) established by the Environmental Agency of the State of São Paulo [28]. Although the concentration of zinc, copper, and boron in this area is also worrisome, the use of plants that could help to remediate the soil was studied on previous works and was not significant in the present work as also discussed below [24, 29].

Mehlich-3 available Ba increased in soils amended with 40 Mg ha^{-1} of peat and 80 Mg ha^{-1} of sugar cane filter cake, with an average of 32.9 mg dm^{-3} in the soils amended with sugar cane filter cake and 36.2 mg dm^{-3} in the soils amended with peat (Table 2), which corresponded to a 12.3% and 14.4% recovery, respectively. The recovery found in this study was lower than the one reported by others which was in a range from 50% to 78% [30]. The correlation between extractable Ba and soil organic carbon was 0.96 $P < 0.05$ (sugar cane filter cake) and 0.95 $P < 0.05$ (peat). However, no significant correlation was found between extractable Ba in the soil and the Ba accumulated in all of the plant tissues.

Table 2: Ba extracted from soil with the Mehlich-3 *method**.

Rate	Sugar cane filter cake	Peat
Mg ha^{-1}	mg dm^{-3}	
0	31.5 a	34.9 a
20	32.9 a	34.1 a
40	32.9 a	36.7 ab
80	34.6 b	39.3 b
Average	32.9 A	36.2 A

*Results presented are the average of 3 replicates. Means followed by the same letter are not significantly different by the Tukey's test at $P \leq 0.05$. Upper case letters, in columns, compare treatments and lower case letters, in lines, compare rate of amendments.

In most plants, the concentration of Ba ranges from 4 to 50 mg kg^{-1} [31], and concentrations of 200 and 500 mg kg^{-1} are considered to be slightly toxic or toxic, respectively [32]. The average Ba concentrations in the shoots after addition of sugar cane filter cake or peat were as follows: 44.47 or 50.97 mg kg^{-1}, respectively, in sunflowers; 29.68 or 30.03 mg kg^{-1}, respectively, in castor oil plants; and 77.23 or 74.46 mg kg^{-1}, respectively, in oilseed radish (Table 3). Similar results have been reported for the same plant species grown in Rhodic Hapludox using $BaSO_4$ additions of 0, 150, and 300 mg kg^{-1}. The plant tissue Ba concentrations found in the present study were higher than previously reported (21.3 mg kg^{-1} for sunflowers, 19.4 mg kg^{-1} for mustard plants, and 10.6 mg kg^{-1} for castor oil plants [30]).

However, Ba concentrations in this study were less than those reported by Suwa et al., 2008 [8] who observed that high Ba concentrations affected soybeans and resulted in reduced development, stomatal closing, and reduced photosynthetic activity. In contrast, Ba accumulation in maize plants grown in soil with much lower Ba concentrations (soil pH in the range of 5.1 to 5.7) has also been reported and no phytotoxic symptoms or nutritional imbalance correlations were observed [6].

In this study no effects or symptoms of phytotoxicity were found in the plants. Moreover, no nutritional imbalance was observed in the soil samples. In the presence of high Ca concentrations, such as those of the area studied, Ba can precipitate [9]. The absence of phytotoxic in this study might bel explained by the high levels of available Ca (294 $mmol_c$ dm^{-3}).

Shoot dry matter yields varied depending on the treatment and plant species (Table 4). Among the species tested, oilseed radish was the least affected by the treatments, and the peat addition promoted a higher dry matter yield in the oilseed radish roots. Sunflowers and castor oil plants showed similar results regarding shoot and root dry matter production, which were both higher when the sugar cane

Table 3: Barium in plant parts of the tested species according to the rate of organic material *treatment**.

	Rate	Oilseed radish				Sunflower				Castor oil plant	
		Root	Straw (S)	Pod (P)	S + P	Root	Straw (S)	Flower (F)	S + F	Roots	Shoots
	Mg ha^{-1}					mg kg^{-1}					
Sugar cane filter cake	0	111.0 a	56.6 a	19.2 a	75.8 a	76.7 a	31.1 a	18.5 a	49.56 a	39.8 b	30.1 ba
	20	109.7 a	56.0 a	21.3 a	77.3 a	76.0 a	32.2 a	15.2 b	47.4 ba	53.1 a	31.7 a
	40	105.0 a	58.5 a	20.7 a	79.2 a	75.0 a	27.8 ba	15.5 ba	43.4 bc	46.5 ba	30.9 a
	80	103.8 a	56.8 a	19.7 a	76.6 a	75.0 a	25.6 b	11.9 c	37.5 c	50.2 b	26.0 b
Average		107.4 A	57.0 A	20.2 A	77.2 A	75.7 A	29.2 A	15.3 B	44.5 B	47.4 A	29.7 A
Peat	0	107.4 a	58.1 a	17.0 a	75.1 a	73.3 a	31.2 a	21.4 a	52.6 a	44.9 a	533.7 a
	20	108.3 a	59.4 a	17.6 a	77.0 a	76.0 a	32.0 a	21.4 a	53.4 a	44.3 a	627.9 a
	40	105.5 a	57.8 a	13.8 a	71.6 a	72.0 a	30.9 a	22.8 a	53.7 a	41.2 a	560.2 a
	80	109.7 a	58.8 a	15.4 a	74.2 a	76.7 a	25.9 b	18.3 b	44.2 b	42.7 a	618.3 a
Average		107.7 A	58.5 A	16.0 A	74.5 B	74.5 A	30.0 A	21.0 A	51.0 A	43.3 B	585.0 A

*Result presented are the average of 3 replicates. Means followed by the same letter are not significantly different by the Tukey's test at $P \leq 0.05$. Upper case letters, in columns, compare plant tissues and lower case letters, in columns, compare rate of amendments.

Table 4: Dry matter yield for different plant parts of the species tested according to the rate of organic material *treatment**.

	Rate	Oilseed radish				Sunflower				Castor oil plant	
		Root	Straw (S)	Pod (P)	S + P	Roos	Straw (S)	Flower (F)	S + F	Roots	Shoots
	Mg ha^{-1}					mg kg^{-1}					
Sugar cane filter cake	0	0.8 a	10.8 a	5.0 a	15.7 a	2.2 a	12.7 b	4.9 a	17.6 b	4.7 a	18.5 b
	20	0.7 a	11.7 a	6.3 a	17.9 a	2.7 a	15.9 ba	3.4 b	19.3 ba	5.8 a	19.5 ba
	40	0.8 a	12.2 a	5.6 a	17.7 a	3.2 a	18.3 a	3.9 ba	22.2 a	5.6 a	21.3 c
	80	0.7 a	12.5 a	5.7 a	18.2 a	2.4 a	15.0 ba	5.2 a	20.2 ba	5.6 a	20.7 bc
Average		0.8 B	11.8 A	5.6 A	17.4 A	2.6 A	15.5 A	4.4 A	19.8 A	5.4 A	20.0 A
Peat	0	0.9 a	12.1 a	4.8 a	16.9 a	1.6 a	12.7 a	3.2 a	15.8 a	4.4 a	18.5 b
	20	1.0 a	11.5 a	6.0 a	17.5 a	2.4 a	14.4 a	3.1 a	17.5 a	4.8 a	19.7 ba
	40	0.9 a	12.6 a	5.0 a	17.6 a	2.0 a	13.3 a	3.3 a	16.7 a	5.1 a	18.9 b
	80	1.0 a	12.0 a	5.9 a	17.9 a	2.5 a	15.5 a	3.0 a	18.5 a	5.4 a	20.8 a
Average		0.9 A	12.0 A	5.5 A	17.5 A	2.1 B	14.0 B	3.2 B	17.1 B	4.9 B	19.5 B

*Results presented are the average of 3 replicates. Means followed by the same letter, are not significantly different by the Tukey's test at $P \leq 0.05$. Upper case letters, in columns, compare plant tissues and lower case letters, in columns, compare rate of amendments.

filter cake was used. Sugar cane filter cake has a low C/N ratio (Table 1), which may explain the trend to be a more useful source of nutrients than peat, as sugar cane is more easily decomposed than peat. Figure 1 shows that the increased organic material positively affected the shoot dry matter yield in the castor oil plants. However, despite the statistical significance of the regression models, from the agronomic or ecological point of view no marked quantitative difference in dry matter production among treatments would be enough to recommend one amendment over the other since the increase in dry matter production was, overall, discrete.

The addition of organic material to the soil affected differently Ba concentration among the three plant species (Table 3). Oilseed radish did not show a significant effect, but an increase in Ba in the castor oil plant roots was observed after the addition of sugar cane filter cake, from 39.8 mg/kg (no addition) to 50.2 mg/kg (80 Mg ha^{-1}), which corresponded to an increase of 26%. In castor bean shoots, Ba increased from 533.7 mg/kg in the control to 618 mg/kg in soils amended with 80 Mg ha^{-1} peat respectively, which represented an increase of 16%. In sunflowers, the Ba concentrated in the flowers and straw+flower tissues was higher when the sugar cane filter cake was used. Moreover, the increase in the organic material rate (sugar cane filter cake and peat) resulted in a linear decrease in the Ba concentration in the flowers of the sunflower plants (Figure 2), up to 15% with sugar cake filter cake and 16% with peat addition.

When peat was used a negative correlation was observed for Ba and P in the castor oil plants ($r = -0.55$) and sunflower tissues ($r = -0.48$) (Table 5).The same trend was observed for K in the oilseed radish ($r = -0.83$) and Ca in the castor oil plants ($r = -0{,}67$) and sunflowers ($r = 0.52$) with the use of sugar cane filter cake (Table 5). A nutritional imbalance of Ca, K, and S in the presence of Ba has been reported by several authors. These reports suggest that the imbalance is related to the plant species [7, 8, 31].

Table 5: Correlation between Ba and P, K and Ca concentrations in shoots.

Element	Oilseed radish		Sunflower		Castor oil plant	
	Sugar canefilter cake	Peat	Sugar cane filter cake	Peat	Sugar cane filter cake	Peat
P	0.51*	0.07NS	0.76*	−0.48NS	−0.78*	0.55*
K	−0.52*	−0.83*	0.12NS	−0.00NS	0.78*	0.42NS
Ca	−0.05NS	0.13NS	−0.52*	−0.04NS	0.73*	−0.67

*Significant at $P < 0.05$ and NS: not significant.

Table 6: Transfer factor (TF) and translocation index (TI) of Ba in the tested species.

Treatment	Mg ha^{-1}	TF			TI (%)		
		Oilseed radish	Sunflower	Castor oil plant	Oilseed radish	Castor oil plant	Sunflower
Sugar cane filter cake	0	0.70	0.47	0.26	89.23	74.64	59.22
	20	0.70	0.46	0.32	90.63	66.71	60.92
	40	0.69	0.45	0.29	90.32	71.77	60.45
	80	0.65	0.41	0.28	90.52	65.70	60.13
Average		0.69	0.45	0.29	90.20	69.57	60.21
Peat	0	0.75	0.52	0.30	89.01	72.86	59.59
	20	0.71	0.49	0.29	88.24	74.76	59.87
	40	0.68	0.48	0.27	89.52	72.69	59.37
	80	0.71	0.47	0.28	88.10	72.98	59.82
Average		0.71	0.49	0.29	88.71	73.35	59.66

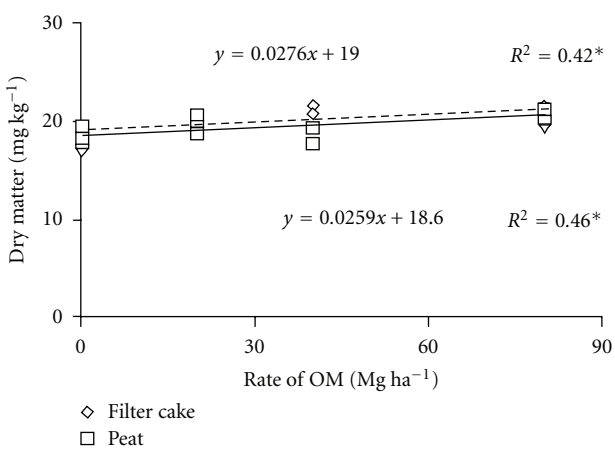


Figure 1: Effect of increasing concentrations of organic materials on shoot dry matter yield in castor oil plants shoots (d.w.). Significant at $P < 0.05$.

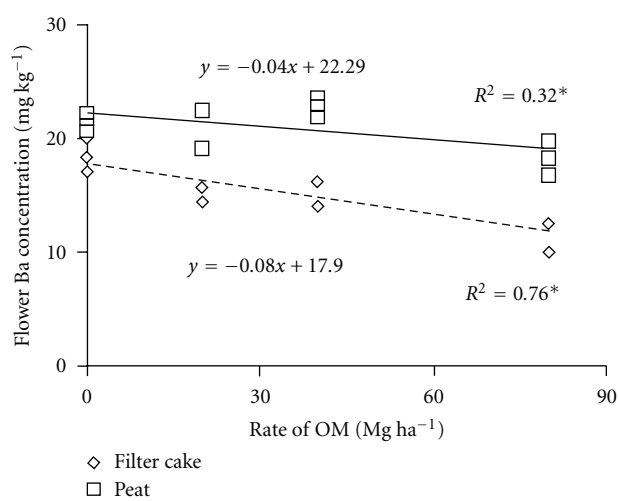


Figure 2: Effect of increasing concentrations of organic materials on Ba concentration in the flowers of sunflowers (d.w.). Significant at $P < 0.05$.

The transfer index (calculated as the Ba shoot concentration divided by the total Ba in the soil) decreased as follows: oilseed radish (0.70) > sunflowers (0.47) > castor oil plants (0.29). The average Ba transfer from the roots to the shoots in the oilseed radish, castor oil plants, and sunflowers was found to be 89%, 71%, and 59%, respectively. These values indicated that the Ba was highly mobile in the xylem of the oilseed radish and castor oil plants. From the total Ba transfer values, at least 50% of the Ba shoot concentration was found in the flowers of the sunflowers, and 35% of the Ba shoot concentration was found in the pods of the oilseed radish (Table 3). Ba concentrations in flowers with the addition of sugar cane filter cake and peat were as follows: 20.2 and 16.0 mg kg^{-1}, respectively, for oilseed radish; 15.3 and 21.0 mg kg^{-1}, respectively, for sunflowers. A high Ba mobility has also been observed in cotton and white beets

during development and flowering, whereas a large amount of Ba accumulation has been reported in the leaves of corn ($105.7\,\mathrm{mg\,kg^{-1}}$) when compared with the grains of corn ($1.05\,\mathrm{mg\,kg^{-1}}$) [6, 33].

The obtained transfer factors (lower than 1) suggested that the tested species were inadequate in accumulating or extracting Ba from soil (Table 6). Similarly, low transfer factors have also been reported for castor oil plants, sunflowers, and mustard plants [30]. Indeed, these authors [30] reported that none of the plants grown in soils containing Ba ranging from 132.3 to $1{,}130\,\mathrm{mg\,kg^{-1}}$ were able to accumulate measurable concentrations of Ba, thus, highlighting the low transfer of this element from soil to plants [34].

A decreasing trend for Ba transfer to oilseed radish and sunflowers was found when the sugar cane filter cake concentration increased (Table 6). In addition to improving the physical and chemical conditions of the soil, organic ligands are promising in the mitigation of heavy metal contaminated soils. Peat and a concentrate containing humic substances from coal favor mustard development in a contaminated soil due to the mitigation of Zn, Cu, Mn, Pb, and B by the organic ligands [14].

4. Conclusions

Under the conditions studied the elevated soil pH reached due to liming overcame the organic matter addition effect and determined the barium availability and its absorption by the plant species grown in the area polluted with scrap metal residue. This is suggested by the absence of phytotoxic effects on plants, the moderate Ba accumulation in shoots compared to the usual content of Ba in plants, the small effect of organic matter treatments on plants dry matter yields, and finally the levels of Ba extracted by Mehlich-3.

Acknowledgment

The authors would like to thank Fundação de Apoio a Pesquisa do Estado de São Paulo (FAPESP) for the financial support with the 2006/60987-0 Research Grant.

References

[1] Cetesb, "Relatório de Totalização de áreas contaminadas e rea- bilitadas-Dezembro de 2010," 2011, http://www.cetesb.sp.gov.br/areas-contaminadas/relacoes-de-areas-contaminadas/15-publicacoes.

[2] J. A. Ippolito and K. A. Barbarick, "Biosolids affect soil barium in a dryland wheat agroecosystem," *Journal of Environmental Quality*, vol. 35, no. 6, pp. 2333–2341, 2006.

[3] L. C. S. Merlino, W. Melo, F. G. Macedo et al., "Barium, cadmium, chromium and lead in maize plants and in an oxisol after eleven years of sewage sludge applications," *Revista Brasileira de Ciência do Solo*, vol. 34, pp. 2031–2039, 2010.

[4] E. Simon, M. Braun, A. Vidic, D. Bogyo, I. Fabian, and B. Tothneresz, "Air pollution assessment ion elemental concentration of leaves tissue and foliage dust along an urbanization gradient in Vienna," *Environmental Pollution*, vol. 159, pp. 1229–1233, 2011.

[5] E. Nogaj, J. Kwapulinski, and H. Misiolek, "Pharyngeal Tonsil as new biomarker of pollution on example of barium," *Polish Journal of Environmental Studies*, vol. 20, pp. 161–172, 2011.

[6] T. A. R. Nogueira, W. J. deMelo, I. M. Fonseca, M. O. Marques, and Z. He, "Barium uptake by maize plants as affected by sewage sludge in a long-term field study," *Journal of Hazardous Materials*, vol. 181, no. 1-3, pp. 1148–1157, 2010.

[7] M. Llugany, C. Poschenrieder, and J. Barceló, "Assessment of barium toxicity in bush beans," *Archives of Environmental Contamination and Toxicology*, vol. 39, no. 4, pp. 440–444, 2000.

[8] R. Suwa, K. Jayachandran, N. T. Nguyen, A. Boulenouar, K. Fujita, and H. Saneoka, "Barium toxicity effects in soybean plants," *Archives of Environmental Contamination and Toxicology*, vol. 55, no. 3, pp. 397–403, 2008.

[9] C. A. Menzie, B. Southworth, G. Stephenson, and N. Feisthauer, "The importance of understanding the chemical form of a metal in the environment: the case of barium sulfate (barite)," *Human and Ecological Risk Assessment*, vol. 14, no. 5, pp. 974–991, 2008.

[10] F. Baldi, M. Pepi, D. Burrini, G. Kniewald, D. Scali, and E. Lanciotti, "Dissolution of barium from barite in sewage sludges and cultures of Desulfovibrio desulfuricans," *Applied and Environmental Microbiology*, vol. 62, no. 7, pp. 2398–2404, 1996.

[11] A. A. Carbonell, R. Pulido, R. D. DeLaune, and W. H. Patrick, "Soluble barium in barite and phosphogypsum amended Mississippi River alluvial sediment," *Journal of Environmental Quality*, vol. 28, no. 1, pp. 316–321, 1999.

[12] G. A. Ulrich, G. N. Breit, I. M. Cozzarelli, and J. M. Suflita, "Sources of sulfate supporting anaerobic metabolism in a contaminated aquifer," *Environmental Science and Technology*, vol. 37, no. 6, pp. 1093–1099, 2003.

[13] C. M. Davidson, M. D. Gibson, E. Hamilton, B. H. MacGillivray, J. Reglinski, and E. Rezabal, "The long-term environmental behaviour of strontium and barium released from former mine workings in the granites of the Sunart region of Scotland, UK," *Chemosphere*, vol. 58, no. 6, pp. 793–798, 2005.

[14] G. C. G. Dos Santos and A. A. Rodella, "Effect of sources of organic matter in the alleviation of the toxic effects of B, Zn, Cu, Mn and Pb to Brassica Juncea," *Revista Brasileira de Ciencia do Solo*, vol. 31, no. 4, pp. 793–804, 2007.

[15] J. C. Corrêa, L. T. Büll, W. D. S. Paganini, and I. A. Guerrini, "Heavy metal exchangeable in an Oxisol with surface application of flue dust, aqueous lime, sewage sludge and limestone," *Pesquisa Agropecuaria Brasileira*, vol. 43, no. 3, pp. 411–419, 2008.

[16] E. E. C. de Melo, C. W. A. do Nascimento, A. C. Q. Santos, and A. S. da Silva, "Availability and fractionation of Cd, Pb, Cu, and Zn in soil as a function of incubation time and pH," *Ciencia e Agrotecnologia*, vol. 32, no. 3, pp. 776–784, 2008.

[17] O. C. Bataglia, A. M. C. Furlani, J. P. F. Teixeira, and J. R. Gallo, *Métodos de Análise Química de Plantas*, Boletim Técnico, 78, Instituto Agronômico, Campinas, Brazil, 1983.

[18] United States Environmental Protection Agency, "Method 3051: microwave assisted acid digestion of sediments, sludges, soil and soils," 2009, http://www.epa.gov/epaoswer/hazwaste/test/3_series.htm.

[19] Empresa Brasileira de Pesquisa Agropecuária, *Brazilian System of Soil Classification*, vol. 306, Embrapa-Centro Nacional de Pesquisa de Solos, Rio de Janeiro, Brazil, 2nd edition, 2006.

[20] B. van Raij, J. C. Andrade, H. Cantarella, and J. A. Quaggio, *Análise Química Para Avaliação da Fertilidade de Solos Tropicais*, Instituto Agronômico, Campinas, Brazil, 2001.

[21] C. De Freitas Zeitouni, R. S. Berton, and C. A. De Abreu, "Phytoextraction of cadmium and zinc from an oxisol contaminated with heavy metals," *Bragantia*, vol. 66, no. 4, pp. 649–657, 2007.

[22] A. M. M. Pires, C. A. Abreu, A. R. Coscione, V. A. Silva, and N. P. Ramos, "Initial growth of sunflower in soils with high concentrations of boron and heavy metals," in *Proceedings of the 17th International Sunflower Conference*, vol. 2, pp. 315–318, Córdoba, Spain, 2008.

[23] G. C. G. dos Santos, A. A. Rodella, C. A. de Abreu, and A. R. Coscione, "Vegetable species for phytoextraction of boron, copper, lead, manganese and zinc from contaminated soil," *Scientia Agricola*, vol. 67, no. 6, pp. 713–719, 2010.

[24] R. A. B. Jorge, C. A. de Abreu, C. A. de Andrade, and O. A. de Camargo, "Filter cake and peat as amendments of contaminated soil with residue of scrap rich in boron," *Bragantia*, vol. 69, no. 2, pp. 467–476, 2010.

[25] A. Mehlich, "Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant," *Communications in Soil Science and Plant Analysis*, vol. 15, no. 12, pp. 1409–1416, 1984.

[26] A. D. Abichequer and H. Bohnen, "Efficiency of phosphorus uptake, translocation and utilization in wheat varieties," *Revista Brasileira de Ciência do Solo*, vol. 22, pp. 21–26, 1998.

[27] S. Lubben and D. Sauerbeck, "The uptake and distribution of heavy metals by spring wheat," *Water, Air, and Soil Pollution*, vol. 57-58, pp. 239–247, 1991.

[28] Companhia de Tecnologia de Saneamento Ambiental, "Guiding values for soils and groundwater in the State of Sao Paulo," 2011, http://www.cetesb.sp.gov.br/Solo/relatorios/tabela_valores_2005.pdf.

[29] M. B. Gabos, G. Casagrande, C. A. Abreu, and J. Paz-Ferreiro, "Uso da matéria orgânica como mitigadora de solo multicontaminado e do girassol como fitoextratora," *Revista Brasileira de Engenharia Agrícola e Ambiental*, vol. 15, no. 12, pp. 1298–1306, 2011.

[30] A. R. Coscione and R. S. Berton, "Barium extraction potential by mustard, sunflower and castor bean," *Scientia Agricola*, vol. 66, no. 1, pp. 59–63, 2009.

[31] F. M. Chaudhry, A. Wallace, and R. T. Mueller, "Barium toxicity in plants," *Communications in Soil Science and Plant Analysis*, vol. 8, no. 9, pp. 795–797, 1977.

[32] I. Pais and J. R. Jones, *The Handbook of Trace Elements*, St. Lucie Press, Boca Ratón, Fla, USA, 1998.

[33] Z. En, A. Vasidov, V. V. Tsipin, T. Tillaev, and G. I. Jumaniyazova, "Study of element uptake in plants from the soil to assess environmental contamination by toxic elements," *Nuclear Instruments and Methods in Physics Research A*, vol. 505, no. 1-2, pp. 462–465, 2003.

[34] J. Pichtel, K. Kuroiwa, and H. T. Sawyerr, "Distribution of Pb, Cd and Ba in soils and plants of two contaminated sites," *Environmental Pollution*, vol. 110, no. 1, pp. 171–178, 2000.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 817107, 10 pages
doi:10.1155/2012/817107

Research Article

Cadmium and Zinc Concentration in Grain of Durum Wheat in Relation to Phosphorus Fertilization, Crop Sequence and Tillage Management

Xiaopeng Gao and Cynthia A. Grant

Agriculture and Agri-Food Canada, Brandon Research Centre, 2701 Grand Valley Road, Brandon, MB, Canada R7A 5Y3

Correspondence should be addressed to Cynthia A. Grant, cynthia.grant@agr.gc.ca

Received 16 November 2011; Revised 3 January 2012; Accepted 7 January 2012

Academic Editor: Philip White

Field experiments were conducted at two locations in Manitoba, Canada, to determine the effect of crop rotation, phosphorus (P) fertilization and tillage on grain yield and grain concentrations of Cd and Zn in durum wheat (*Triticum durum* L.). Compared to conventional tillage (CT), reduced tillage (RT) management decreased grain Cd and increased grain yield and grain Zn in half of the site-years. The type of preceding crops of spring wheat-flax or canola-flax had little influence. Rate and timing of P application had little effect on grain Cd, but increasing P rate tended to decrease grain Zn. No interactive effect was detected among tested factors. Grain Zn was not related to grain Cd, but positively to other nutrients such as Fe, Mn, P, Ca, K, and Mg. Both grain Zn and Fe correlated positively with grain protein content, suggesting protein may represent a sink for micronutrients. The study suggested that the tillage management may have beneficial effects on both grain yield and quality. Phosphorus fertilizer can remain available for subsequent crops and high annual inputs in the crop sequence may decrease crop grain Zn. Understanding the environment is important in determining the impact of agricultural management on agronomic and nutrient traits.

1. Introduction

Cadmium (Cd) accumulation in soils and cereal crops and its transfer to the human diet is a widespread problem around the world. Durum wheat (*Triticum durum* L.) is of particular concern because it accumulates more Cd than the other commonly grown cereals with accumulation increasing in the order of rye < barley < oats < bread wheat < durum wheat [1]. Cadmium concentration in durum wheat grain harvested on Canadian prairies have been reported to range from less than 50 to more than 300 μg kg^{-1} [2], at times exceeding the 200 μg kg^{-1} limit set by the Codex Alimentarius Commission [3]. In addition, approximately 2.1×10^6 ha durum wheat, which occupies 10% of worldwide durum production area, is grown in the western Prairie region of Canada [4]. Therefore, there is a desire in the Canadian farming industry to control the Cd levels in the durum grain, either by improved agricultural management practices [5] or by breeding low Cd-accumulating cultivars [6, 7].

Accumulation of metal elements in crop grains can be regulated by several physiological processes, including uptake from the soil solution, root-to-shoot translocation, and retranslocation into the grain during maturation. Zinc (Zn) and Cd are chemically similar and can compete for common transport mechanisms for uptake and translocation in the crop [5]. In contrast to Cd, Zn concentrations in grains of cereal crops such as wheat are often lower than desired level as sources of minerals for the human diet [8]. High consumption of cereal-based foods with low Zn concentration can result in Zn malnutrition in human. On the Canadian prairies, Zn concentration in wheat grain generally ranges between 20 and 30 mg kg^{-1} [9]. These concentrations are not adequate for human nutrition in diets with wheat constituting the main source of essential minerals [8]. Therefore, it is desirable to increase the Zn concentration in wheat grain to reduce the risk of Zn deficiency in the human diet.

Phosphorus fertilizer is a major input for crop production. The fertilizer efficiency, however, is often quite low

during the season of application because of the low mobility and rapid immobilization of phosphate after being applied to the soil. Application of P fertilizer can influence soil Cd availability and Cd accumulation in crops, either by direct addition of Cd as a contaminant in P fertilizers, or by indirect impact on soil properties, plant nutrition, and plant growth [5, 10]. For example, application of high-Cd P fertilizers led to increases in soil and plant Cd in many long-term field studies [11–13]. Phosphorus fertilizer is also frequently reported to reduce Zn concentration in crops and potential mechanisms include interactions between Zn and P in the soil [14], less translocation of Zn from the roots to the shoots [15], metabolic disorder within plant cells [16], and dilution effect [17]. In a short-term growth chamber study [18], the immediate increase of Cd accumulation in durum wheat following phosphate application was mostly due to increased Cd uptake and translocation by decreasing Zn accumulation in durum wheat induced by P fertilization, rather than to a direct addition effect. Similar results were also reported in other studies [19, 20], where using reagent-grade phosphate increased Cd concentration, but decreased Zn concentration of durum wheat. Though these studies showed the long-term or short-term effect of P fertilization on phytoavailability of Cd and Zn in soil-plant system, there is little information available on the effect of the residual P from P fertilizer that was applied to preceding crops.

On the Canadian prairies, rotation of wheat with oilseeds or pulse crops is widely adopted and has been reported to increase grain yield and quality of subsequent crops through reduced disease incidence, improved water use efficiency, soil physical and chemical properties, and enhanced soil ecological environments [21–23]. Several studies have addressed the impact of crop rotation on Cd concentration in crops. Oliver et al. [24] reported a greater Cd concentration in spring wheat grown after lupin than after cereal and proposed the effect was due to increased Cd availability by rhizosphere acidification and root release of citric acid in lupin. In a 3-year field trial in Manitoba,Canada, seed Cd concentration in flax was lower when grown after bread wheat (*Triticum aestivum* L.) than after canola (*Brassica napus* L.), which was attributed to an increase in mycorrhizal colonization or to the lower Cd concentration in the decomposing wheat residue relative to canola residue [25].

Reduced tillage practices are widely adopted by farmers on the Canadian prairies because of the benefits in conserving soil moisture, preventing soil erosion and reducing cost of production [26]. Difference in tillage management may have an impact on pH and nutrient stratification in the soil profile or residue decomposition [27], and consequently affect availability of Cd and Zn. The effect of tillage management on uptake of Cd and Zn in crops has been investigated in several field studies and the results are inconsistent. Tiller et al. [28] reported a 30% greater Cd concentration in spring wheat grain under direct drilling, compared to reduced tillage. Franzluebbers and Hons [29] reported that soil under a no-tillage system contained greater amounts of extractable Zn than under a conventional tillage system. In some other field studies, however, long-term tillage practices did not consistently influence Cd concentration in soils and crops [25, 30, 31].

Information on the influence of agricultural management practices on crop yield and on Cd and Zn concentration and accumulation in crops is important in order to select management practices that optimize both crop productivity and quality. Although many studies have evaluated the impact of tillage system, crop sequence and fertilizer practices in the short term, effects of these management practices can persist and have impacts on subsequent crops. For example, P fertilizer not used by a crop in the year of production will remain in the soil and may exert residual effect on the following crop. Effects of tillage may increase over time as changes in soil organic matter and nutrient stratification intensify. However, information on the persistence of management effects on the yield and quality of later crops in a cropping system are limited.

Based on field studies conducted at two locations over a four-year period, yield and concentrations of Cd and Zn of flax were shown to be influenced by tillage system, type of preceding crop and P application to the preceding crop [25, 32], but information is lacking about whether such effects can persist to influence subsequent crops. Therefore, to address this question, the field study described by Grant et al. [25, 32] was extended for an additional year to determine if preceding crop, P fertilization and tillage system would have a continuing effect on the yield and quality of the durum wheat grown in the third year of the cropping sequence.

2. Materials and Methods

2.1. Experimental Treatment and Cultural Practices. Field plot experiments were conducted at two different locations near Brandon, Manitoba. Both sites were classified as clay loam, Orthic Black Chernozemic soils. Initial characteristics of the soils are provided in Table 1 [25]. Sampling and determination methods for soil characteristics were also described by Grant et al. [25]. One site (MZTRF) had been under reduced tillage (RT) management for 6 years before the study was initiated, while the second site (BRC North) had been under conventional tillage (CT) management until the establishment of the study. Neither of the soils was industrially contaminated and as such both are representative of soils commonly used for durum wheat production on the Canadian prairies.

The study was conducted over a 5 yr period from 1999–2003, with 3 yr cropping cycles being conducted at each of two locations in 1999-2000-2001, 2000-2001-2002, and 2001-2002-2003.

In the first phase of the 3 yr cycle, either spring wheat (cv. AC Barrie) or canola (cv. G3295) was seeded under CT or RT, with P fertilizer treatments of 0, 11 and 22 kg P ha^{-1} as monoammonium phosphate (MAP). In the second phase, each plot where wheat or canola had been grown in the preceding season was divided into two subplots. Flax (cv. AC Emerson) was seeded into each subplot, with P fertilizer treatment of 0 or 11 kg P ha^{-1}. Tillage treatment was kept the same as the preceding season. For the first and second phases, spring wheat, canola, and flax were harvested at maturity

Table 1: Soil characteristics of the experimental sites prior to initiation of the study.

Soil property	Soil depth (cm)	BRC North			MZTRF		
		2001*	2002	2003	2001	2002	2003
pH	0–15	7.82	8.37	7.83	8.43	7.87	7.54
	15–30	—	—	8.17	—	—	7.71
	30–60	—	—	8.43	—	—	7.95
EC (μS cm^{-1})	0–15	290	289	603	261	293	585
	15–30	218	290	537	228	228	495
	30–60	299	419	560	226	276	523
N (mg kg^{-1})	0–15	8.34	3.78	2.50	7.92	4.19	7.50
	15–30	3.31	2.56	2.75	3.42	2.03	3.19
	30–60	2.41	3.56	1.17	2.92	1.81	1.81
P (mg kg^{-1})	0–15	12.53	11.63	6.75	9.36	11.34	9.94
K (mg kg^{-1})	0–15	267	202	259	9.36	298	290
DTPA-Cd (μg kg^{-1})	0–15	102.6	73.3	72.0	89.6	134.8	103.6
DTPA-Zn (mg kg^{-1})	0–15	0.77	0.60	0.80	1.01	1.08	1.20

* Soil samples were taken in the first phase of the study, prior to seeding the preceding canola and wheat.

with a plot combine and the crop residue was chopped and returned to the plot where it had originated. Details on the cultural practices and sampling are available by Grant et al. [25, 32].

In the third phase, each subplot where flax had been grown in the preceding season was seeded with durum wheat (cv. AC Avonlea). Ammonium nitrate was banded in all subplots prior to seeding at a rate of 100 kg N ha^{-1}, adjusted for the amount of N added in the MAP. Each subplot where no P was applied in the preceding season received 11 kg P ha^{-1} side-banded as MAP, whereas the subplots where 11 kg P ha^{-1} was applied in the preceding season received no P. This provided a comparison between the effects of P applied in the current season and the residual effects of P applied in the previous year. Again, tillage treatment was continued as in the preceding season. Therefore, as shown in Table 2, the experimental design was comprised of two tillage system, two crop rotation, and six P fertilizer treatments, for a total of 24 combinations. The 3 yr cycle of spring wheat-flax-durum wheat or canola-flax-durum wheat was repeated three times at each site. This paper reports data on durum wheat over three growing seasons (2001–2003) on two sites, for a total of 6 site-years.

The treatments, with four replicates, were arranged as a randomized complete block, with a split-plot layout for a total of 96 subplots at each location. Main blocks were the two tillage treatments and subplots were 12 combinations of crop rotation and P fertilizer treatments. Tillage blocks were initiated in the first year of the study and continued until the completion of the study. The RT consisted of only fertilizer banding and seeding operations and the CT received two tillage passes in the fall and one pass in the spring with a cultivator equipped with tine harrows. The preceding crops were established on a different area within the tillage blocks each year. Registered herbicides were applied as required and according to recommendations of the Manitoba Guild to Crop Protection.

Table 2: Description of the experimental design in the study.

Treatments	Phases		
	1st	2nd	3rd
Tillage	CT	CT	CT
	ZT	ZT	ZT
Crop rotation	Spring wheat	Flax	Durum wheat
	Canola	Flax	Durum wheat
P fertilization (kg MAP ha^{-1})	0	0	25
	0	25	0
	25	0	25
	25	25	0
	50	0	25
	50	25	0

2.2. Sampling and Analysis. Prior to seeding of the durum wheat, 0–15 cm depth soil core samples were collected in each plot. Soil samples were air-dried and sieved. Available P was extracted from the soil with 0.5 mol L^{-1} $NaHCO_3$ [33]. Cadmium and Zn were extracted using DTPA [34]. Soil pH and EC were determined by a glass electrode using a 1 : 2 soil : water ratio. Concentrations of P and Zn in the extract were measured with an ARL 3410 ICP unit, and Cd on a Varian 300/400 atomic absorption spectrophotometer at a wavelength of 228.8 nm using a graphite furnace with deuterium correction (detection limit 0.01 μg Cd L^{-1}). Reliability of the analysis was assessed by including certified soil reference materials and repeated samples in each set of soil extracts. Measured concentrations matched the stated ranges in the standards.

At crop maturity, grain yield of durum wheat was measured by harvesting the center four rows of the plot using a plot combine. Grain was dried at 30°C to a constant moisture

level and the weight recorded for each plot. Grain samples were ground and digested with a boiling acid mixture (HNO_3 + $HClO_4$) for Cd and Zn [35]. The concentrations of Zn in the digest were determined on an ARL 3520 inductively coupled plasma (ICP) and Cd concentrations were determined on a Varian 300/400 atomic absorption spectrophotometer (AAS) at a wavelength of 228.8 nm using a graphite furnace with deuterium correction (detection limit 0.01 μg Cd L^{-1}). The reliability of the analysis was assessed by including certified plant reference materials and duplicate samples in each set of digests.

2.3. Statistical Analysis. Analysis of variance (Proc Mixed of SAS) was conducted on data separated by site-year. All interactions among fixed effect variables (tillage, crop rotation and P fertilization) were tested in the models and results showed that all interactions for grain yield and grain concentrations of elements were not significant. Therefore, only main effects of treatments were presented in this study. Means of the treatments were compared using protected least significance difference (LSD) at the 5% level of probability. Relationships between parameters were determined using linear regression analysis. Principal component analysis (PCA) was used to describe relationships among variables and was performed using the correlation matrix method. Loading plots were generated using principal components 1 and 2 as axes, and variables were plotted along the axes. PCA was conducted with Minitab Statistical Software (Minitab 15) and other analyses were performed with SAS program, release 9.1 (SAS Institute Inc., Cary, USA).

3. Results

3.1. Grain Yield. Grain yield was greater with RT than CT in 3 site-years and less in 1 site-year (Table 3). Crop rotation and P fertilization, however, had little effect on grain yield. Grain yield was greater when canola-flax rather than wheat-flax were the preceding crops in only one of 6 site-years, being BRC-North site in 2002. Differences in grain yield were only observed among P treatments at MZTRF in 2002. There was no significant interaction among the treatments in their effect on grain yield.

3.2. Grain Cd Concentration and Accumulation. Grain Cd concentration ranged from 66.6 to 150.4 μg kg^{-1} dry weight (56.6 to 127.5 μg kg^{-1} fresh weight based on approximately 15% water content). Grain Cd was less with RT than CT in 3 site-years and greater in 1 site-year (Table 4). Accumulation of Cd in the grain was also less with RT in 2 of 6 site-years and greater in 1 site-year. Effects of crop rotation and P fertilizer on grain Cd concentration and accumulation were insignificant in all but one site-year, being MZTRF-2001, where grain Cd concentration and accumulation was greater when following canola-flax than wheat-flax. Accumulation of Cd was increased by P fertilizer only at MZTRF in 2002, and was not affected in other site-years. There was no significant interaction among the treatments in their effect on Cd concentration and accumulation in the grain.

3.3. Grain Zn Concentration and Accumulation. Grain Zn concentration ranged from 23.0 to 42.3 mg kg^{-1} dry weight. Grain Zn concentration was higher with RT than CT in 3 of 6 site-years (Table 5). Compared to CT, RT increased grain Zn accumulation in 2001 and 2003, but decreased it in 2002 on BRC-North site. In contrast, grain Zn accumulation was not affected by tillage management on MZTRF site. Grain Zn concentration and accumulation in durum wheat was higher when following wheat-flax than canola-flax only at MZTRF site in 2003, and was not affected in other site-years. Application of P fertilizer to either the previous crop or the current durum wheat crop did not affect grain Zn concentration. With increased application rate, however, grain Zn concentration showed a significantly decreasing trend in 2 of 6 site-years. Accumulation of Zn was not affected by P management in any site-years. Similar to Cd, grain Zn concentration and accumulation was not affected by any interactions among the treatments.

3.4. Correlation Analysis. Correlation analysis (Table 6) showed that grain Cd was positively correlated with grain Fe, P, K, Mg, and negatively correlated with grain Mn, Cu, and Ca. A slightly negative but significant relationship was observed between yield and grain Cd ($r = -0.135$, $P < 0.001$). Both grain Fe and Zn were highly significantly positively correlated ($P < 0.001$) with other elements including Mn, P, Ca, K, and Mg and with protein content. Grain Fe, but not Zn, was negatively correlated with grain yield and positively correlated with grain Cd. Grain Fe and Zn were also positively correlated. Grain P was positively correlated with other measured element concentrations and negatively correlated with grain yield. The full dataset used for correlation matrix analysis is presented in Supplementary Table 1 (available online at doi:10.1155/2012/817107).

Grain concentrations of Cd, Zn and P across site-years correlated significantly and positively with their extractable concentrations in soils (Table 6). Durum grain Cd was significantly related to soil extractable Cd concentration ($r = 0.339$, $P < 0.001$), as well as to soil extractable P concentration ($r = 0.309$, $P < 0.001$). Grain concentrations of Zn, Fe, and P, but not Cd, were negatively related with soil pH.

PCA of the dataset across 6 site-years extracted two major principal components, explaining 78% of the total variation in data (Figure 1). PC1 explained 54.4% of the variation, and was loaded positively with grain Zn, Mn, P, K, Mg, Fe, and soil P, and negatively with grain yield and soil Cd. PC2 explained 23.6% of the variation, and was loaded positively with grain Ca and negatively with grain Cd and Cu. The PCA showed strong associations between grain protein and grain concentrations of minerals such as Zn, Mn, P, K, Mg, and Fe. In the biplot, the long and outside-scattered distribution of the six site-year vectors indicated that the environments were likely to have a great influence on the measured parameters. Grain Cd concentration was greatest in BRC-2003 and least in MZTRF-2002. In contrast, grain Zn concentration on either site was generally greater in 2001 than in other years. The least grain yield was recorded in BRC-2001 even though the soil had a relatively high P availability.

Table 3: Grain yield (kg ha^{-1}) of durum wheat as affected by tillage system, crop rotation, and P fertilization on at two sites over 3 years.

		BRC North			MZTRF		
		2001	2002	2003	2001	2002	2003
Tillage							
Conventional		1790	3086	2696	3282	3221	2912
Reduced		1888	2835	3042	3235	3566	2870
LSD		71	137	269	79	139	101
Crop rotation							
wheat-flax-durum		1823	2851	2888	3234	3393	2933
Canola-flax-durum		1854	3070	2851	3283	3394	2849
LSD		70	137	269	79	139	101
P fertilization							
0-0-25		1894	2993	2927	3291	3300	2900
0-25-0		1733	2801	2677	3294	3131	2895
25-0-25		1839	2919	2860	3217	3552	2869
25-25-0		1832	3043	2900	3142	3284	2837
50-0-25		1871	2931	2939	3265	3546	2945
50-25-0		1865	3077	2911	3343	3546	2901
LSD		123	237	466	137	241	174
Analysis of variance							
Source	DF			Pr ≥ F			
Tillage	1	0.0079	0.0005	0.0127	n.s.	<0.0001	n.s.
Crop rotation	1	n.s.	0.0022	n.s.	n.s.	n.s.	n.s.
P fertilization	5	n.s.	n.s.	n.s.	n.s.	0.0015	n.s.

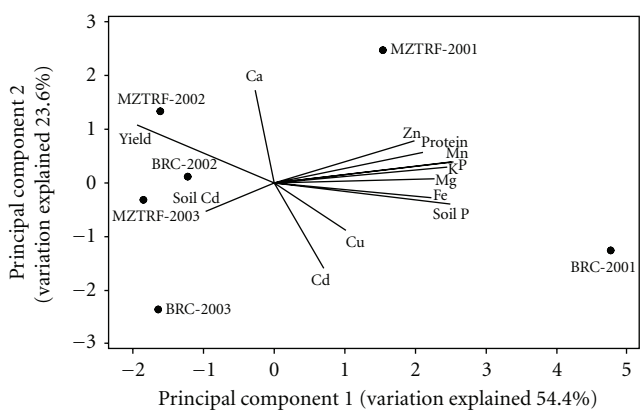


Figure 1: Principal component analysis (based on correlation matrix) of grain yield, grain concentrations of Cd, Zn, Cu, Fe, Mg, K, P, Mn, Ca, grain protein concentration, and selected soil properties (soil P and soil Cd) across 6 site-years. Biplot vectors are trait factor loadings for principal component 1 (PC1) and PC2. Data at each site-year are averaged across P fertilizer treatments.

4. Discussion

Grain yield was relatively low at the BRC-North site in 2001 (Table 3, Figure 1), due to adverse weather conditions, but was average for the area in the other site-years. The range of Cd concentrations in durum grain in this study was also within the expected range, being comparable to that reported in our previous studies on the Canadian prairies [31, 36], and well below the proposed Codex maximum limit in wheat grain [3]. This suggests the durum wheat production in the area is generally safe for human consumption. However, Zn concentrations in the grain were generally less than the biofortification target of 40–60 mg kg^{-1}, which is required to counteract human Zn deficiency in many parts of the world [37]. Therefore, it is desirable to increase Zn concentration in durum grain, either by genetic improvement or agricultural practices.

Of the factors evaluated, tillage management showed the greatest influence on grain yield, as well as on grain Cd and Zn concentrations and accumulations in durum wheat. Compared to CT, RT significantly increased grain yield and Zn, and decreased grain Cd in half of the site-years (Tables 3, 4, 5). The decrease in Cd, however, was neither necessarily related to the increase in Zn, nor to the increase in grain yield, as suggested by the nonrelationship or weak relationship among these three variables (Table 6). It should be noted that the observed effects of tillage on grain Cd and Zn were not expected because our previous studies suggested that tillage system had very limited influence on either Cd [29] or Zn [38] concentrations in durum grain. Also, in the second phase of the current study where the flax was the tested crop, tillage management also did not exert any consistent effect on flaxseed concentrations of Cd and Zn [25]. It is unclear why there was an effect of tillage in this growing season

Table 4: Grain Cd concentration (μg kg^{-1}) and accumulation (mg ha^{-1}) of durum wheat as affected by tillage system, crop rotation, and P fertilization on at two sites over 3 years.

		Grain Cd concentration (μg kg^{-1})						Grain Cd accumulation (mg ha^{-1})					
		BRC North			MZTRF			BRC North			MZTRF		
		2001	2002	2003	2001	2002	2003	2001	2002	2003	2001	2002	2003
Tillage													
Conventional		120.1	122.8	144.2	82.5	86.1	83.5	216	380	401	269	279	247
Reduced		120.4	70.1	119.8	81.8	61.3	99.8	228	205	380	267	221	289
LSD		9.1	13.9	15.9	4.8	8.7	11.3	20	48	80	18	31	37
Crop rotation													
wheat-flax-durum		117.3	92.0	131.1	78.6	70.5	90.8	215	270	394	255	239	270
Canola-flax-durum		123.3	100.9	132.9	85.6	76.9	92.5	229	316	387	282	260	266
LSD		9.1	13.9	15.9	4.8	8.7	11.3	20	48	80	18	31	37
P fertilization													
0-0-25		117.1	89.9	132.2	83.1	68.4	88.6	223	272	400	274	226	258
0-25-0		110.3	97.5	116.2	80.3	72.9	87.9	191	287	318	265	226	257
25-0-25		116.9	94.8	133.9	82.3	81.8	94.6	216	283	393	265	291	277
25-25-0		120.7	98.9	126.0	78.5	68.3	83.1	223	307	369	248	226	239
50-0-25		131.1	99.7	150.4	87.1	84.3	111.1	244	304	459	285	295	329
50-25-0		125.5	97.8	133.2	81.5	66.6	84.6	234	302	404	272	237	247
LSD		15.7	24.1	27.5	8.3	15.1	19.5	35	84	139	32	54	64
Analysis of variance													
Source	DF						Pr ≥ F						
Tillage	1	n.s.	<0.001	0.003	n.s.	<0.001	0.005	n.s.	<0.001	n.s.	n.s.	0.004	0.025
Crop rotation	1	n.s.	n.s.	n.s.	0.005	n.s.	n.s.	n.s.	n.s.	n.s.	0.005	n.s.	n.s.
P fertilization	5	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	0.014	n.s.

but not in the previous seasons. Some studies suggested that tillage management can have an impact on metal availability in soils by its influence on soil properties such as pH, organic carbon content and cation exchange capacity [29, 39]. This assumption, however, is not supported in this study because soil concentrations of extractable Cd and Zn were not consistently affected by tillage management (data not present). Instead, the difference in mycorrhizal association could play a role. Compared to RT, annual soil disturbances produced by CT may reduce mycorrhizal colonization in crops [40] and thus influence root uptake of Cd [41] or Zn [42]. In addition, tillage effects may vary from year to year because of differences in weather conditions. Under prairie conditions, RT tends to have greater benefits under warmer, drier conditions where moisture conservation become important. Similarly, seasonal differences in weather may interact with tillage effects on both root distribution in different soil layers and rate of decomposition of crop residues, to influence metal uptake. While the impact of tillage is not consistent across different studies on the Canadian prairies [25, 31, 32, 36], the RT practices may be beneficial to farmers in many regions not only because of the benefits in conserving soil moisture, preventing soil erosion and reducing the cost of production [26], but also for potential improvements in grain yield and grain quality, as has been shown in the current study.

There was no consistent difference in grain yield or grain concentration of Cd or Zn between durum wheat following spring wheat-flax as compared to canola-flax. In contrast, in the second phase of this study, flaxseed following canola had significantly higher Cd concentration and lower Zn concentration than following spring wheat [25]. These results suggest that the impact of spring wheat or canola as a preceding crop was significant only on the following crop, and did not persist to affect the third crop in the sequence. Potential mechanisms explaining preceding crop effect on flax were discussed in Grant et al. [25], including difference in Cd and Zn content in the crop residue which was returned to the soil after harvest, difference in mycorrhizal association between mycorrhizal spring wheat and nonmycorrhizal canola, as well as difference in root-induced rhizosphere chemical changes between canola and wheat.

In spite of the low initial soil P concentration at the two locations, there was no statistically significant main effect on grain yield from the different amounts or timings of P fertilization through the three-year cropping sequence (Table 3). However, in 4 of 6 site-years, there was a statistical or numerical advantage from 160 to 250 kg ha^{-1} of grain yield with application of the P to the durum wheat rather than in the preceding crop at the lowest rate of application (Table 3), a difference that was significant at the $P < 0.02$–0.28 rate using contrast analysis (data not present). This differential

Table 5: Grain Zn concentration (mg kg^{-1}) and accumulation (g ha^{-1}) of durum wheat as affected by tillage system, crop rotation, and P fertilization on at two sites over 3 years.

		Grain Zn concentration (mg kg^{-1})						Grain Zn accumulation (g ha^{-1})					
		BRC-North			MZTRF			BRC-North			MZTRF		
		2001	2002	2003	2001	2002	2003	2001	2002	2003	2001	2002	2003
Tillage													
Conventional		38.0	26.6	23.7	35.6	37.0	25.3	68.2	82.4	65.9	116.9	119.3	74.6
Reduced		40.1	26.0	26.3	37.4	35.6	24.9	75.5	74.2	82.9	120.9	126.6	72.1
LSD		1.9	1.3	2.5	1.8	2.3	1.5	4.7	6.5	13.9	5.9	8.9	6.4
Crop rotation													
wheat-flax-durum		39.5	26.2	25.7	36.9	36.2	26.1	72.0	74.9	77.1	119.6	123.1	77.3
Canola-flax-durum		38.7	26.5	24.3	36.1	36.4	24.1	71.7	81.7	71.7	118.3	122.8	69.3
LSD		1.9	1.3	2.5	1.8	2.3	1.5	4.7	6.5	13.9	5.9	8.9	6.4
P fertilization													
0-0-25		37.9	27.4	24.9	38.0	35.7	25.6	71.9	82.3	76.1	125.1	118.1	74.8
0-25-0		42.3	27.4	26.5	37.4	38.7	27.4	73.2	77.6	73.4	123.0	121.0	80.6
25-0-25		39.8	25.6	23.7	37.1	34.9	23.8	72.8	74.7	71.1	119.4	124.1	69.3
25-25-0		39.4	26.9	26.7	35.7	37.0	26.1	72.2	82.9	79.3	112.0	122.5	74.7
50-0-25		37.9	25.0	23.4	35.7	34.3	23.0	71.4	73.5	71.8	116.5	121.1	68.4
50-25-0		37.3	25.6	24.8	35.2	37.0	24.6	69.5	78.8	74.8	117.6	130.7	72.1
LSD		3.3	2.2	4.3	3.1	4.1	2.6	8.2	11.3	24.1	10.3	15.4	11.2
Analysis of variance													
Source	DF						Pr ≥ F						
Tillage	1	0.036	n.s.	0.034	0.043	n.s.	n.s.	0.003	0.014	0.018	n.s.	n.s.	n.s.
Crop rotation	1	n.s.	n.s.	n.s.	n.s.	n.s.	0.013	n.s.	0.042	n.s.	n.s.	n.s.	0.015
P fertilization	5	0.034	n.s.	n.s.	n.s.	n.s.	0.015	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.

was much lower or nonexistent when higher amounts of P were applied through the cropping sequence. It appears that if the level of input through the cropping sequence is low, there may be a small yield advantage to providing some P as a starter near the seed row in the current crop. The benefit of the use of starter P diminishes as the P input through the rotation decreases.

In other field studies, that is, in Sweden [43] and in the Canadian prairies [13], Cd concentration in grain and seed of several crops consistently increased with increasing P application, which was attributed to either the direct addition of the Cd contained in the fertilizer or the indirect effect on soil chemical properties. The MAP used in this study, however, contained only 3.7 mg Cd kg^{-1} and application of 75 kg fertilizer over 3 growing seasons would add only 278 mg of Cd to the soil per hectare, making it unlikely to have a strong effect on Cd concentration in crop grain. Similarly, in studies in the USA, long-term application of fertilizers containing less than 5 mg Cd kg^{-1} did not increase Cd concentration in soil and crops [44]. In 2 of 6 site-years, P fertilizer treatment exerted a significant effect on grain Zn concentration, with the trend being that grain Zn decreased with increased application rate over the three growing seasons (Table 5). Similarly, in the second phase of this study, seed Zn concentration of flax decreased with application of P in half of the site-years [25]. The P-induced inhibition of Zn accumulation is frequently reported and may be related to the P-Zn interaction in soil [14] and depression in mycorrhizal association caused by increased P supply [45].

The impacts of the agricultural management on grain yield and grain quality were highly unstable. Their performance varied across locations and years. For example, the reduced tillage significantly improved grain yield and grain quality as compared to the conventional tillage in half of the site-years. The effect, however, was absent or even opposite in other site-years. Also, increasing P fertilizer rate decreased grain Zn in 2 site-years, but not in the others. Therefore, the environmental conditions such as location or meteorological factors are important determinants for agronomic and nutrient traits. This assumption is also confirmed by the wide range of the site-year criteria in the PCA biplot (Figure 1). Still, only one wheat genotype was tested in the current study, neglecting the potential genotype by environment interactions, as been shown in other studies concerning grain Cd [46] and grain Zn [47]. Hence, greater efforts taking care of genotype by environment interactions are needed to produce improved grain quality for human health.

Results of the linear regression and PCA showed grain Zn concentration was correlated positively to other nutrients such as Fe, Mn, P, Ca, K, and Mg, but was not correlated to grain Cd (Table 6, Figure 1). The relationships favor the possibility of producing durum grain with moderately high nutrients, while maintaining low concentration of Cd. Also, both grain Zn and Fe concentrations correlated positively

Table 6: Correlation coefficients (r) of grain concentrations of Cd, Zn, Fe and P to other elements concentrations, grain yield, grain protein concentration, and soil characteristics (The full dataset is presented in Supplementary Table 1).

	Cd	Zn	Fe	P
Grain				
Zn	−0.022	—	0.507***	0.688***
Fe	0.294***	0.507***	—	0.598***
Mn	−0.150***	0.239***	0.441***	0.504***
Cu	−0.134***	0.096*	0.067	0.101*
P	0.223***	0.688***	0.598***	—
Ca	−0.365***	0.408***	−0.017	0.168***
K	0.081*	0.755***	0.579***	0.833***
Mg	0.191***	0.491***	0.390***	0.691***
Protein	0.036	0.425***	0.529***	0.636***
Yield	−0.135***	−0.014	−0.302***	−0.292***
Soil				
pH	−0.080	−0.635***	−0.193**	−0.255***
EC	−0.094	−0.047	−0.041	−0.095
Olsen-P	0.309***	0.324***	0.537***	0.553***
DTPA-Cd	0.339***	0.213***	0.235***	0.012
DTPA-Zn	0.044	0.523***	0.193***	0.201**

*, **, *** indicate significant at 0.05, 0.01, and 0.001, respectively.

with grain protein content. This finding is consistent with other field trials [48] and suggested a possible link between grain protein and the levels of the two micronutrients.

5. Conclusions

In summary, of the factors evaluated, tillage had the most consistent effect on grain concentrations of Cd and Zn. Compared to CT, RT increased grain yield, grain Zn, and decreased grain Cd in half of the site-years and should therefore be recommended in tested area. The effect of growing wheat rather than canola as a preceding crop on crop yield or trace element concentration did not persist to affect durum wheat grown as the third crop in the sequence. There was evidence of a small impact of starter P on grain yield, but only if the levels of P applied through the cropping sequence were low, indicating that P fertilizer applied in preceding years can remain plant-available, reducing the requirements for P input in following crops. Rate and timing of P application had little effect on grain concentration of Cd, but increasing P rate tended to decrease grain Zn concentration. The linear and multivariate regressions revealed that grain Zn was not related to grain Cd, but was positively correlated with other nutrients such as Fe, Mn, P, Ca, K, and Mg, suggesting the possibility of producing durum grain with moderately high nutrients, while maintaining low concentration of Cd. Grain protein may represent a sink for micronutrients such as Zn and Fe. No interactive effect was detected among tested factors. Results of the study suggest that tillage management can have persistent effects on both grain yield and quality, that impacts of preceding crop do not persist past the first subsequent season and that P fertilizer can remain available for subsequent crops, reducing the response to annual inputs if P levels are maintained at adequate levels throughout the cropping sequence.

Acknowledgments

The authors gratefully acknowledge the financial support of Saskatchewan Flax Commission, International Plant Nutrition Institute, Westco Fertilizers, Ltd., Agrium, Ltd., United Grain Growers, and the Matching Investment Initiative of Agriculture and Agri-Food Canada for funding the study. The technical assistance of Brian Hadley, Mike Svistovski, Kim McDonald, and David Bancur is greatly appreciated.

References

[1] Jansson G., *Cadmium in arable crops: the influence of soil factors and liming*, Ph.D. thesis, Department of Soil Sciences, The Swedish University of Agricultural Sciences, Uppsala Sweden, 2002.

[2] C. A. Grant and L. D. Bailey, "Nitrogen, phosphorus and zinc management effects on grain yield and cadmium concentration in two cultivars of durum wheat," *Canadian Journal of Plant Science*, vol. 78, no. 1, pp. 63–70, 1998.

[3] "Codex general standard for contaminants and toxins in foods," CODEX STAN 193-1995, Rev. 5, 2009.

[4] Canada Statistics, "Crops and horticulture," 2010, http://www40.statcan.gc.ca/l01/ind01/l3_920_2024-eng.htm?hili_none.

[5] C. A. Grant, W. T. Buckley, L. D. Bailey, and F. Selles, "Cadmium accumulation in crops," *Canadian Journal of Plant Science*, vol. 78, no. 1, pp. 1–17, 1998.

[6] J. M. Clarke, D. Leisle, R. A. DePauw, and L. L. Thiessen, "Registration of five pairs of durum wheat genetic stocks near-isogenic for cadmium concentration," *Crop Science*, vol. 37, no. 1, p. 297, 1997.

[7] M. J. McLaughlin, D. R. Parker, and J. M. Clarke, "Metals and micronutrients: food safety issues," *Field Crops Research*, vol. 60, no. 1-2, pp. 143–163, 1999.

[8] I. Cakmak, W. H. Pfeiffer, and B. McClafferty, "Biofortification of durum wheat with zinc and iron," *Cereal Chemistry*, vol. 87, no. 1, pp. 10–20, 2010.

[9] E. J. Gawalko, R. G. Garrett, and T. W. Nowicki, "Cadmium, copper, iron, manganese, selenium, and zinc in Canadian spring wheat," *Communications in Soil Science and Plant Analysis*, vol. 33, no. 15-18, pp. 3121–3133, 2002.

[10] C. A. Grant and S. C. Sheppard, "Fertilizer impacts on cadmium availability in agricultural soils and crops," *Human and Ecological Risk Assessment*, vol. 14, no. 2, pp. 210–228, 2008.

[11] K. C. Jones and A. E. Johnston, "Cadmium in cereal grain and herbage from long-term experimental plots at Rothamsted, UK," *Environmental Pollution*, vol. 57, no. 3, pp. 199–216, 1989.

[12] M. A. Kashem and B. R. Singh, "The effect of fertilizer additions on the solubility and plant-availability of Cd, Ni and Zn in soil," *Nutrient Cycling in Agroecosystems*, vol. 62, no. 3, pp. 287–296, 2002.

[13] C. A. Grant and L. D. Bailey, "Effects of phosphorus and zinc fertiliser management on cadmium accumulation in flaxseed," *Journal of the Science of Food and Agriculture*, vol. 73, no. 3, pp. 307–314, 1997.

[14] J. O. Agbenin, "Phosphate-induced zinc retention in a tropical semi-arid soil," *European Journal of Soil Science*, vol. 49, no. 4, pp. 693–700, 1998.

[15] T. S. Verma and R. S. Minhas, "Zinc and phosphorus interaction in a wheat-maize cropping system," *Fertilizer Research*, vol. 13, no. 1, pp. 77–86, 1987.

[16] M. Haldar and L. N. Mandal, "Effect of phosphorus and zinc on the growth and phosphorus, zinc, copper, iron and manganese nutrition of rice," *Plant and Soil*, vol. 59, no. 3, pp. 415–425, 1981.

[17] S. R. Olsen, "Micronutrient interaction," in *Micronutrients in Agriculture*, J. J. Mortvedt, P. M. Giordano, and W. L. Lindsay, Eds., pp. 243–264, Soil Science Society of America, Wis, USA, 1972.

[18] X. Gao, D. N. Flaten, M. Tenuta, M. G. Grimmett, E. J. Gawalko, and C. A. Grant, "Soil solution dynamics and plant uptake of cadmium and zinc by durum wheat following phosphate fertilization," *Plant and Soil*, vol. 338, no. 1, pp. 423–434, 2011.

[19] M. Choudhary, L. D. Bailey, and C. A. Grant, "Effect of zinc on cadmium concentration in the tissue of durum wheat," *Canadian Journal of Plant Science*, vol. 74, no. 3, pp. 549–552, 1994.

[20] Y. Jiao, C. A. Grant, and L. D. Bailey, "Effects of phosphorus and zinc fertilizer on cadmium uptake and distribution in flax and durum wheat," *Journal of the Science of Food and Agriculture*, vol. 84, no. 8, pp. 777–785, 2004.

[21] P. R. Miller, Y. Gan, B. G. McConkey, and C. L. McDonald, "Pulse crops for the northern Great Plains: I. Grain productivity and residual effects on soil water and nitrogen," *Agronomy Journal*, vol. 95, no. 4, pp. 972–979, 2003.

[22] C. A. Campbell, R. P. Zentner, F. Selles et al., "Quantifying short-term effects of crop rotations on soil organic carbon in southwestern Saskatchewan," *Canadian Journal of Soil Science*, vol. 80, no. 1, pp. 193–202, 2000.

[23] C. Hamel, K. Hanson, F. Selles et al., "Seasonal and long-term resource-related variations in soil microbial communities in wheat-based rotations of the Canadian prairie," *Soil Biology and Biochemistry*, vol. 38, no. 8, pp. 2104–2116, 2006.

[24] D. P. Oliver, J. E. Schultz, K. G. Tiller, and R. H. Merry, "The effect of crop rotation and tillage practices on cadmium concentration in wheat grain," *Australian Journal of Agricultural Research*, vol. 44, no. 6, pp. 1221–1234, 1993.

[25] C. A. Grant, M. A. Monreal, R. B. Irvine, R. M. Mohr, D. L. McLaren, and M. Khakbazan, "Preceding crop and phosphorus fertilization affect cadmium and zinc concentration of flaxseed under conventional and reduced tillage," *Plant and Soil*, vol. 333, no. 1, pp. 337–350, 2010.

[26] R. P. Zentner, G. P. Lafond, D. A. Derksen, and C. A. Campbell, "Tillage method and crop diversification: effect on economic returns and riskiness of cropping systems in a thin black chernozem of the Canadian prairies," *Soil and Tillage Research*, vol. 67, no. 1, pp. 9–21, 2002.

[27] C. A. Grant and L. D. Bailey, "The effect of tillage and KCl addition on pH, conductance, NO_3-N, P, K and Cl distribution in the soil profile," *Canadian Journal of Soil Science*, vol. 74, no. 3, pp. 307–314, 1994.

[28] K. G. Tiller, D. P. Oliver, M. J. McLaughlin et al., "Managing cadmium contamination of agricultural land," in *Remediation of Soils Contaminated by Metals*, I. K. Iskandar, Ed., pp. 225–255, Science and Technology Letters, Northwood, UK, 1997.

[29] A. J. Franzluebbers and F. M. Hons, "Soil-profile distribution of primary and secondary plant-available nutrients under conventional and no tillage," *Soil and Tillage Research*, vol. 39, no. 3-4, pp. 229–239, 1996.

[30] L. M. Shuman and D. V. McCracken, "Tillage, lime, and poultry litter effects on soil zinc, manganese, and copper," *Communications in Soil Science and Plant Analysis*, vol. 30, no. 9-10, pp. 1267–1277, 1999.

[31] X. Gao, F. Akhter, M. Tenuta, D. N. Flaten, E. J. Gawalko, and C. A. Grant, "Mycorrhizal colonization and grain Cd concentration of field-grown durum wheat in response to tillage, preceding crop and phosphorus fertilization," *Journal of the Science of Food and Agriculture*, vol. 90, no. 5, pp. 750–758, 2010.

[32] C. A. Grant, M. A. Monreal, R. B. Irvine, R. M. Mohr, D. L. Mclaren, and M. Khakbazan, "Crop response to current and previous season applications of phosphorus as affected by crop sequence and tillage," *Canadian Journal of Plant Science*, vol. 89, no. 1, pp. 49–66, 2009.

[33] S. R. Olsen, C. V. Cole, F. S. Watanabe, and L. A. Dean, "Estimation of available phosphorus in soils by extraction with sodium bicarbonate," Circular number 939, USDA, 1954.

[34] W. L. Lindsay and W. A. Norvell, "Development of a DTPA soil test for zinc, iron, manganese, and copper," *Soil Science Society of America Journal*, vol. 42, no. 4, pp. 421–428, 1978.

[35] R. L. Westerman, *Soil Testing and Plant Analysis*, Soil Science Society of America, Madison, Wis, USA, 1990.

[36] X. Gao, K. R. Brown, G. J. Racz, and C. A. Grant, "Concentration of cadmium in durum wheat as affected by time, source and placement of nitrogen fertilization under reduced and conventional-tillage management," *Plant and Soil*, vol. 337, no. 1, pp. 341–354, 2010.

[37] R. D. Graham, R. M. Welch, D. A. Saunders et al., "Nutritious subsistence food systems," *Advances in Agronomy*, vol. 92, pp. 1–74, 2007.

[38] X. Gao and C. A. Grant, "Interactive effect of N fertilization and tillage management on Zn biofortification in durum wheat (*Triticum durum*)," *Canadian Journal of Plant Science*, vol. 91, no. 6, pp. 951–960, 2011.

[39] K. R. J. Smettem, A. D. Rovira, S. A. Wace, B. R. Wilson, and A. Simon, "Effect of tillage and crop rotation on the surface stability and chemical properties of a red-brown earth (Alfisol) under wheat," *Soil and Tillage Research*, vol. 22, no. 1-2, pp. 27–40, 1992.

[40] C. G. Castillo, R. Rubio, J. L. Rouanet, and F. Borie, "Early effects of tillage and crop rotation on arbuscular mycorrhizal fungal propagules in an Ultisol," *Biology and Fertility of Soils*, vol. 43, no. 1, pp. 83–92, 2006.

[41] X. Gao, M. Tenuta, D. N. Flaten, and C. A. Grant, "Cadmium concentration in flax colonized by mycorrhizal fungi depends on soil phosphorus and cadmium concentrations," *Communications in Soil Science and Plant Analysis*, vol. 42, no. 15, pp. 1882–1897, 2011.

[42] M. H. Ryan, J. K. McInerney, I. R. Record, and J. F. Angus, "Zinc bioavailability in wheat grain in relation to phosphorus fertiliser, crop sequence and mycorrhizal fungi," *Journal of the Science of Food and Agriculture*, vol. 88, no. 7, pp. 1208–1216, 2008.

[43] A. Andersson and G. Simon, "Levels of Cd and some other trace elements in soils and crops as influenced by lime and fertilizer level," *Acta Agriculturae Scandinavica*, vol. 4l, pp. 3–11, 1991.

[44] J. J. Mortvedt, "Cadmium levels in soils and plants from some long-term soil fertility experiments in the United States of America," *Journal of Environmental Quality*, vol. 16, no. 2, pp. 137–142, 1987.

[45] M. A. Monreal, C. A. Grant, R. B. Irvine, R. M. Mohr, D. L. McLaren, and M. Khakbazan, "Crop management effect on arbuscular mycorrhizae and root growth of flax," *Canadian Journal of Plant Science*, vol. 91, no. 2, pp. 315–324, 2011.

[46] X. Gao, R. M. Mohr, D. L. McLaren, and C. A. Grant, "Grain cadmium and zinc concentrations in wheat as affected by genotypic variation and potassium chloride fertilization," *Field Crops Research*, vol. 122, no. 2, pp. 95–103, 2011.

[47] A. K. Joshi, J. Crossa, B. Arun et al., "Genotype × environment interaction for zinc and iron concentration of wheat grain in eastern Gangetic plains of India," *Field Crops Research*, vol. 116, no. 3, pp. 268–277, 2010.

[48] F. J. Zhao, Y. H. Su, S. J. Dunham et al., "Variation in mineral micronutrient concentrations in grain of wheat lines of diverse origin," *Journal of Cereal Science*, vol. 49, no. 2, pp. 290–295, 2009.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 769357, 10 pages
doi:10.1155/2012/769357

Research Article

Oilseed Meal Effects on the Emergence and Survival of Crop and Weed Species

Katie L. Rothlisberger, Frank M. Hons, Terry J. Gentry, and Scott A. Senseman

Department of Soil and Crop Sciences, Texas A&M University, 370 Olsen Boulevard, 2474 TAMU, College Station, TX 77843-2474, USA

Correspondence should be addressed to Katie L. Rothlisberger, krothlisberger@ag.tamu.edu

Received 28 October 2011; Revised 20 December 2011; Accepted 23 December 2011

Academic Editor: Philip White

Oilseed crops are being widely evaluated for potential biodiesel production. Seed meal (SM) remaining after extracting oil may have use as bioherbicides or organic fertilizers. Brassicaceae SM often contains glucosinolates that hydrolyze into biologically active compounds that may inhibit various pests. *Jatropha curcas* SM contains curcin, a phytoxin. A 14-day greenhouse study determined that *Sinapis alba* (white mustard), *Brassica juncea* (Indian mustard), *Camelina sativa*, and *Jatropha curcas* applied to soil at varying application rates [0, 0.5, 1.0, and 2.5% (w/w)] and incubation times (1, 7, and 14 d) prior to planting affected seed emergence and seedling survival of cotton [*Gossypium hirsutum* (L.)], sorghum [*Sorghum bicolor* (L.) Moench], johnsongrass (*Sorghum halepense*), and redroot pigweed (*Amaranthus retroflexus*). With each species, emergence and survival was most decreased by 2.5% SM application applied at 1 and 7 d incubations. White mustard SM incubated for 1 d applied at low and high rates had similar negative effects on johnsongrass seedlings. Redroot pigweed seedling survival was generally most decreased by all 2.5% SM applications. Based on significant effects determined by ANOVA, results suggested that the type, rate, and timing of SM application should be considered before land-applying SMs in cropping systems.

1. Introduction

Research involving oilseed crops for biodiesel production has increased due to greater needs for renewable energy sources. Biodiesel is an EPA-approved renewable fuel that can be produced from oilseed crops. The oil extracted from seed is chemically reacted with an alcohol, such as methanol, to form chemical compounds known as fatty acid methyl esters, or "biodiesel." The oil contained in the seed is most often extracted mechanically using a screw press. The residue remaining after oil extraction is referred to as either a press cake or seed meal (SM). In order for biodiesel production to be economically and environmentally sustainable, a feasible and profitable means of byproduct or SM disposal and/or usage needs to be developed. Utilization of SM in organic agricultural production systems offers a possible solution.

Oilseeds have the potential to produce significant energy and renewable fuels and include such oilseeds as soybean [*Glycine max* (L.) Merr.], canola and rapeseed (*Brassica napus)*, Indian mustard (*Brassica juncea*), white mustard (*Sinapis alba*), physic nut or jatropha (*Jatropha curcas*), camelina (*Camelina sativa*), and castor bean (*Ricinus communis*). Brassicaceae oilseeds have been reported to contain 30 to 40% oil by weight [1], while jatropha seed contains a similar range of 30 to 37% [2]. Recent interest in jatropha is due primarily to its purported ability to grow on marginal lands. Therefore, its cultivation would be less likely to displace food-producing crops [3], but it is limited to subtropical and tropical environments. Jatropha and generally all oilseeds are rich in protein, containing a good balance of amino acids. The SM of jatropha reportedly contains more nutrients than either chicken or cattle manure [4].

Many oilseed meals, such as from soybean, have been used as additives in animal feed because of their high nutrient content, but certain plants within the Brassicaceae family cannot be used in the same manner because of their biocidal properties. Upon enzymatic hydration by myrosinase, a number of allelochemicals are produced in

Brassicaceae SMs as secondary biologically active compounds of glucosinolates, which are β-thioglycosides with a sulphonated oxime moiety and a variable side-chain derived from amino acids [5]. Myrosinase is physically separated from the glucosinolates until the plant tissue is disrupted [6]. Glucosinolates are grouped as either aliphatic, aromatic, or indolyl based on the nature of their side chain or R group. Seed meals with individual side chains in combination with environmental conditions such as pH, moisture levels, Fe^{2+} concentration, and the presence of coenzymes, determine which hydrolysis products will form. Allelochemical persistence and biocidal activity in soil will influence the ability of seed to germinate and survive. Potential allelochemicals include isothiocyanates (ITCs), ionic thiocyanates (SCN^-), nitriles, and oxazolidinediones (OZT).

Glucosinolate-containing SMs incorporated into soil have been reported to have possible herbicidal, insecticidal, nematicidal, and fungicidal effects [7]. A field study by Rice et al. [8] showed that white mustard, Indian mustard, and rapeseed SMs significantly reduced redroot pigweed (*Amaranthus retroflexus* L.) biomass by 59–93% compared to the control. A greenhouse study by Ju et al. [9] reported that SCN^-, liberated from white mustard SM, inhibited the growth of tobacco (*Nicotiana tabacum* L. cv. Delhi 76) and bean (*Phaseolus vulgaris* L. cv. Contender). Though not in the mustard family, jatropha SM also contains toxic compounds such as curcin, a toxalbumin, and other equally negative substances such as phorbol esters [3]. Phorbol esters are the likely source of toxicity in jatropha. These compounds decompose rapidly, usually within days, as they are sensitive to light, elevated temperatures, and atmospheric oxygen [10].

Oilseed meals may potentially be applied to agricultural soils as sources of organic nutrients and/or organic pesticides. However, concerns arise from the harmful effects that crop species may experience from SMs used in this manner. The main objective of this paper was to determine the potential effects of white mustard, Indian mustard, camelina, and jatropha SMs added to soil at varying application rates and incubation times on the emergence and early survival of both crop and weed species.

2. Materials and Methods

2.1. Soil and SM Collection and Characterization. Greenhouse studies were conducted using soil collected from the Texas AgriLife Research and Extension Center near Overton, TX. Soil at this site is characterized as Darco loamy fine sand (loamy, siliceous, semiactive, thermic Grossarenic Paleudults) with a pH of 5.6. The soil was air dried for approximately 21 days, thoroughly mixed and stored until further use. This soil was chosen due to its sandy texture and low native fertility.

Oilseed species chosen for this study were *Sinapis alba* cv. Ida Gold (L.A. Hearne Seeds, Monterey County, CA), *Brassica juncea* cv. Pacific Gold (L.A. Hearne Seeds, Monterey County, CA), *Jatropha curcas*, and *Camelina sativa* (Texas Agrilife Research and Extension, College Station, TX). Jatropha fruit was dehulled by hand prior to seed pressing. A motor-driven screw press operating at 95–100°C was used to extract the oil from seed. The oil constituted approximately 20–30% of the various seeds by weight, and approximately 90–95% of the total oil content was extracted. The SMs were stored at approximately 0°C until incorporation into soil. Both the soil and SMs were analyzed for total organic C and total N by a combustion procedure [11–13]. The soil was analyzed for extractable P, K, Ca, Mg, and S by Mehlich III [14, 15] and analysis by ICP, and micronutrients (Cu, Fe, Mn, and Zn) by extraction with DTPA-TEA, followed by ICP analysis [16], and extractable NO_3-N by cadmium reduction following extraction by 1 N KCl [17]. Mineral compositions of SM (B, Ca, Cu, Fe, K, Mg, Mn, Na, P, S, and Zn) were determined by ICP analysis of nitric acid digests. Soil electrical conductivity (EC) was determined in a 1 : 2 soil-to-water extract using deionized water, with the actual determination made using a conductivity probe [18]. Soil texture was determined using the hydrometer procedure [19].

Glucosinolate concentrations of white mustard and jatropha were determined by high performance liquid chromatography (HPLC) using methods of two previous studies [20, 21] based on ISO 9167 [22] and quantified glucosinolate concentrations of Indian mustard and camelina SMs, respectively. Expected retention behavior, such as time and sequence, and absorption spectra were used to identify individual glucosinolate peaks. Sinigrin monohydrate (Science Lab, Houston, TX) was utilized as an internal standard to calculate the major glucosinolate concentration.

2.2. Experimental Design and Data Collection. An emergence and survival study was conducted in a temperature-controlled glasshouse using cotton [*Gossypium hirsutum* (L.)], sorghum [*Sorghum bicolor* (L.) Moench], johnsongrass (*Sorghum halepense*), and redroot pigweed (*Amaranthus retroflexus*) as the crop and weed species. The study was a complete factorial within a completely randomized design with four replications of 36 treatment combinations, including: SM type (white mustard, Indian mustard, camelina, and jatropha), application rate [0.5, 1.0, and 2.5% on dry weight basis (w/w)], and incubation time (1, 7, and 14 d prior to planting). Before mixing with soil, SMs were finely crushed using a mortar and pestle. Approximately 340 g of soil-SM mixture were added to ~500-mL growth cups and incubated for the designated times at 32 to 35°C in the glass house. The soil was not disturbed other than at planting. The gravimetric water content of mixtures was kept constant at $0.24\,g\,g^{-1}$ by weighing and adding distilled water daily. Nonamended soil was used as the control treatment for each crop or weed species.

On 29 July 2009, ten sorghum or cotton, 50 redroot pigweed, or 100 johnsongrass viable seed were planted into each individual treatment replication. The actual number of seed planted was based on the average germination percentage of 100 crop/weed seed, which was determined prior to the start of the experiment (data not shown). Counting of emerged seedlings began the first day following planting and continued on a daily basis for 14 d. Seedlings were considered emerged when visible above the soil surface.

Table 1: Total nutrient concentrations of oilseed meals and total C and N and extractable nutrients in Darco soil.

Concentration	Soil	Oilseed meal			
	Darco	White mustard	Indian mustard	Camelina	Jatropha
pH	5.6	5.0	6.0	6.6	7.0
Organic C (%)	0.37	49.17	50.35	44.88	47.58
Total N (%)	0.08	5.09	5.00	5.36	3.46
C : N	4.6	9.7	10.1	8.4	13.8
NO_3-N (mg kg^{-1})	7.9	—	—	—	—
P (mg kg^{-1})	28	8848	11818	8695	8058
K (mg kg^{-1})	42	11014	11368	14978	15397
Ca (mg kg^{-1})	191	6341	6092	6832	11470
Mg (mg kg^{-1})	26	3473	4470	4270	4748
S (mg kg^{-1})	14	—	—	—	—
Na (mg kg^{-1})	97	493	588	550	1291
Fe (mg kg^{-1})	15.1	40.1	47.0	45.2	40.1
Zn (mg kg^{-1})	1.8	65.1	68.1	65.4	30.6
Mn (mg kg^{-1})	7.5	35.9	57.7	64.6	35.9
Cu (mg kg^{-1})	0.2	9.9	10.2	14.5	15.9

On the 14th and final day of data collection, survival counts were made based on the number of viable seedlings present within each replicate. Viable seedlings were defined as having a well-developed root and shoot system and as being at a comparable or more mature growth stage relative to the controls.

2.3. Statistical Analysis. Relative emergence was calculated as the percentage of planted seed emerged in SM treatments relative to those emerged in controls. Relative survival was based on the number of viable seedlings in treatments as a percent of control seedlings. Statistical analysis was conducted using SAS version 9.2. The effects of main factors and their interactions on crop and weed emergence and survival were analyzed using a mixed analysis of variance (ANOVA) procedure at a significance level of $P < 0.05$. Main and interaction means when significant were separated using Fisher's protected LSD.

3. Results

3.1. Soil and SM Characteristics. Results showed the Darco soil to be deficient in plant available N, P, K, and Mg. The soil was sufficient in Ca, S, and Cu, and somewhat high to moderate in Fe, Zn, and Mn (Table 1). This sandy soil (79.3% sand, 14.2% clay, and 6.5% silt) exhibited an EC value of 37 μmhos cm^{-1}; therefore, its salinity effects are negligible. Compositional analysis of SMs indicated that these materials may potentially supply significant amounts of nutrients for plant growth (Table 1). White mustard, Indian mustard, and camelina SMs had similar concentrations of total C and N (45 to 50% and 5%, resp.). Total N was less in jatropha SM. Carbon : N ratios ranged from 8.4 to 10.1% for glucosinolate-containing SMs and was 13.8% for jatropha SM. Phosphorus concentration of Indian mustard SM was higher at 1.2% compared to the other three meals that averaged 0.9% P. Potassium concentration of jatropha SM was 1.5%, which was greater than the average of the three remaining SMs at 1.3%. Nutrient concentrations of SMs were comparable to values previously reported for Brassicaceae SMs to average 50% C, 5.9% N, and 1.3% P by weight [1].

Glucosinolate extracts from SMs were utilized as an indicator of the potential biocidal activity that may be produced when *Brassicaceae* SMs are incorporated into soil. Other than jatropha, each SM in this study was determined to have its own individual glucosinolate profile. As mentioned previously, jatropha does not contain glucosinolates. The dominant glucosinolate compound found in white mustard SM was 4-hydroxybenzyl glucosinolate (glucosinalbin or sinalbin) at a concentration of 149.6 μmol g^{-1} on dry weight basis and a standard deviation of 2.3 μmol g^{-1}. Indian mustard SM contained several compounds with the dominant one being 2-propenyl glucosinolate (sinigrin) at a concentration of 159.1 $\pm$ 15.9 μmol g^{-1}. These results correspond to those of Hansson et al. [7] and Rice et al. [8] who found the dominant compound contained in Indian mustard SM to be sinigrin at a concentration of 123.8 $\pm$ 15.3 μmol g^{-1} and 152.0 $\pm$ 12.3 μmol g^{-1}, respectively. Camelina SM contained three dominant compounds with the most prominent being 10-methylsufinyldecyl (12.2 $\pm$ 7.5 μmol g^{-1}) [21].

3.2. Effects on Johnsongrass. Within each main factor (SM source, application rate, and incubation time), observed effects were significant for both relative emergence and survival of johnsongrass (Table 2). Rate exhibited the most significant effect on emergence, while all three main effects were highly significant ($P < 0.001$) for survival. Camelina and white mustard SM resulted in significantly lower emergence (78.8 and 79.0%, resp.) for johnsongrass compared with jatropha SM (91.0%) (Figure 1). Johnsongrass in the 0.5% jatropha SM treatment had a relative emergence greater

Table 2: ANOVA results for the main and interactive effects of seed meal source, application rate, and incubation time on cotton, sorghum, Johnsongrass, and pigweed emergence (emerg), and survival (surv). SM denotes seed meal source.

Effect	Cotton		Sorghum		Johnsongrass		Pigweed	
	emerg	surv	emerg	surv	emerg	surv	emerg	surv
				P value				
SM	<.0001	0.0349	0.6148	<.0001	0.0283	<.0001	0.2307	0.0024
Rate	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001
SM∗Rate	0.0541	0.2411	0.8481	0.0031	0.0315	0.0374	0.0899	0.0018
Incubation	0.1191	<.0001	0.0266	0.007	0.0185	<.0001	<.0001	<.0001
SM∗Incubation	<.0001	0.0182	0.0009	0.1825	0.2107	<.0001	0.0017	0.0095
Rate∗Incubation	0.0041	0.0001	0.0059	0.3865	0.0056	0.0285	0.0002	0.0715
SM∗Rate∗Incubation	0.3804	0.0433	0.0084	0.0428	<.0001	0.0029	0.0978	0.0008

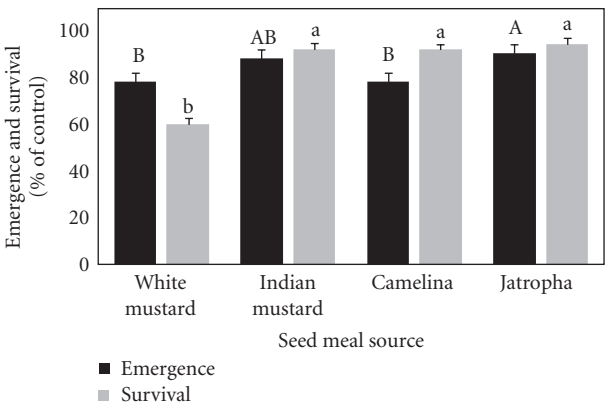


Figure 1: Main effect of "seed meal source" on Johnsongrass emergence and survival. Means within emergence or survival followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Uppercase letters separate emergence means and lowercase letters separate survival means. Data are means (four replications) ±SE.

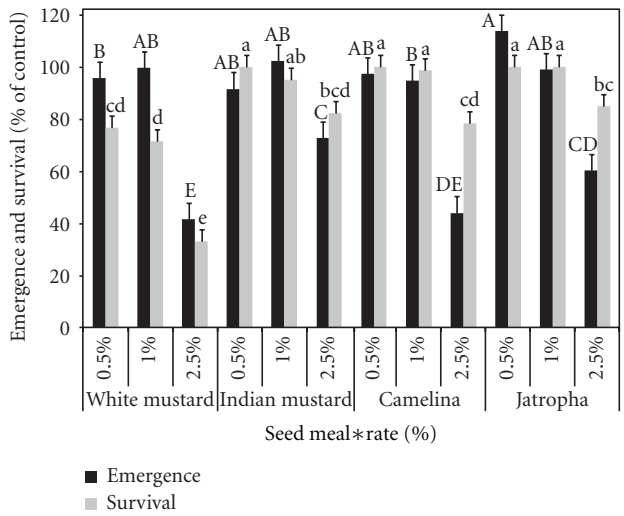


Figure 2: Interactive effects of "seed meal type and rate" on Johnsongrass emergence and survival. Means followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Uppercase letters separate emergence means and lowercase letters separate survival means. Data are means (four replications) ±SE.

than 100% (114%) because emergence in this treatment was greater than that of the control (Figure 2). This indicates that jatropha SM added at a rate of 0.5% does not cause injury, but does provide available nutrients for plant growth that the control does not.

Johnsongrass, redroot pigweed, cotton, and sorghum all showed significantly less emergence and survival with an SM application rate of 2.5% (Figure 3). Relative survival of johnsongrass seedlings in white mustard treatments was also significantly less (60.4%) than with any of the other three SMs (92.3–94.9%) (Figure 1). Incubation time exhibited significantly different effects on relative emergence and survival of johnsongrass (Table 2). The 7 d incubation resulted in significantly less relative emergence than when incubated for 14 d (78.0 and 90.8%, resp.), but not 1 d (84.5%). However, the 1 d incubation did result in significantly less relative survival than either 7 or 14 d (67.0, 91.9, and 96.2%, resp.) (data not shown).

Johnsongrass was more resistant than the two crops to phytotoxins in SMs, especially at higher application rates (Figure 3). The treatment combination that was most effective at suppressing johnsongrass emergence was 2.5% white mustard SM incubated for 7 d (16.4%) and was significantly less than for any other treatment combination (Table 3). Seedling survival was most affected by 2.5% white mustard SM applied only 1 d prior to planting (4.4%) (Table 3). The relative survival of johnsongrass seedlings with the latter treatment was significantly less than for any other treatment combination, other than 1.0% white mustard incubated 1 day (14.6%). With a short period, such as 1 d, between SM incorporation and seeding, there was sufficient time for SCN^- production to reach toxic quantities from 1.0% white mustard SM to suppress johnsongrass growth. Therefore, if applied at correct timings, 1.0% white mustard SM is as effective at suppressing johnsongrass as 2.5% white mustard SM.

3.3. Effects on Redroot Pigweed. Seed meal type did not affect relative emergence of redroot pigweed, but did significantly influence relative survival (Table 2). Camelina and

Table 3: Three-way interaction of "seed meal source, application rate (applic) and incubation time (incub)" on johnsongrass and pigweed emergence (emerg) and survival (surv). Incubation refers to the length of time in days after SM was added to soil and prior to seeding. Data are the means (four replications) within weed species ($n = 144$).

Seed meal	Applic	Incub	Johnsongrass		Pigweed	
			Emerg	Surv	Emerg	Surv
	%	d	% of control			
White mustard	0.5	1	83.2	30.1	103.5	97.4
	0.5	7	85.5	100.0	48.4	96.4
	0.5	14	118.5	100.0	109.8	100.0
	1.0	1	83.2	14.6	29.8	81.3
	1.0	7	117.3	100.0	6.3	18.8
	1.0	14	98.3	100.0	25.5	75.0
	2.5	1	74.0	4.4	7.0	18.8
	2.5	7	16.4	28.1	0.0	0.0
	2.5	14	34.5	66.5	0.0	0.0
Indian mustard	0.5	1	87.8	100.0	50.9	92.9
	0.5	7	106.4	100.0	54.7	75.0
	0.5	14	80.7	100.0	139.2	100.0
	1.0	1	100.0	85.3	24.6	87.5
	1.0	7	95.5	100.0	4.7	75.0
	1.0	14	111.8	100.0	115.7	100.0
	2.5	1	100.0	47.0	0.0	0.0
	2.5	7	20.9	100.0	0.0	0.0
	2.5	14	97.5	100.0	49.0	100.0
Jatropha	0.5	1	95.4	100.0	45.6	90.2
	0.5	7	127.3	100.0	101.6	100.0
	0.5	14	118.5	100.0	133.3	100.0
	1.0	1	109.9	100.0	17.5	75.0
	1.0	7	83.6	100.0	29.7	75.0
	1.0	14	103.4	100.0	156.9	100.0
	2.5	1	52.7	59.2	0.0	0.0
	2.5	7	74.5	100.0	0.0	0.0
	2.5	14	53.8	95.3	0.0	0.0
Camelina	0.5	1	91.6	100.0	108.8	100.0
	0.5	7	83.6	100.0	40.6	90.8
	0.5	14	116.8	100.0	172.5	100.0
	1.0	1	96.9	95.8	15.8	29.2
	1.0	7	90.0	100.0	4.7	25.0
	1.0	14	97.5	100.0	98.0	95.0
	2.5	1	38.9	67.9	0.0	0.0
	2.5	7	34.5	75.0	0.0	0.0
	2.5	14	58.8	92.3	0.0	0.0
$LSD_{0.05}$			30.7	23.0	NS	33.7

white mustard SMs significantly reduced redroot pigweed survival compared with Indian mustard (48.9, 54.2, and 70.1%, resp.) (Figure 4). Redroot pigweed seed and seedlings were extremely sensitive to SM treatments applied at 2.5% (Table 2, Figure 3). Incubation times of 1 and 7 d produced significantly lower relative emergence and survival percentages relative to 14 d (33.6, 24.2, and 83.3% emergence, respectively, and 56.0, 46.3, and 72.5% survival, resp.) (Figure 5). Relative emergence and survival were 0% for all 2.5% treatments, with the exception of Indian mustard incubated for 14 d (49.0% emergence and 100% survival) and white mustard incubated for 1 d (7.0% emergence and 18.8% survival) (Table 3). Numerically, relative survival of seedlings in treatments of 2.5% white mustard applied 1 d

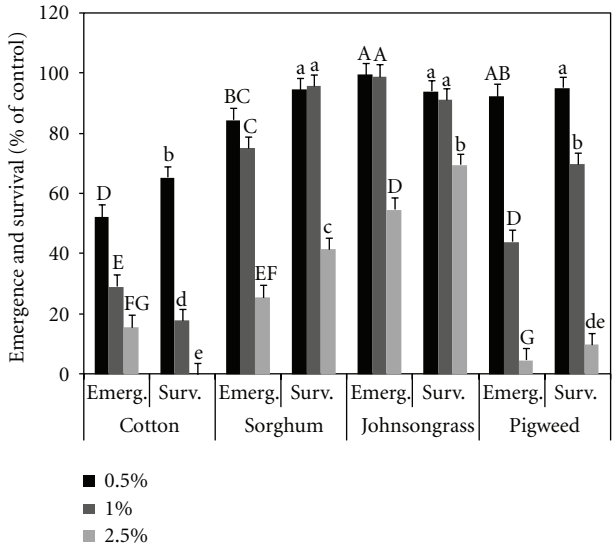


Figure 3: Seed meal rate effect on cotton, sorghum, Johnsongrass, and pigweed emergence and survival. Means followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Uppercase letters separate emergence means and lowercase letters separate survival means. Data are means (four replications) ±SE.

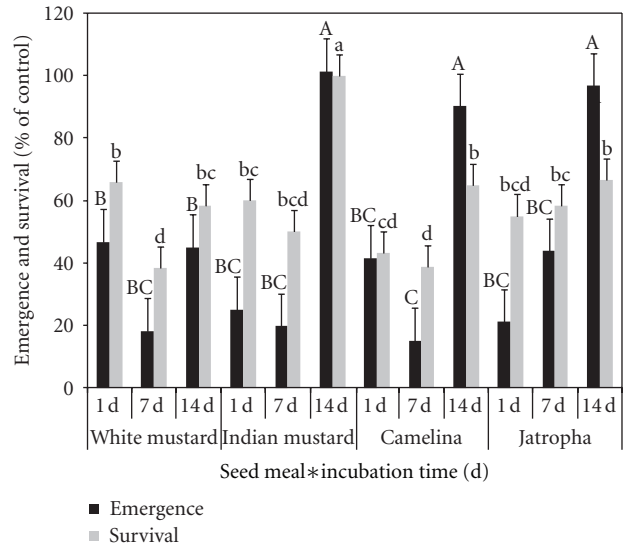


Figure 5: Interactive effects of "seed meal source and incubation time" on pigweed emergence (uppercase letters) and survival (lowercase letters). Means within emergence or survival followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Data are means (four replications) ±SE.

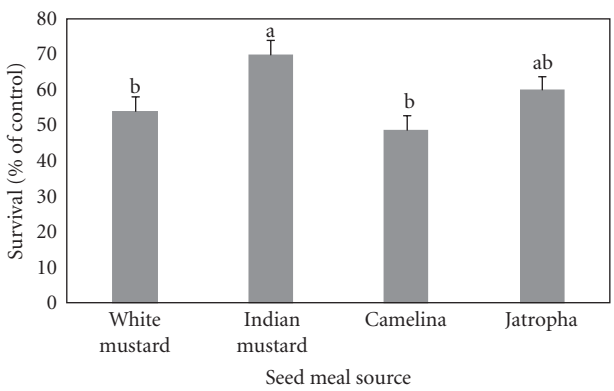


Figure 4: Main effect of "seed meal source" on pigweed survival. Means followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. The effect of "seed meal source" was not significant for pigweed emergence; therefore, data is not shown. Data are means (four replications) ±SE.

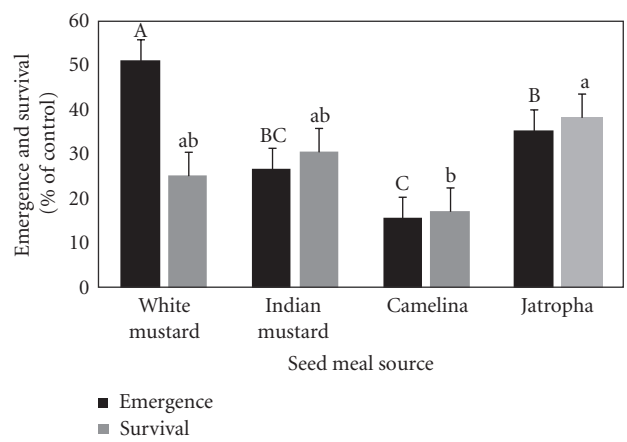


Figure 6: Main effect of "seed meal source" on cotton emergence (uppercase letters) and survival (lowercase letters). Means within emergence or survival followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Data are means (four replications) ±SE.

before planting was higher than all other 2.5% treatments, but statistically there were no significant differences for any of the test plants (Tables 2 and 3).

3.4. Effects on Cotton. Of the three main effects, incubation time was the only one that did not show significant treatment effects on emergence of cotton seed (Table 2). Camelina SM resulted in significantly lower emergence (15.7%) than white mustard (51.4%) and jatropha (35.5%), but not Indian mustard (26.9%) (Figure 6). Seedling survival showed somewhat different results, with camelina treatments showing numerically the lowest survival (17.1%), but being only significantly less compared to treatments with jatropha (38.3%), which resulted in the highest survival percentage (Figure 6).

As with both weed species, treatment combinations including 2.5% SM exhibited significantly reduced seedling emergence and survival (Table 2, Figure 3). Incubation time significantly altered seedling survival, but not emergence (Table 2). One day incubation prior to planting had the most negative impact on seedling survival, but not emergence (Figure 7). The longer incubation time of 14 d increased average seedling survival to 46.4%, but relative emergence was still only 31.9% for this incubation treatment. This result

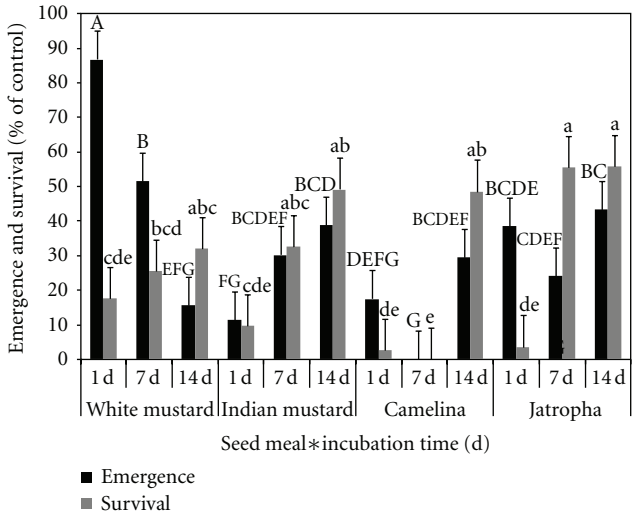


Figure 7: Interactive effect of "seed meal source and incubation time" on cotton emergence and survival. Emergence (uppercase letters) or survival (lowercase letters) means followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Data are means (four replications) ±SE.

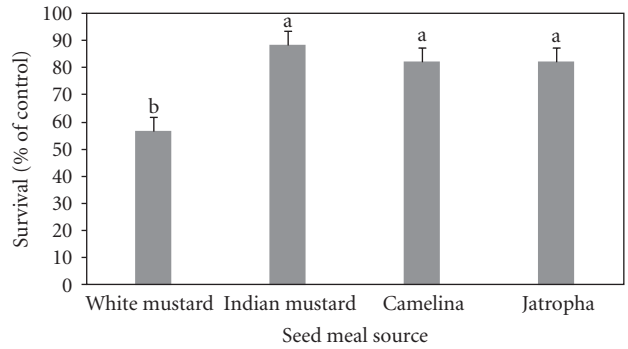


Figure 8: Main effect of "seed meal source" on sorghum survival. Means followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. The effect of "seed meal source" was not significant for sorghum emergence; therefore, the data is not shown. Data are means (four replications) ±SE.

likely indicates the necessity for SM incubation longer than 14 d prior to planting cotton.

The two-way interaction of "seed meal source and application rate" was not significant for either relative emergence or survival of cotton (Table 2). From the two-way interaction of "seed meal source and incubation time" (Table 2, Figure 7), which was significant for both emergence and survival, rates of glucosinolate hydrolysis might be inferred. Hydrolysis of glucosinolates in white mustard SM based on emergence apparently increased over the incubation period, decreased with Indian mustard, and showed greatest toxicity at 7 d for camelina. White mustard SM applied 1 d prior to planting resulted in the highest emergence rate (86.8%) relative to other treatments, but the survival rate of the seedlings was poor (17.7%) (Figure 7). Longer incubation periods of white mustard SM resulted in decreased emergence, but increased seedling survival. The most negative effects on cotton emergence and survival with Indian mustard SM were observed with 1 d incubation (11.4% emergence and 9.8% survival), while camelina and jatropha SMs were most detrimental at 7 d incubation (Figure 7).

The three-way interaction of "seed meal source, application rate, and incubation time" was not significant for cotton emergence, and only slightly for survival (Table 2). White mustard applied at 2.5% and incubated for 1 d resulted in significantly higher cotton emergence (94.7%) compared to any other treatment containing of 2.5% SM (0 to 36.8%) (Table 4). Relative survival of seedlings in this treatment, however, failed to be significantly different than white mustard added at 2.5% and incubated for 7 or 14 d. Seed of certain species, especially cotton and sorghum, sometimes emerged, but did not survive. The treatment most effective at suppressing johnsongrass and redroot pigweed growth, 2.5% white mustard SM at 1 or 7 d incubation (Table 3), also resulted in 0% survival of cotton seedlings (Table 4).

3.5. Effects on Sorghum. Of the three main effects, SM source was the only one not significant for sorghum emergence, but all three were significant for seedling survival (Table 2). Sorghum seedling survival was significantly less when treated with white mustard SM (56.6%) relative to all other SMs (82.1% to 88.3%) (Figure 8). Application of 2.5% SM resulted in both significantly reduced emergence and seedling survival (25.6 and 41.5%, resp.) compared to other rates (75.1 to 84.6% emergence and 94.8 to 95.8% survival) (Figure 3).

The three-way interaction was significant for both relative emergence and survival (Table 2). As with cotton, emergence of sorghum planted in treatments with white mustard SM decreased with increasing incubation time, while survival increased from 1 to 7 d of incubation (Figure 9, Table 4). White mustard SM at 2.5% and 1 d incubation had significantly greater relative emergence (75.9%) than any other 2.5% SM treatment combination (2.9 to 45.7%) (Table 4). No treatment combinations were able to completely inhibit emergence, but all treatments containing 2.5% white mustard SM resulted in 0% relative survival of sorghum (Table 4).

4. Discussion

The use of oilseed meals as soil amendments has several potential benefits, but there are also possible detriments. Primarily, SMs might serve to replenish soil organic matter (SOM) in cropping systems where, for instance, stover has been removed for use as biofuel feedstocks. Used in this manner, meals from certain oilseeds have the potential to add significant organic C and nutrients to soil, while controlling or inhibiting weed growth. Our results suggest that in order to suppress weeds, white mustard SM should be applied at rates between 1 and 2.5%, which will also supply a substantial amount of N (1120 to 2800 kg N ha^{-1}). Wang et al. [20] reported 3035 kg N ha^{-1} and 4263 kg N ha^{-1}

Table 4: Three-way interaction of "seed meal source, application rate (applic), and incubation time (incub)" on cotton and sorghum emergence (emerg) and survival (surv). Incubation refers to the length of time in days after SM was added to soil and prior to seeding. Data are the means (four replications) within crop species ($n = 144$).

Seed meal	Applic	Incub	Cotton		Sorghum	
			Emerg	Surv	Emerg	Surv
	%	d	% of control			
White mustard	0.5	1	76.3	48.9	106.9	46.3
	0.5	7	72.7	70.5	88.5	100.0
	0.5	14	33.3	67.0	60.0	96.4
	1.0	1	89.5	4.1	62.1	66.3
	1.0	7	59.1	6.4	73.1	100.0
	1.0	14	13.9	29.2	65.7	100.0
	2.5	1	94.7	0.0	75.9	0.0
	2.5	7	22.7	0.0	3.8	0.0
	2.5	14	0.0	0.0	2.9	0.0
Indian mustard	0.5	1	26.3	29.3	69.0	100.0
	0.5	7	90.9	97.8	100.0	100.0
	0.5	14	75.0	108.8	94.3	94.4
	1.0	1	5.3	0.0	72.4	100.0
	1.0	7	0.0	0.0	103.8	100.0
	1.0	14	25.0	38.9	68.6	100.0
	2.5	1	2.6	0.0	13.8	50.0
	2.5	7	0.0	0.0	26.9	50.0
	2.5	14	16.7	0.0	45.7	100.0
Jatropha	0.5	1	34.2	10.9	65.5	100.0
	0.5	7	45.5	128.2	103.8	100.0
	0.5	14	75.0	99.8	71.4	100.0
	1.0	1	44.7	0.0	75.9	100.0
	1.0	7	27.3	38.5	76.9	100.0
	1.0	14	55.6	67.6	80.0	100.0
	2.5	1	36.8	0.0	34.5	20.8
	2.5	7	0.0	0.0	38.5	68.8
	2.5	14	0.0	0.0	5.7	50.0
Camelina	0.5	1	28.9	8.2	93.1	100.0
	0.5	7	0.0	0.0	76.9	100.0
	0.5	14	69.4	116.6	85.7	100.0
	1.0	1	10.5	0.0	62.1	83.3
	1.0	7	0.0	0.0	103.8	100.0
	1.0	14	19.4	29.2	57.1	100.0
	2.5	1	13.2	0.0	37.9	58.3
	2.5	7	0.0	0.0	3.8	25.0
	2.5	14	0.0	0.0	17.1	75.0
$LSD_{0.05}$			NS	43.6	32.1	34.2

present in soil after 51 d of incubation with mustard SM (6.1% N) applied at a rate of 1.0 and 2.5%, respectively. Nitrogen applied in excess to soils and not synchronous with plant uptake may be lost from the system and could pose significant environmental risks. Seed meals applied at appropriate rates contain nutrient concentrations capable of potentially enhancing the productivity of low nutrient soils.

The absence of differences in C : N ratios of glucosinolate containing SMs and the low buffering capacity of Darco soil suggests that there should be no confounding allelopathic effects on emergence and/or survival. As mentioned above, white mustard SM applied to soil at 2.5% and incubated for 1 or 7 d prior to planting was most inhibitory to johnsongrass, which was the more difficult of the two weeds to control.

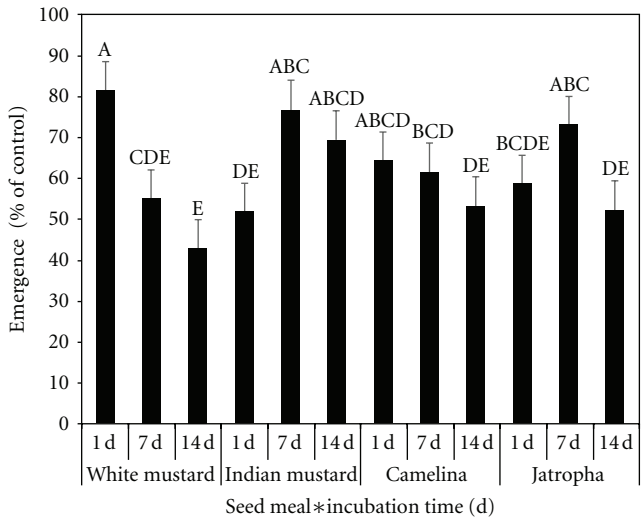


Figure 9: Interactive effect of "seed meal source and incubation time" on sorghum emergence. Emergence means followed by the same letter are not different at $P < 0.05$ by Fisher's protected LSD. Interaction effects on sorghum survival were not significant ($P = 0.1825$); therefore, the data is not shown. Data are means (four replications) ±SE.

While relative emergence of johnsongrass was significantly higher in treatments of "1% white mustard incubated for 1 d" compared to the most inhibitory treatment, relative survival of seedlings in this treatment failed to be significantly different than with the 2.5% SM application. It is likely that an application rate ranging from 1 to 2.5% SM would be adequate to suppress johnsongrass growth.

Redroot pigweed emergence and survival was suppressed by all SM treatments of 2.5%, excluding Indian mustard SM incubated for 14 d. It is hypothesized that after 14 d of incubation the toxicity associated with Indian mustard SM dissipated sufficiently so that its inhibitory effects were reduced compared to other SMs. These results are in contrast to results reported by Rice et al. [8], who found that Indian mustard SM applied at 3% was the only SM of the three studied (white mustard, Indian mustard, and rapeseed) to suppress redroot pigweed biomass compared to the no-meal treatment.

The treatment combination of "2.5% white mustard SM with 7 or 14 d incubation" prior to planting was extremely detrimental to cotton and sorghum in our study, indicating that this SM likely must be incubated for a longer period of time before planting agricultural crops. Previous studies have shown the phytotoxin associated with white mustard, SCN^-, decreased to almost background concentrations after 44 d at an application rate of 2 t ha^{-1} [7]. Phytotoxin dissipation in soil is highly dependent on SM application rates, soil water concentration, microbial activity, glucosinolate release efficiency, and rate of reaction.

Due to the decrease in cotton seed emergence from 1 to 14 d of incubation when planted in white mustard SM treatments, the rate of white mustard glucosinolate hydrolysis was assumed to be slower relative to the other SMs. Glucosinolates in Indian mustard SM may have had the fastest rate of reaction since cotton seed emergence was lowest for treatments with 1 day incubation. Isothiocyanate concentrations of Indian mustard and rapeseed tissues have been shown to be highest 24 hrs after incorporation and then dropping to less than half of the maximum in 72 hrs [23]. Other studies have reported SCN^- to have a longer half-life in soil compared with 2-propenyl isothiocyanate, the major phytotoxin produced from Indian mustard [24, 25]. Research has further shown that 60% of SCN^- remained after 6 days [25], whereas the average half-life of 2-propenyl ITC in six different soils was 48 h [24]. The rate of glucosinolate hydrolysis and ITC persistence are dependent on many soil and environmental factors and for this reason are somewhat unpredictable, but they appear to be a feasible means of determining the point at which phytotoxins are at maximum concentrations and consequently, most detrimental to plant viability.

5. Conclusion

Mechanical weed control is a commonly used practice in organic farming systems but is not always feasible, successful, or economical. This study demonstrated the ability of oilseed meals to suppress and, in some cases, control johnsongrass and redroot pigweed by as much as 96%. While weed suppression is achievable, factors such as soil characteristics, SM source, application rate, and incubation time prior to planting agronomic crops must be optimized to control weeds without damaging crops. The more nominal and practical SM application rate of 0.5% was much less effective in suppressing weeds compared to higher rates, especially 2.5%. Rates of SM needed to effectively control weeds, however, may also supply very large quantities of nutrients, particularly N, that could have negative environmental consequences. Further research, including but not limited to plant injury, crop yield, mammalian toxicology isothiocyanate, isothiocyanate biological activity, and soil persistence, is needed before SMs can be routinely recommended for organic production systems.

References

[1] A. Snyder, M. J. Morra, J. Johnson-Maynard, and D. C. Thill, "Seed meals from brassicaceae oilseed crops as soil amendments: influence on carrot growth, microbial biomass nitrogen, and nitrogen mineralization," *HortScience*, vol. 44, no. 2, pp. 354–361, 2009.

[2] G. R. Rao, G. R. Korwar, A. K. Shanker, and Y. S. Ramakrishna, "Genetic associations, variability and diversity in seed characters, growth, reproductive phenology and yield in Jatropha curcas (L.) accessions," *Trees*, vol. 22, no. 5, pp. 697–709, 2008.

[3] A. J. King, W. He, J. A. Cuevas, M. Freudenberger, D. Ramiaramanana, and I. A. Graham, "Potential of Jatropha curcas as a source of renewable oil and animal feed," *Journal of Experimental Botany*, vol. 60, no. 10, pp. 2897–2905, 2009.

[4] G. Francis, R. Edinger, and K. Becker, "A concept for simultaneous wasteland reclamation, fuel production, and socio-economic development in degraded areas in India: need,

potential and perspectives of Jatropha plantations," *Natural Resources Forum*, vol. 29, no. 1, pp. 12–24, 2005.

[5] R. F. Mithen, "Glucosinolates and their degradation products," *Advances in Botanical Research*, vol. 35, pp. 213–232, 2001.

[6] A. L. Gimsing and J. A. Kirkegaard, "Glucosinolates and biofumigation: fate of glucosinolates and their hydrolysis products in soil," *Phytochemistry Reviews*, vol. 8, no. 1, pp. 299–310, 2009.

[7] D. Hansson, M. J. Morra, V. Borek, A. J. Snyder, J. L. Johnson-Maynard, and D. C. Thill, "Ionic thiocyanate (SCN-) production, fate, and phytotoxicity in soil amended with Brassicaceae seed meals," *Journal of Agricultural and Food Chemistry*, vol. 56, no. 11, pp. 3912–3917, 2008.

[8] A. R. Rice, J. L. Johnson-Maynard, D. C. Thill, and M. J. Morra, "Vegetable crop emergence and weed control following amendment with different," *Renewable Agriculture and Food Systems*, vol. 22, no. 3, pp. 204–212, 2007.

[9] H. Y. Ju, B. B. Bible, and C. Chong, "Influence of ionic thiocyanate on growth of cabbage, bean, and tobacco," *Journal of Chemical Ecology*, vol. 9, no. 8, pp. 1255–1262, 1983.

[10] R. E. E. Jongschaap, W. J. Corré, P. S. Bindraban, and W. A. Brandenburg, *Claims and Facts on Jatropha Curcas L.*, Plant Research International, Wageningen, The Netherlands, 2007.

[11] S. L. McGeehan and D. V. Naylor, "Automated instrumental analysis of carbon and nitrogen in plant and soil samples," *Communications in Soil Science & Plant Analysis*, vol. 19, no. 4, pp. 493–505, 1988.

[12] E. E. Schulte and B. G. Hopkins, "Estimation of soil organic matter by weight lost-on-ignition," in *Soil Organic Matter: Analysis and Interpretation*, F. R. Magdoff, M. A. Tabatabai, and E. A. Hanlon Jr., Eds., Special Publication No. 46, pp. 21–32, Soil Science Society of America, Madison, Wis, USA, 1996.

[13] D. A. Storer, "A simple high sample volume ashing procedure for determination of soil organic matter," *Communications in Soil Science & Plant Analysis*, vol. 15, no. 7, pp. 759–772, 1984.

[14] A. Mehlich, "New extractant for soil test evaluation of phosphorus, potassium, magnesium, calcium, sodium, manganese, and zinc," *Communications in Soil Science and Plant Analysis*, vol. 9, pp. 477–492, 1978.

[15] A. Mehlich, "Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant," *Communications in Soil Science & Plant Analysis*, vol. 15, no. 12, pp. 1409–1416, 1984.

[16] W. L. Lindsay and W. A. Norvell, "Development of a DTPA soil test for zinc, iron, manganese, and copper," *Soil Science Society of America Journal*, vol. 42, pp. 421–428, 1978.

[17] D. R. Keeney and D. W. Nelson, "Nitrogen—inorganic forms," in *Methods of Soil Analysis, Part 2*, A. L. Page et al., Ed., pp. 643–687, ASA and SSSA, Madison, Wis, USA, 1982.

[18] J. D. Rhoades, "Soluble salts," in *Methods of Soil Analysis, Part 2*, A. L. Page et al., Ed., pp. 167–168, ASA and SSSA, Madison, Wis, USA, 1982.

[19] P. R. Day, "Particle fractionation and particle-size analysis," in *Methods of Soil Analysis, Part 1*, C. A. Black et al., Ed., pp. 545–567, ASA and SSSA, Madison, Wis, USA, 1965.

[20] A. S. Wang, P. Hu, E. B. Hollister et al., "Impact of Indian mustard (*Brassica juncea*) and flax (*Linum usitatissimum*) seed meal applications on soil carbon, nitrogen, and microbial dynamics," *Applied and Environmental Soil Science*, vol. 2012, Article ID 351609, 14 pages, 2012.

[21] P. Hu, A. S. Wang, A. S. Engledow et al., "Inhibition of the germination and growth of *Phymatotrichopsis omnivora* (Cotton root rot) by oilseed meals and isothiocyanates," *Applied Soil Ecology*, vol. 49, pp. 68–75, 2011.

[22] International Organization for Standarization, *Rapeseed–Determination of Glucosinolates Content–part 1: Method Using High-Performance Liquid Chromatography*, ISO 9167-1:1992-(E), Geneva, Switzerland, 1992.

[23] M. J. Morra and J. A. Kirkegaard, "Isothiocyanate release from soil-incorporated Brassica tissues," *Soil Biology and Biochemistry*, vol. 34, no. 11, pp. 1683–1690, 2002.

[24] V. Borek, M. J. Morra, P. D. Brown, and J. P. McCaffrey, "Transformation of the glucosinolate-derived allelochemicals allyl isothiocyanate and allylnitrile in soil," *Journal of Agricultural and Food Chemistry*, vol. 43, no. 7, pp. 1935–1940, 1995.

[25] P. D. Brown and M. J. Morra, "Fate of ionic thiocyanate (SCN-) in soil," *Journal of Agricultural and Food Chemistry*, vol. 41, no. 6, pp. 978–982, 1993.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 940954, 8 pages
doi:10.1155/2012/940954

Research Article

Promoting Cassava as an Industrial Crop in Ghana: Effects on Soil Fertility and Farming System Sustainability

S. Adjei-Nsiah[1] and Owuraku Sakyi-Dawson[2]

[1] *Forest and Horticultural Crops Research Centre, Kade, Institute of Agricultural Research, College of Agriculture and Consumer Sciences, University of Ghana, P.O. Box 68, Legon, Ghana*
[2] *Department of Agricultural Extension, School of Agriculture, College of Agriculture and Consumer Sciences, University of Ghana, P.O. Box 68, Legon, Ghana*

Correspondence should be addressed to S. Adjei-Nsiah, y_nsiah@yahoo.co.uk

Received 21 November 2011; Accepted 2 January 2012

Academic Editor: María Cruz Díaz Álvarez

Cassava is an important starchy staple crop in Ghana with per capita consumption of 152.9 kg/year. Besides being a staple food crop, cassava can be used as raw material for the production of industrial starch and ethanol. The potential of cassava as an industrial commercial crop has not been exploited to a large extent because of perceptions that cassava depletes soils. Recent finding from field studies in the forest/savannah transitional agroecological zone of Ghana indicates that when integrated in the cropping system as a form of rotation, cassava contributes significantly to maintenance of soil fertility, and thus large scale production of cassava for industrial use can contribute to poverty reduction in an environmentally responsive way. This paper discusses the role of cassava cultivation in soil fertility management and its implication for farming system sustainability and industrialization.

1. Introduction

Cassava is an important starchy staple crop in Ghana with per capita consumption of 152.9 kg/year [1]. Besides being a staple food crop, cassava can be used as raw material for the production of industrial starch and ethanol. In Ghana, cassava is cultivated as a monocrop or intercropped with other food crops, either as the dominant or subsidiary crop. In terms of quantity produced, cassava is the most important root crop in Ghana followed by yams and cocoyams, but cassava ranks second to maize in terms of area planted. The production of cassava in Ghana ranged from 10,217,929 MT to 12,260,330 MT in the period 2007–2009 covering an area of 800,531 ha to 885,800 ha [1]. Ghana currently produces about 12,260,000 MT of cassava annually. Out of this, 8,561,700 MT is available for human consumption while national consumption is estimated at only 3,672,700 MT resulting in surplus of about 4,889,000 MT which can be exploited for the production of industrial starch or ethanol.

Despite its importance, the potential of cassava as an industrial crop has not been exploited to any appreciable extent in Ghana, with the perception that cassava depletes soils [2, 3]. However, recent studies in the forest/savannah transitional agroecological zone as well as the semideciduous forest zone of Ghana have demonstrated that, when integrated in the cropping system as a form of rotation, cassava has the potential of maintaining soil fertility.

In most parts of Africa, cassava is planted just before the land is left to fallow [4, 5]. In the forest/savannah transitional agroecological zone of Ghana, farmers often rotate maize with cowpea and when they observe decline in productivity, the land is cropped to cassava for a period ranging between 12 to 18 months after which the maize/cowpea rotation is resumed [6]. Farmers in Benin also use cassava as a "strategy for regenerating soil fertility" [7], and the term used for cassava cultivation in Benin "jachère manioc" literally means "cassava fallow". According to [8], cassava is frequently grown on marginal soils. This is attributed to increasing population densities which often result in land pressure and successively shorter fallow periods thereby compelling farmers to allocate more of their land to cassava production [9]. Cassava is also frequently grown on marginal soils

because of its efficiency in nutrient capture and removal [10]. In the forest/savannah transitional agroecological zone of Ghana where cassava is widely grown and used as soil fertility regenerating crop, cassava cultivation is intertwined with several factors such as ethnicity, access to resources (including labour, cash, and land), gender, and wealth.

This paper discusses the role of cassava cultivation in soil fertility management and its implication for farming system sustainability and industrialization.

2. Material and Methods

The research study on which this paper is largely based was part of a larger research program called "Convergence of Sciences—inclusive technology development for a better integrated crop and soil management" (CoS), which was implemented by University of Ghana for Ghana and Universite d'Abomey Calavi for Republic of Benin with technical backstopping from Wageningen University and Research Centre, Netherlands.

2.1. The Study Area and Population. The study was conducted in the Wenchi Municipal (7°27 and 8°30 N, 1° and 2°36 W) in the forest/savanna transitional agroecological zone of Ghana. The relief of Wenchi is gently undulating to flat. The soils, which are mainly Lixisols, are fragile with shallow top soils underlain with compact concretions and impermeable iron pans [11]. Temperatures are relatively high with a monthly mean of about 30°C. Rainfall is bimodal and starts in April and ends in November with a dry spell in August. The rainy season is followed by a long dry season from November to March. The annual rainfall is about 1300 mm with about 107 rainy days. Wenchi Municipal, which has a total population of 97,058 (2000 census), is ethnically diverse with about 20% of the population being migrants from the three northern regions of Ghana and the neighbouring Burkina Faso.

2.2. Research Approach and Methods. Wenchi was selected after an initial exploratory study carried out according to the ideas and principles of "technography" [12], which revealed the existence of local soil fertility management strategies, some of which seemed to contradict with dominant scientific beliefs. Among these was the inclusion of late maturing cassava varieties in rotational sequences in the cropping systems in the area as a soil fertility management strategy [13]. This study was thus conducted to explore the efficacy of the farmers' soil fertility management strategies and their relevant social context.

In order to ground the research in the needs of the farming communities, a diagnostic study was carried out in the study area between July 2002 and July 2003 using Participatory Rural Appraisal tools such as drawing of a community territory map (to identify the differences in soil fertility patterns), a transect walk (to reveal the diversity of the landscape), and analysis of soil fertility management strategies and group discussions. Group discussions (10–40) were held in the village centre and/or on farmers' fields.

In addition, two sets of individual interviews with farmers were conducted to collect qualitative and quantitative data. In the first interview, which involved 40 farmers, the selection of farmers was done through stratified sampling. A list of farmers in the community was obtained from the village committee secretary and every tenth name from the list was selected for individual interviewing. The second interview which involved 38 farmers was conducted later to look at the farming characteristics of the various subcommunities in the village using a wealth ranking exercise. For this interview, 6–10 persons were selected from each wealth category within each subcommunity. The individual interviews were semistructured in nature and served both to get more quantitative data on farm size, household composition and the farming system, and to obtain a better qualitative understanding of the soil fertility management strategies and their underlying rationale.

The diagnostic study was followed by farmer participatory on-farm experimentations with three (3) farmer research groups established soon after the diagnostic study to evaluate the agronomic efficacy of the soil fertility management practices being used by the farmers. Six cropping sequences: cassava cropping; pigeonpea cropping; mucuna/maize/mucuna rotation; cowpea/maize/cowpea rotation; maize/maize/maize; and *Imperata cylindrica* fallow were evaluated on both farmer-managed and researcher-managed plots for their effects on soil fertility and yield of subsequent maize test crop. To deepen our understanding of soil fertility management, we carried out further exploration of diversity among the farmers according to gender, ethnicity, and wealth. Farmers were selected from three communities in Wenchi according to ethnicity and gender for interview using semistructured questionnaires. We conducted two sets of interviews. For the first interview, the native households were categorised into male-headed and female-headed households. Subsequently, a stratified sample was selected consisting of 20 males from male-headed households, 20 females from male-headed households, and 20 females from female-headed households. In the case of the migrants, every farmer in the community was interviewed because of the small size of their population. As migrant women do not have their own farming enterprises, only males were interviewed. In the second interview, the farmers were selected through a wealth ranking exercise. Fifteen farmers were selected from each of three wealth categories for interviewing. In addition focus group discussions were held with chiefs, community leaders, family heads, and opinion leaders about land tenure systems in Wenchi.

3. Results and Discussion

3.1. Land Tenure and Cropping Systems in Wenchi. Four main types of holders of land were identified in Wenchi. These were as follows.

(1) The chief's holding known as the stool land or the traditional land. This is the land the chief holds in trust for the stool. These lands are managed by the "Abusahenes" (literally meaning share cropping chiefs) who are responsible

for managing the chief's natural resources, especially land in the traditional area.

(2) Family lands. This refers to the lands that belong to individual extended families or lineages. The family land is usually put under an Abusuapanyin (the head in the line of the inherited siblings) who administers the family land and distributes it among the other siblings with rights in the land.

(3) Individual lands. These are the lands that the first native individual was able to acquire and cultivate. Individual lands are also acquired as gifts from parents.

(4) Government lands. These refer to lands under reafforestation by the Forestry Services Division of the Forestry Commission of Ghana. These lands are given out to prospective farmers to grow their food crops while planting and maintaining trees for the commission. This form of arrangement whereby tenant farmers are given land to plant their food crops by the forestry commission while planting and tending trees for the commission is known as *taungya*.

Access to land for farming in Wenchi involves a spectrum ranging from rights acquired through renting to right of use of a piece of land temporarily. Traditionally, ownership of land is based on kinship, but vested in the traditional authority. Among the Akans in Wenchi, a system of family land exists in which having brought a virgin forest land under cultivation yields rights of usufruct "ownership" as long as the land is kept within a long duration of cultivation. Thus rights could be passed on to the next generation, where it now becomes a family land. Members of the matrilineal family who cleared the land have the right to farm the land. Both men and women in the family have usufruct right in the land. One can also gain access to patrilineal family land. Since migrants who settle permanently cannot own land in the community, the current land tenure arrangement implies that migrants can only access land for farming through renting, sharecropping, or *taungya*.

Land renting is by far the dominant form of contractual arrangement by which migrants gain access to land in Wenchi. Land can be rented from a family, an individual or stool. For family and individual lands, the land is usually rented for a period of 1-2 years and occasionally 3–5 years, depending on the financial needs of the landowner. When an immediate cash need arises, especially for unexpected emergencies such as funerals, marriages, medical bills, court cases, and construction works, land is usually rented out beyond 2 years. Rent is paid in advance before the tenant is allowed to cultivate the land. Advance payment is partly due to the fact that landowners prefer to receive the agreed upon rent as soon as possible before it loses its value. Lands are rented out for short periods because of fear of overexploitation of the land by migrants.

Farmers produce about six to eight different types of crops in Wenchi (Figure 1). The most important food crops in terms of area cultivated were maize, cassava, and yams while cowpea and groundnut were the major grain legumes in the cropping system. Tree crops were restricted to cocoa and cashew. In terms of plot size, maize is the most important crop in Wenchi with both the natives and migrant farmers, having plot size of about 1.3 and 2.2 ha, respectively (Table 1). Natives and migrants differ mainly with respect to the cultivation of tree crops as well as long duration crops such as pigeon pea and cassava. For cassava, the average acreage for natives is about 0.75 ha while it is only 0.3 ha for migrants. This is closely related to social dynamics around land tenure (both security and duration of tenure).

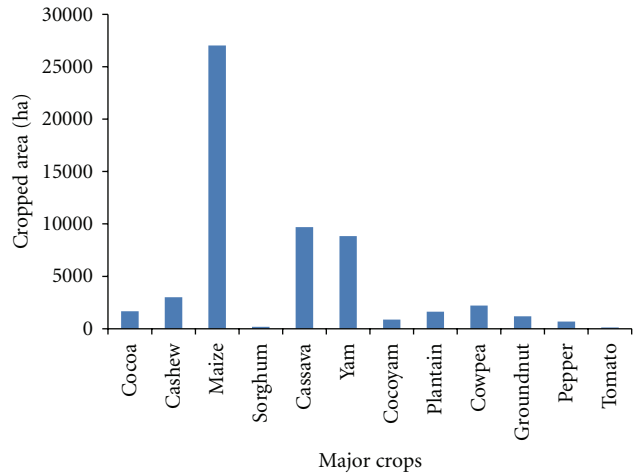


Figure 1: Area cropped (ha) to major crops in Wenchi (Source: MoFA, Wenchi).

3.2. Indigenous Soil Fertility Management Strategies and Their Use. Farmers in Wenchi use the following cropping practices for maintaining the productivity of their farmlands: rotations involving cassava; rotations involving legumes such as cowpea, groundnut, and pigeon-pea; and mounding or ridging.

3.2.1. Crop Rotation in General. Farmers believe that different crops feed from different depths and on different nutrients in the soil. Hence they tend to rotate or intercrop different crops on the same piece of land when they observe yield decline of a particular crop.

3.2.2. Rotation Involving Pigeon Pea. Pigeon pea is usually grown as intercrop with other food crops such as maize, yam, and cassava in a form of relay. Pigeonpea is usually the last crop that is cultivated in this system. After harvesting the yam, maize, and the pigeon pea, the pigeonpea is cut back to allow the cassava to grow to maturity. When the cassava is harvested, the land is allowed to fallow under the pigeon pea for another one or two years, after which it is cut down, burnt and the land replanted to yam, maize, cassava, and pigeonpea again. From the point of view of the farmers, pigeonpea canopy protects the soil from the direct action of the sun and therefore prevents the soil from becoming hardened. According to the farmers, pigeonpea forms canopy after one year and shades out obnoxious weeds by suppressing their growth. The farmers also explained that leaf litter covers the soil, reduces soil erosion, improves infiltration, prevents heating of the soil, and enhances earthworm activity. Crops grown after pigeonpea, especially maize, are perceived by farmers to look greener, grow faster, and yield more.

Table 1: Mean acreage (ha) for selected crops in three communities in Wenchi District.

Community	N	Root and tubers		Cereals		Legumes		Tree crops	
		Cassava	Yams	Maize	Sorghum	Cowpea	Pigeonpea	Cocoa	Cashew
Asuoano	37	0.76	0.68	1.70	0	0.28	0.10	0.40	0.10
Beposo	37	0.65	0.56	1.80	0	0.16	0.12	0.20	0.22
Konkomba*	30	0.20	0.44	1.60	0.1	0.20	0	0	0.03
Residential status									
Native	58	0.75	0.60	1.3	0	0.21	0.10	0.33	0.2
Migrants	46	0.30	0.50	2.2	0.01	0.20	0	0	0.10

* Predominantly migrant community. Source: Adjei-Nsiah (Unpublished).

3.2.3. Bush Fallows. In this case, when farmers observe a decline in fertility of their soils after cropping for three to four successive years, they allow the land to lie fallow for 2-3 years before they go back and crop the land again. According to the farmers, fallowing the land for 2-3 years allows the land to regenerate its fertility. They mentioned that as the land is allowed to fallow, young trees begin to grow and shade the soil so that the land is not exposed to the direct action of the sun thereby keeping the soil moist all the time. They also reason that during the fallow period the litter of the vegetation on the land fertilises the soil as it decomposes.

3.2.4. Rotation with Cowpea. Farmers rotate maize with cowpea, which has a growing period of about 60–70 days, because of its food value and marketability and to maintain the fertility of their farmlands. According to the farmers, maize grown after cowpea grows faster and yields higher even if inorganic fertilizer is not applied to the maize. They mentioned that the nodules formed on the roots contain "energy" which is released for the growth of the maize when they decompose. Farmers also attribute the yield increase in maize after cowpea, to an increase in fertility of the soil as a result of the decomposition of the cowpea foliage that is left on the land after harvest. However, they remarked that if the land is not immediately used for cropping after harvesting the cowpea the fertility of the land is lost since cowpea foliage decomposes rapidly.

3.2.5. Construction of Ridges and Mounds. Farmers construct ridges or mounds on less fertile plots on fallowed land. On grasslands, farmers either plough the land and/or construct mounds or ridges. Farmers construct mounds or ridges or plough their land for two reasons: firstly, to control problematic weeds that invade the land as a result of decline in fertility, and, secondly, to improve the productivity of the soil. As they construct the ridges or mounds, the weeds and leaves on the land mix with the soil and fertilize the soil as they decompose. Farmers reason that the decomposed weeds and leaves when mixed with the soil improve the fertility of the soil and increase the yield of maize planted. According to the farmers, the construction of the mounds and ridges also loosens the soil, which becomes compact after continuous cropping. This allows water to percolate into the soil when it rains.

Table 2: Percentage of native and migrant farmers at Asuoano in 2002 practising various soil fertility management strategies.

Strategy (%)	Native farmers $N = 22$	Migrant farmers $N = 16$
Cassava cropping	82	44
Bush fallow	77	19
Pigeonpea	59	6
Rotation with cowpea/groundnut	18	50
Mounding/ridging	14	100

Source: [6].

3.2.6. Rotation Involving Cassava. Farmers often crop a piece of land for a period ranging from three to four years to maize and cowpea and when they observe decline in the fertility of the soil, they crop the land to cassava for 18–24 months after which they resume their maize/cowpea rotation. The farmers attribute the role of cassava in soil fertility regeneration to its ability to protect the soil from soil erosion through its canopy and its high leaf litter production, which also shades off the soil from the direct action of the sun and thus increases the activities of soil micro- and macroorganisms. The farmers attribute these beneficial roles of cassava to the fact that the varieties of cassava that the farmers grow are the spreading types that form a closed canopy and completely shade off the soil within few months after planting. The use of cassava for soil fertility regeneration is not only peculiar to Wenchi. Reference [7] also reported on the extensive use of cassava for soil fertility regeneration in some parts of Benin.

While the natives widely apply bush fallowing, and rotation involving long duration crops such as cassava and pigeonpea for maintaining the fertility of their farmlands, migrants who do not own land in the communities but depend largely on sharecropping and land renting for gaining access to land for farming use short-term rotational strategies such as rotations involving short duration crops such as cowpea and groundnuts (Table 2).

3.3. Farmers' Agronomic and Social Evaluation of Soil Fertility Management Strategies. Table 3 shows the effect of cropping sequence and N rate on maize grain yield and weed biomass associated with the maize crop 8 weeks after planting

Table 3: Effect of crop sequence and N rate on (a) maize grain and (b) weed biomass associated with the maize crop at 8 weeks after planting on researcher-managed plots.

Crop sequence	N rate (Kg ha^{-1})		Mean
	O	60	
(a)			
Speargrass fallow	1050	2848	1949
Cassava	3002	2738	2870
Pigeonpea	2422	2328	2697
Cowpea-maize-cowpea	1670	2128	1999
Mucuna-maize-*Mucuna*	2970	4195	3582
Maize-maize-maize	1380	2972	1754
Mean	2082	2868	
SED: crop sequence (CS) = 318.4; N Rate (NR) = 115.3; CS × NR = 375.8P < F : CS = 0.001; NR = 0.001; CS × NR = 0.01			
(b)			
Speargrass fallow	585	790	686
Cassava	270	300	285
Pigeonpea	390	500	445
Cowpea-maize-cowpea	325	395	360
Mucuna-maize-*mucuna*	300	345	323
Maize-maize-maize	240	430	335
Mean	351	460	
SED: Crop sequence (CS) = 65.9; N Rate (NR) = 52.4; CS × NR = 112.1P < F : CS = 0.001; NR = 0.05; CS × NR = NS			

Source: [14].

[14]. According to these data, yields of maize ranged from 1.0 t ha^{-1} with spear grass fallow to 3.0 t ha^{-1} with plots previously under cassava when mineral fertilizer was not applied to the maize and on the fertilized plots yields ranged from 2.1 t ha^{-1} with the continuous maize plot to 4.2 t ha^{-1} with plots previously under *mucuna*/maize rotation. The cropping sequences did not have significant effects on soil chemical properties. Lower weed biomass was also associated with the maize crop grown on plots previously under cassava cropping. Weed biomass after cassava in the unfertilized plots was roughly half that found after the speargrass fallow and further reduced to a third of that found after speargrass fallow when cassava was followed by maize with fertilizer.

The beneficial effects of cassava on maize grain yield were mainly due to the relatively high amount of recycled N returned to the soil through the leaf litter and green leafy biomass of cassava which was ploughed back into the soil just before the maize test crop was planted. According to criteria set by [15], cassava litterfall is an important source of easily mineralisable N due to its high N (2.5%) and low lignin content, resulting in high decomposition rates [16]. While it is true that the major beneficial effect of cassava on subsequent maize crop was due to the high N cycling properties of cassava litter and the green leafy biomass which was returned into the soil just before the maize test crop was planted, we cannot exclude the potential role of mycorrizal associations as suggested by the very large initial effect of cassava on maize dry matter yield at 3 weeks after planting [14]. Beneficial effects of higher mycorrizal inoculums at the start of the crop season have repeatedly been reported for maize [17]. Unfortunately, mycorrizal associations were not studied. Other possible effects may include reduction in weed incidence as a result of the suppression of weeds by the cassava canopy. In agricultural systems, shade suppresses weeds growing on the site and interrupts continuous reseeding of the field [18] Cassava/maize rotation resulted in the highest return on investment both when N fertilizer was applied to the maize crop or not for the 2-year period (Table 4). This was due to low input use and labour requirement of cassava as well as the high cassava root yield obtained in this study which was around 31 t ha^{-1} which was far above the current national average of 14 t ha^{-1} [1].

While both native male and female farmers prefer cassava/maize rotation, migrant farmers prefer rotation involving cowpea (Table 5). Migrant farmers cite market and tenure insecurity as reasons for preferring cowpea/maize to cassava/maize rotation and this is reflected in the total farmland allocated to cassava by migrant farmers compared to native farmers (Table 1). Among the migrants, ethnicity, history, and context of migration as well as quality of relationships with the native community also played a major role in soil fertility management [19]. Migrants who had stayed in Wenchi for a longer period and considered their stay in Wenchi as permanent had managed to build long-standing relationship with the natives and had relatively secured and long-duration access to land and tended to use rotation involving cassava for maintaining soil fertility. There was one group of migrants who tended to look at their stay in Wenchi as temporal which had implication for soil fertility management. This group of migrants, although did not own land, tended to have large farms and seemed to succeed in accumulating wealth on the basis of soil mining. Native farmers, particularly women, on the other hand prefer cassava/maize rotation to cowpea/maize rotation due to the flexibility in the labour requirement and minimal use of external input in the cultivation of cassava as well as the role of cassava in food security [9, 19].

3.4. Implication of Large Scale Cassava Cultivation for System Sustainability. In farming systems with minimal application of external inputs, management of organic resources plays a major role in maintaining both nutrient availability and soil organic matter [15]. In a cereal-based farming system as that found in the forest/savanna transitional agroecological zone of Ghana, where external input use is minimal, most recycling of N and P occurs through cassava litterfall and green leafy biomass of cassava incorporated into the soil after cassava harvest [16, 20]. Cassava litterfall and green leafy biomass of cassava are important sources of easily mineralizable N due to their high nitrogen (2.5 and 3.5% for litterfall and green leafy biomass resp.) leading to high decomposition rates [14]. Maize roots may also benefit from cassava association with mycorrizal [17]. Thus, rotation involving cassava, within smallholder agriculture, has the

Table 4: Estimated costs of production, gross revenue and returns on investment of (a) various crop sequences (b) maize grown after the sequences with N application to the maize and (c) maize grown after the sequences without N application to the maize.

Crop sequence	Economic yield (kg ha^{-1})	Total revenue (US$)	Cost of production (US$) Land	Input	Labour	Total cost	Net revenue	Return on investment
(a) Crops in the sequence								
[1]Cassava	31,000	2545.1	41.7	41.7	635.0	718.4	1826.7	254
[2]Pigeonpea	1,870	623.3	41.7	8.3	221.5	271.5	351.8	130
[3]*Mucuna*-maize-*mucuna*	2,016	365.1	41.7	41.7	247.4	330.8	34.3	10
[4]Cowpea-maize-cowpea	2,536 *(1,230)	1079.0	41.7	106,1	475.1	622.9	456.1	73
[5]Maize-maize-maize	3,287	595.2	41.7	36.1	386.1	463.8	456.1	28
[6]Speargrass fallow	0	0	41.7	0	0	41.7	−41.7	−100
(b) Maize after crop sequence with N application								
CS 1	2,738	495.9	13.9	104.2	190.2	308.3	187.6	61
CS 2	2,974	538.5	13.9	104.2	196.5	314.6	223.9	71
CS 3	4,194	759.4	13.9	104.2	245.9	364.0	395.4	108
CS 4	2,331	422.1	13.9	104.2	177.0	295.1	127.0	43
CS 5	2,126	385.0	13.9	104.2	175.4	293.5	91.4	31
CS 6	2,848	515.7	13.9	104.2	224.4	342.5	173.3	51
(c) Maize after crop sequence without N application								
CS 1	3,000	543.2	13.9	13.9	175.6	203.4	339.8	167
CS 2	2,423	438.8	13.9	13.9	165.5	193.3	245.5	127
CS 3	2,961	537.7	13.9	13.9	209.7	237.5	300.2	126
CS 4	1,772	302.8	13.9	13.9	155.2	183.0	119.8	66
CS 5	1,380	249.9	13.9	13.9	153.0	180.7	69.1	38
CS 6	1,048	189.8	13.9	13.9	173.7	200.9	−11.1	−6

[1]US$ 82.1 t^{-1}.
[2]US$ 333.3 t^{-1}.
[3]US$ 337.5 t^{-1} for cowpea and US$181.1 t^{-1} for maize.
*Yield of maize.
CS 1: cassava; CS 2: pigeonpea; CS 3: *Mucuna*-maize-*Mucuna*; CS 4: cowpea-maize-cowpea; CS 5: maize-maize-maize; CS 6: speargrass fallow (source: [14]).

potential of maintaining a reasonable supply of N and P to cereal crops, particularly maize considering the minimal use of external inputs in a maize-based farming system.

Although nutrient recycling is expected to be improved through incorporation of litterfall and harvested crop residues into the soil, it is likely that promoting cassava as an industrial crop may accelerate the depletion of nutrient stocks, particularly potassium through root harvest. Thus on the long term, K may become the most limiting nutrient especially on soils with K values as low as 0.2 cmol K kg^{-1} as those found in Wenchi [10, 14]. Increasing K input through mineral fertilizer application is difficult as potassium fertilizers are not readily available in rural markets, and smallholder farmers hardly apply fertilizers to cassava. Long-term K balances are therefore needed to address this issue. However, K removal may be reduced by half if the cassava stems are not removed from the field for planting [21].

In a place like Wenchi, where the population of the farming community is very heterogeneous, if we want to contribute to better conditions for farming system sustainability through large scale integration of cassava in the farming system, efforts must be oriented to design a range of social arrangements that will meet the specific needs and circumstances of different categories of people. One of such social arrangements is to invest in negotiation of and experimentation with new kinds of contractual and/or land tenure arrangements, involving also supporting control and sanctioning systems. There is also the need to work towards new institutional arrangements that may contribute to reduction of uncertainties and conflicts around land tenure. Possibility in this respect may include (a) contractual provisions for renting that create a link between the level of rent and the level of revenue obtained, (b) clear and agreed-upon rules as to who can contract out what land and who should be involved as witness, (c) increased involvement of local authorities in validating contracts, and (d) strengthening customary institutions to manage land-related conflicts at the local level.

4. Conclusion

The study shows the importance of cassava in the predominantly maize-based farming system of Wenchi, Ghana. Our

Table 5: Preferential ranking of different soil fertility management practices by native and migrant farmers in Wenchi.

Management practice	Ranking order							
	Natives				Migrants			
	Asuoano[a] $N = 10$	Beposo[b] $N = 5$	Droboso[c] $N = 7$	Average	Asuoano[d] $N = 6$	Beposo[d] $N = 6$	Droboso[e] $N = 5$	Average
(a) Ranking by natives and migrants								
Cassava	1	1	1	1	2	2	1	1.7
Pigeonpea	2	5	2	3	4	4	4	4
Mucuna/maize/*Mucuna*	7	6	4	5.7	5	6	6	5.6
Groundnut/maize/groundnut	4	3	3	3.3	3	3	3	3
Cowpea/maize/cowpea	3	2	5	3.3	1	1	2	1.3
Maize/maize/maize	8	7	6	7	7	7	7	7
Cowpea/cowpea/cowpea	5	4	7	5.3	6	5	5	5.3
Bush fallow	6	8	8	7.3	8	8	8	8
(b) Ranking by female and male Bonos	Females $N = 13$	Males $N = 10$						
Cassava	1	1						
Pigeonpea	2	3						
Mucuna/maize/*Mucuna*	5	7						
Groundnut/maize/groundnut	3	4						
Cowpea/maize/cowpea	4	2						
Maize/maize/maize	8	8						
Cowpea/cowpea/cowpea	7	5						
Bush fallow	6	6						

[a]Consisted of 6 males and 4 females; [b]Consisted of 4 males and 1 female; [c]Consisted of 6 females and 1 male; [d]Dagarbas; [e]Walas (source: [14]).

study has shown that cassava plays an important role in the predominantly maize-based farming system in Wenchi partly due to its nutrient recycling properties and also partly due to its role in food security as well as its flexibility in external input use and labour requirement. Even when there is no strong market demand for cassava, farmers still integrate cassava in their rotational system. As more farmers are resorting to putting their land under cassava than fallowing in the farming system, cassava cultivation could therefore serve as an entry point for farming system sustainability. There is however the need to (i) study the long term K balances to address the issue of K losses through removal of storage roots; (ii) evaluate the nutrient recycling capacities of different cassava genotypes; (iii) develop crop rotation/sequencing and soil management options that can improve and/or sustain the productivity of cassava through integrated soil fertility management (ISFM); (iv) design a range of social arrangements that will encourage investment in soil fertility through integration of cassava in the farming system by various categories of farmers in Wenchi. With the rising pressure on land as a result of population increase, the use of external nutrient inputs seems inevitable in the near future.

Acknowledgments

Financial support provided by Interdisciplinary Research and Education Fund (INREF) of Wageningen University and Research Centre, the Netherlands, and the Dutch Ministry for International Cooperation (DGIS) is gratefully acknowledged.

References

[1] MoFA, *Facts and Figure, Agriculture in Ghana*, Ministry of Food and Agriculture, Accra, Ghana, 2009.

[2] M. Sitompul, S. Setijono, J. van der Heid, and M. van Noordwijk, "Crop yields and sustainability of cassava-based cropping systems on an ultisol in Lampung," *Agrivita*, vol. 15, pp. 19–28, 1992.

[3] S. Budidarsono, T. P. Tomich, B. Lusiana, and M. van Noordwijk, "Profitability assessment of transmigration land use system, degraded to imperata cylindrica Grassland," Southeast Asia Policy Research Working Paper no. 6, Bogor, Indonesia, 1998.

[4] F. I. Nweke, D. S. C. Spencer, and J. K. Lynam, *The Cassava Transformation. Africa's Best-Kept Secret*, Mitchigan State University Press, East Lansing, Mich, USA, 2002.

[5] R. J. Hillocks, "Cassava in Africa," in *Cassava Biology, Production and Utilization*, R. J. Hillocks, J. M. Thresh, and A. Bellotti, Eds., pp. 41–54, CABI, Wallingford, UK, 2002.

[6] S. Adjei-Nsiah, C. Leeuwis, K. E. Giller et al., "Land tenure and differential soil fertility management practices among native and migrant farmers in Wenchi, Ghana: implications for interdisciplinary action research," *NJAS—Wageningen Journal of Life Sciences*, vol. 52, no. 3-4, pp. 331–348, 2004.

[7] A. Saïdou, T. W. Kuyper, D. K. Kossou, R. Tossou, and P. Richards, "Sustainable soil fertility management in Benin: learning from farmers," *NJAS—Wageningen Journal of Life Sciences*, vol. 52, no. 3-4, pp. 349–369, 2004.

[8] A. G. O. Dixon, J. M. Ngeve, and E. N. Nukenine, "Genotype x environment effects on severity of cassava bacterial blight disease caused by Xanthomonas axonopodis pv. manihotis," *European Journal of Plant Pathology*, vol. 108, no. 8, pp. 763–770, 2002.

[9] S. Berry, *No Condition is Permanent: The Sociological Dynamics of Agrarian Change in Sub-Saharan Africa*, The University of Wisconsin Press, Madison, Wis, USA, 1993.

[10] R. H. Howeler, "Cassava mineral nutrition and fertilization," in *Cassava Biology, Production and Utilization*, R. J. Hillocks, J. M. Thresh, and A. Bellotti, Eds., pp. 115–147, CABI, Wallingford, UK, 2002.

[11] R. D. Asiamah, T. Adjei-Gyapong, E. Yeboah, J. E. Fening, E. O. Ampontua, and E. Gaisie, "Report on soil characterization and evaluation at four primary multiplication sites (Mampong, Wenchi, Asuansi and Kpeve) in Ghana," Tech. Rep. 200, Soil Research Institute, Kumasi, Ghana, 2000.

[12] P. Richards, TechnographicStudies, "Paper presented at the Initiation Workshop on 'Convergence of Sciences' Project," February 2001, Accra, Ghana.

[13] S. K. Offei, S. K., and O. Sakyi-Dawson, Technographic studies on cowpea as a private sector crop in Ghana, "Report presented at an International Workshop on the 'Convergence of Sciences' project," March 2002, Cotonou, Benin http://www.dpw.wageningen-ur.nl/forum.

[14] S. Adjei-Nsiah, T. W. Kuyper, C. Leeuwis, M. K. Abekoe, and K. E. Giller, "Evaluating sustainable and profitable cropping sequences with cassava and four legume crops: effects on soil fertility and maize yields in the forest/savannah transitional agro-ecological zone of Ghana," *Field Crops Research*, vol. 103, no. 2, pp. 87–97, 2007.

[15] C. A. Palm, K. E. Giller, P. L. Mafongoya, and M. J. Swift, "Management of organic matter in the tropics: translating theory into practice," *Nutrient Cycling in Agroecosystems*, vol. 61, no. 1-2, pp. 63–75, 2001.

[16] S. Adjei-Nsiah, *Cropping systems, land tenure and social diversity in Wenchi, Ghana: implication for soil fertility management*, Ph.D. thesis, Wageningen University, Wageningen, The Netherlands, 2006.

[17] A. O. Osunde, A. Bala, M. S. Gwam, P. A. Tsado, N. Sanginga, and J. A. Okogun, "Residual benefits of promiscuous soybean to maize (*Zea mays* L.) grown on farmers' fields around Minna in the southern Guinea savanna zone of Nigeria," *Agriculture, Ecosystems and Environment*, vol. 100, no. 2-3, pp. 209–220, 2003.

[18] A. de Rouw, "The fallow period as a weed-break in shifting cultivation (tropical wet forests)," *Agriculture, Ecosystems and Environment*, vol. 54, no. 1-2, pp. 31–43, 1995.

[19] S. Adjei-Nsiah, C. Leeuwis, O. Sakyi-Dawson, K. E. Giller, K. E., and T. W. Kuyper, "Exploring diversity among farmers for orienting inter-disciplinary action research on cropping system management in Wenchi, Ghana: the significance of time horizons," *International Journal of Agricultural Sustainability*, vol. 5, pp. 176–194, 2007.

[20] A. M. Fermont, *Cassava and soil fertility in intensifying smallholder farming systems of East Africa*, Ph.D. thesis, Wageningen University, Wageningen, The Netherlands, 2009.

[21] S. Adjei-Nsiah, "Yield and nitrogen accumulation in five cassava varieties and their subsequent effects on soil chemical properties in the forest/savanna transitional agro-ecological zone of Ghana," *Journal of Soil Science and Environmental Management*, vol. 1, pp. 15–20, 2010.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 870948, 5 pages
doi:10.1155/2012/870948

Research Article

Response of Maize (*Zea mays* L.) to Different Rates of Palm Bunch Ash Application in the Semi-deciduous Forest Agro-ecological Zone of Ghana

S. Adjei-Nsiah

Forest and Horticultural Crops Research Centre, Institute of Agricultural Research, College of Agriculture and Consumer Sciences, University of Ghana, P.O. Box 68, Legon, Ghana

Correspondence should be addressed to S. Adjei-Nsiah, y_nsiah@yahoo.co.uk

Received 15 November 2011; Accepted 2 January 2012

Academic Editor: Rosario García Moreno

The effects of palm bunch ash (PBA) and mineral fertilizer application on grain yield and nutrient uptake in maize and soil chemical properties were studied in both the major and minor rainy seasons in the semi-deciduous forest agro-ecological zone of Ghana. In both the major and minor rainy seasons, the response of maize to four levels (0, 2, 4, and 6 tons per hectare) of palm bunch ash and 200 kg per hectare of NPK (15-15-15) application was evaluated using randomised complete block design. Results of the study showed that application of palm bunch ash significantly ($P < 0.05$) increased soil pH, soil phosphorus, and exchangeable cations. Maize grain yield varied significantly ($P < 0.05$) among the different treatments in both the major and minor rainy seasons. The highest maize grain yield of 4530 and 6120 kg ha^{-1} was obtained at PBA application rate of 2 tons ha^{-1} for the major and minor rainy seasons, respectively.

1. Introduction

Empty fruit bunch (EFB) is one of the major waste products generated from processing fresh fruit bunch (FFB) in palm fruit processing mills. About 22% of FFB processed into oil end up as EFB [1]. Currently, Ghana produces about 1,900,000 metric tons of FFB annually [2] which, when processed into oil, generate 418,000 MT of EFB annually. In the large industrial estates, EFB is either incinerated in the mills as a means of getting rid of these wastes' as well as, providing energy for the boilers in FFB sterilization. However, the small-scale mills which process about 60% of the total FFB produced in the country [3] burn the EFB as a means of disposing them, resulting in heaps of ash dotted around small-scale mills in the major oil palm producing areas in Ghana. There is currently no large-scale use for palm bunch ash in Ghana, although it could be used for the manufacture of local soap due to its high potassium content. The palm bunch ash (PBA) produced by burning EFB, which constitutes about 6.5% by weight of the EFB, contains 30–40% K_2O [1] and could thus be used as source of potassium fertilizer. Most soils in the forest part of southern Ghana where oil palm is cultivated are acidic due to the nature of the parent material, high rainfall regime, intensity, and associated leaching of nutrients which requires sustainable liming. Preliminary analysis of bunch ash of different ages from processing mills in Kade (unpublished results) indicates that besides K, palm bunch ash has high pH and contains varying amounts of other nutrients such as calcium (Ca), phosphorus (P), and magnesium (Mg). These properties of palm bunch ash make it suitable as a liming material and fertilizer supplement. Studies [4–6] have shown that application of wood ash significantly increased the effective cation exchange capacity and base saturation and decreased the concentration of exchangeable aluminium in the soil. In southern Nigeria, [7] and [8] found palm bunch ash as an effective fertilizer and liming material for increasing soil fertility, pH, and nutrient uptake by crops such as maize and cassava.

In Ghana, mineral fertilizers are rarely used by smallholder farmers due to prohibitive cost as a result of privatization and removal of government subsidies [9]. In recent

Table 1: Chemical and physical soil characteristics of surface soil (0–20 cm) of experimental plots before planting.

pH	OC	Total N	P Bray	K	Mg	Ca	Na	Al + H	Sand	Silt	Clay
(1 : 1 H_2o)	%	%	ppm			Me 100 g^{-1}				%	
4.8	2.2	0.22	6.78	0.64	1.87	5.85	0.21	1.00	40	52	8

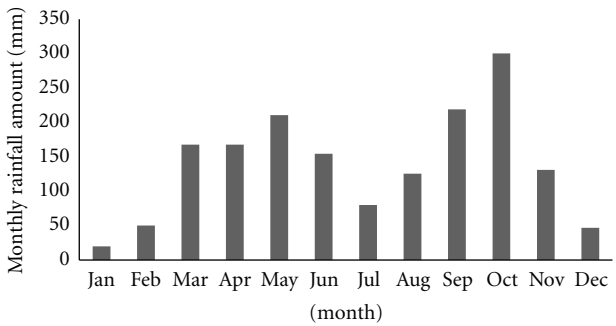


Figure 1: Monthly rainfall distribution at the study site during the experimental period.

years, there has been a growing interest in the tropical world in using crop residues for improving soil productivity in order to reduce the use of external inputs of inorganic fertilizers [10–12]. Moreover, there is abundance of palm bunch ash in the oil palm belt of southern Ghana where the present study was carried. These reasons necessitated the need to research into the possible use of palm bunch ash as liming material and fertilizer supplement for improving the productivity of staple food crops grown in this region. The objective of this study was to investigate the effect of palm bunch ash and NPK fertilizer on the yield and nutrient content of maize (Zea mays) and soil chemical properties in the semi-deciduous forest zone of Ghana.

2. Materials and Methods

2.1. Study Site. The study was carried out at the Forest and Horticultural Crops Research Centre, Kade which lies within latitude 6° 09′ and 6° 06′ N and longitude 0° 55′ and 0° 49′ W in the Kwaebibirem district of the Eastern Region of Ghana. The centre which is located in the semi-deciduous forest agro-ecological zone of Ghana is 135.9 m above sea level. The study site is characterized by a bimodal rainfall pattern with peaks in June and October with a short break in August and a dry season from December to March. The total annual rainfall amount during the experimental period as presented in Figure 1 was 1672.2 mm. The soils at the experimental site which are mainly forest ochrosol derived from precambium phyllitic rocks [13] are deep and well drained and are generally classified as Acrisols in the FAO-UNESCO Revised Legend [14]. The chemical and physical properties of the surface soil of the experimental plots are presented in Table 1.

2.2. Experimental Layout. The experimental plot which was dominated by *Chromolaena odorata* had been fallowed for 1 year. Cassava had been grown on this field earlier. The *C. odorata* was initially cleared by slashing with a cutlass. Four weeks later, herbicide (glyphosate) was applied at the rate of 900 g a.i ha^{-1}. The trial which was conducted in a randomised complete block design consisted of five treatments replicated four times in four blocks. The treatments which were applied to a local maize variety, *Obatanpa* were 0 tha^{-1} PBA, 2 tha^{-1} PBA, 4 tha^{-1} PBA, 6 tha^{-1} PBA, and 200 kg ha^{-1} NPK (15-15-15). The experiment was carried out in two seasons: the major rainy season which starts from April and ends in July and the minor rainy season which starts in August and ends in November.

In the major rainy season, the maize was planted on 25 April, 2010 at a spacing of 1 m by 50 cm at 3 seeds per hole which was thinned to 2 seeds per hole at 10 days after planting. Plot size was 15 by 5 m giving a plant population of 4 plants/square meter. In the minor rainy season, the plot size was 15 by 5 m and the maize was planted on 19 August, 2010 at 1 m by 20 cm at 3 seeds per hole which was thinned to one plant per hole 10 days after planting giving a plant population of 5 plants/square meter. The PBA was applied at 10 days after planting in a form of ring.

At tasseling, maize plant height was measured from the ground level to the point of the plant from where the tassel emerges and ear leaf samples were collected and oven-dried at 65°C for 3 days and milled for analysis. At maturity, maize ears and stover were harvested from the three middle rows leaving 1 m border at both ends. The cobs were weighed and a subsample of 10 cobs per plot was taken, weighed, and oven-dried at 70°C for 2 days. The grains were then removed and weighed again to determine the dry matter (DM). The stover was weighed fresh and subsample taken to determine the DM.

2.3. Leaf and Soil Analysis. Maize ear leaf samples collected at tasseling were analysed for N, P, K, Ca, and Mg. The N was determined using micro-Kjeldal method, P by molybdenum blue calorimetric, K by flame photometer, and Ca and Mg by atomic absorption spectrophotometer.

Prior to commencement of the trial, surface soil (0–20 cm) samples were collected from the experimental site and analyzed for both chemical and physical properties. During the harvest, soil samples were also collected from the 0–20 cm depth of each plot and analyzed for soil chemical properties. Soil pH was determined in water suspension at 1 : 1 ratio; organic C by Walkley-Black procedure; total N by Kjeldahl method; available P by Bray-1 method and exchangeable bases (K, Na, Ca, and Mg) by 1 M NH4 OAC method [15].

To reduce cost, only soil and ear leaf samples collected during the major rainy season planting were analysed and presented in Tables 3 and 4, respectively.

Table 2: Chemical properties of the palm bunch ash used in the experiment.

pH	OC	Total N	P Bray	Exchangeable cations			
				K	Ca	Mg	Na
(1 : 2.5 H_2o)		%	ppm	me 100 g^{-1}			
10.90	0.55	0.08	270.27	583.42	35.24	29.24	20.51

Table 3: Chemical properties of the 0–20 cm layer of the soil 110 days after application of the palm bunch ash and NPK for the major rainy season planting.

	pH	OC	Total N	P Bray	K	Mg	Ca	Na	Al + H
	(1 : 1 H_2o)	%	%	ppm	Me 100 g^{-1}				Me 100 g^{-1}
0 tha^{-1}	5.10	1.26	0.11	7.04	0.50	2.76	5.62	0.17	1.00
2 tha^{-1}	5.83	1.57	0.16	19.77	1.23	2.63	4.27	0.19	0.45
4 tha^{-1}	5.93	1.43	0.15	16.00	1.29	3.78	6.31	0.30	0.57
6 tha^{-1}	5.90	1.86	0.17	22.89	0.76	3.25	5.07	0.26	0.50
NPK	5.08	1.34	0.15	14.72	0.63	1.47	5.58	0.11	1.00
LSD	0.49	0.78	0.057	6.63	0.47	1.48	2.02	0.07	0.054
Prob.	1%	NS	NS	1%	1%	5%	NS	0.1%	1%

2.4. Statistical Analysis. Data were subjected to analysis of variance (ANOVA) using the general linear model (GLM) procedure [16].

3. Results and Discussion

Results of the initial soil analysis for the experimental site are presented in Table 1. The soil of the experimental site was strongly acidic, moderately high in N and exchangeable Ca and Mg. The PBA had pH of 10.90 (H_2O 1 : 2.5), 0.55% organic carbon, 0.08% N and 35.24, 29.24, and 583.42 me/100 g soil exchangeable Ca, Mg, and K, respectively (Table 2). The pH value for the PBA reported in this study is significantly higher than that reported by [7] who reported a value of 8.8. The high pH of the PBA used in the present study could be attributed to the fact that it was a fresh ash and had not been exposed to rain. Preliminary studies carried out at Kade, Ghana shows that, when PBA is exposed to rain for a long time, the pH goes down (unpublished results) probably as a result of leaching of cations. The pH of the experimental plot at the start of the experiment was 4.8 (Table 1) compared to an average of 5.8–5.9 at 110 days after the application of PBA (Table 3). The pH of 4.8 of the soil at the start of the experiment suggests the soil to be strongly acidic according to [17] and hence, the justification for the investigation into the possible use of PBA as soil amendment.

The increase in the pH of the soil after the application of the PBA was due to the high pH level of the PBA. PBA is alkaline and contains relatively high values of Ca and Mg and thus has a liming effect on the soil. The increase in soil pH was also due to decrease in Al^{3+} as a result of precipitation of Al as hydroxyl-Al [6] as ash has been found to contain oxides and hydroxides of potassium, sodium, calcium, and magnesium [18] resulting in low exchangeable acidity in the ash-amended plots (Table 3). This could also be responsible for the significantly higher potassium, sodium, calcium, and magnesium levels in the PBA amended plots compared with the control [12]. The high soil OC and nutrient contents of the PBA-treated plots compared with the control plots also confirms the findings of [7] who reported significant increase in OM and nutrient contents of acid soils after application of PBA. The increased available P content of the soil with increased application of PBA could be attributed to release of P from complexes of Al and Fe under increasing soil pH [6]. The increase in soil nutrients as a result of application of PBA could also be attributed to increased microbial activities in the soil and increased organic matter production with its concomitant increased availability of N, P, K, and Mg [4, 19]. Data on ear leaf analysis as shown in Table 4 indicate that compared with the control, increased application of PBA resulted in decreased leaf nutrient content except P and K which increased with application of PBA. The reduction in leaf nutrient content, especially for N, Mg, and Ca with increasing application of PBA could be attributed to excessive uptake of K [4, 20]. Table 5 shows that PBA application significantly affected the maize plant height. The least plant height of 183 and 259 cm in the major and minor rainy seasons, respectively, was recorded in the control plot. Relatively higher plant height was recorded in the minor rainy season.

Palm bunch ash and NPK fertiliser application increased both maize grain yield and stover yield both in the major and minor rainy seasons (Table 5). In the major rainy season fertiliser application resulted in about 100% increase in maize grain yield, while PBA application resulted in about 68–78% increase in maize grain yield over the control. In the minor rainy season mineral fertiliser application resulted in about 21% increase in maize grain yield, while PBA application resulted in between 11–22% increase in maize grain yield over the control. The highest increase in maize grain yield in both seasons was obtained at the application of

Table 4: Effect of palm bunch ash and NPK fertilizer on maize ear leaf nutrient composition for the major rainy season planting.

Treatment	N	P	K	Mg	Ca
			%		
0 tha^{-1} PBA	2.66	0.25	0.87	0.19	0.36
2 tha^{-1} PBA	2.61	0.33	1.00	0.16	0.26
4 tha^{-1} PBA	2.59	0.33	1.00	0.15	0.26
6 tha^{-1} PBA	2.56	0.37	0.98	0.15	0.23
NPK	2.72	0.32	0.96	0.19	0.38
LSD	0.12	0.05	0.18	0.04	0.06
Prob.	NS	1%	NS	5%	0.1%

NS: not significant.

Table 5: Effect of PBA and NPK fertilizer on growth and yield of maize for the major and minor rainy season plantings.

Treatment	Plant height (cm)		Maiza grain yield at 12% moisture content (tha^{-1})		Maiza stover DM (tha^{-1})	
	Major season	Minor season	Major season	Minor season	Major season	Minor season
0 tha^{-1} PBA	183	259	2550	5030	1990	2975
2 tha^{-1} PBA	221	293	4530	6120	3720	4485
4 tha^{-1} PBA	213	291	4480	5830	3470	3825
6 tha^{-1} PBA	232	291	4290	5583	3700	3575
NPK	234	284	5120	6060	4430	4743
LSD	21.0	15.0	1140	763	670	1076
Prob.	0.1%	1%	1%	5%	0.1%	5%

2 tons ha^{-1} PBA. The increase in maize grain yield and root yield of cassava with PBA or Wood ash has been reported by several workers [6, 7, 12, 19–21] who attributed it to increase in soil nutrient content and uptake of nutrients by maize as well as higher organic matter in the ash.

4. Conclusions

The results of this study suggest that pH of acid soils can be corrected and leached nutrients replaced by recycling of palm bunch ash. Application of PBA contributes to improvement in soil chemical properties of acid soils by raising soil pH and the level of macronutrients such as N, P, K, Ca, and Mg in the soil. This may enhance yield of crops through improved nutrient uptake by crops. In both seasons, application of 2 tha^{-1} gave the highest maize grain yield. Thus in oil palm growing areas in Ghana, where soils are acidic, palm bunch ash which are found in abundance in these areas could be used as a liming material and as a fertiliser supplement to improve the yield of staple food crops.

Acknowledgment

The author gratefully acknowledges the financial contribution made towards this work by the Forest and Horticultural Crops Research Centre, Kade of the Institute of Agricultural Research, University of Ghana, Legon.

References

[1] K. H. Lim and A. R. Zaharah, "Decomposition and N and K release by oil palm empty fruit bunches applied under mature palms," *Journal of Oil Palm Research*, vol. 12, pp. 55–62, 2000.

[2] FAO Statistical Databases, 2009, http://faostat.fao.org/site/639/default.aspx.

[3] J. Opoku and F.A. Asante, "Palm oil production in Ghana," Final report on the status of the oil palm industry in Ghana, German Technical Co-operation (GTZ), Accra, Ghana, 2008.

[4] A. Saarsalmi, E. Mälkönen, and S. Piirainen, "Effects of wood ash fertilization on forest soil chemical properties," *Silva Fennica*, vol. 35, no. 3, pp. 355–368, 2001.

[5] B. P. Bougnom, J. Mair, F. X. Etoa, and H. Insam, "Composts with wood ash addition: a risk or a chance for ameliorating acid tropical soils?" *Geoderma*, vol. 153, no. 3-4, pp. 402–407, 2009.

[6] C. N. Mbah, J. N. Nwite, C. Njoku, and I. A. Nweke, "Response of maize (*Zea mays* L.) to different rates of wood-ash application in acid ultisol in Southeast Nigeria," *African Journal of Agricultural Research*, vol. 5, no. 7, pp. 580–583, 2010.

[7] M. A. Awodun, S. O. Ojeniyi, A. Adeboye, and A. S. Odedina, "Effect of oil palm bunch refuse ash on soil and plant nutrient composition and yield of maize," *Eurasian Journal of Sustainable Agriculture*, vol. 1, pp. 50–54, 2007.

[8] S. O. Ojeniyi, P. O. Ezekiel, D. O. Asawalam, A. O. Awo, S. A. Odedina, and J. N. Odedina, "Root growth and NPK status of cassava as influenced by oil palm bunch ash," *African Journal of Biotechnology*, vol. 8, no. 18, pp. 4407–4412, 2009.

[9] H. Gerner, E. O. Asante, E. Owusu Benoah, and K. Marfo, *Ghana Fertilizer Privatization Scheme. Private Sector Role and Public Sector Responsibility in Meeting Needs of Farmers*, IFDC-Africa, Lome, Togo, 1995.

[10] H. Khalid, Z. Zin, and T. M. Anderson, "Nutrient cycling in an oil palm plantation. The effects of residue management practices during replanting on dry matter and nutrient uptake of young palms," *Journal of Oil Palm Research*, vol. 12, pp. 29–37, 2000.

[11] L. S. Ayeni, O. M. Ayeni, O. P. Oso, and S. O. Ojeniyi, "Effect of sawdust and wood ash applications in improving soil chemical properties and growth of cocoa (*Theobroma cacao*) seedlings in the nurseries," *Medwel Agricultural Journal*, vol. 3, pp. 323–326, 2008.

[12] F. O. Adekayode and M. R. Olojugba, "The utilization of wood ash as manure to reduce the use of mineral fertilizer for improved performance of maize (*Zea mays* L.) as measured in the chlorophyll content and grain yield," *Journal of Soil Science and Environmental Management*, vol. 1, pp. 40–45, 2010.

[13] P. M. Ahn, "Soils of the lower tano basin, South Western Ghana," Ghana Ministry of Food and Agriculture, Soils and Landuse Survey Memoir No. 2, 1961.

[14] F.A.O., "FAO Unesco Soil Map of the world," World Soil Resources Report, vol. 60, FAO, Rome, Italy, 1998.

[15] Agronomy Society of America-Soil Science Society of America, *Methods of Soil Analysis, Part 2, Chemical and Microbiological Properties-Agronomy Monograph No. 9*, ASA-SSA, Madison, Wis, USA, 2nd edition, 1982.

[16] SAS, *SAS User's Guide: Statistics*, SAS Institute, Cary, NC, USA, 1996.

[17] N. C. Brady, *The Nature and Properties of Soils*, Macmillan, New York, NY, USA, 9th edition, 1984.

[18] A. Demeyer, J. C. Voundi Nkana, and M. G. Verloo, "Characteristics of wood ash and influence on soil properties and nutrient uptake: an overview," *Bioresource Technology*, vol. 77, no. 3, pp. 287–295, 2001.

[19] S. O. Ojeniyi, B. C. Awanlemhen, and S. A. Adejoro, "Soil plant nutrients and maize performance as influenced by oilpalm bunch ash plus NPK fertilizer," *Journal of American Science*, vol. 6, no. 12, pp. 456–460, 2010.

[20] U. S. Offor, G. I. Wilcox, and C. N. Agbagwaa, "Potentials of palm bunch ash on yield of *Zea mays*," *Journal of Agriculture and Social Research*, vol. 10, no. 2, pp. 132–134, 2010.

[21] P. O. Ezekiel, S. O. Ojeniyi, D. O. Asawalam, and A. O. Awo, "Root growth, dry root yield and NPK content of cassava as influenced by oil palm bunch ash on ultisols of southeast Nigeria," *Nigerian Journal of Soil Science*, vol. 19, pp. 6–10, 2009.

Hindawi Publishing Corporation
Applied and Environmental Soil Science
Volume 2012, Article ID 104826, 13 pages
doi:10.1155/2012/104826

Review Article

Managing the Nutrition of Plants and People

Philip J. White,[1] Martin R. Broadley,[2] and Peter J. Gregory[3, 4]

[1] *Ecological Sciences Group, The James Hutton Institute, Invergowrie, Dundee DD2 5DA, UK*
[2] *Division of Plant and Crop Sciences, University of Nottingham, Sutton Bonington Campus, Loughborough LE12 5RD, UK*
[3] *Centre for Food Security, School of Agriculture, Policy and Development, University of Reading, Reading RG6 6AR, UK*
[4] *East Malling Research, New Road, East Malling, Kent ME19 6BJ, UK*

Correspondence should be addressed to Philip J. White, philip.white@hutton.ac.uk

Received 1 November 2011; Accepted 7 December 2011

Academic Editor: Rosario García Moreno

One definition of food security is having sufficient, safe, and nutritious food to meet dietary needs. This paper highlights the role of plant mineral nutrition in food production, delivering of essential mineral elements to the human diet, and preventing harmful mineral elements entering the food chain. To maximise crop production, the gap between actual and potential yield must be addressed. This gap is 15–95% of potential yield, depending on the crop and agricultural system. Current research in plant mineral nutrition aims to develop appropriate agronomy and improved genotypes, for both infertile and productive soils, that allow inorganic and organic fertilisers to be utilised more efficiently. Mineral malnutrition affects two-thirds of the world's population. It can be addressed by the application of fertilisers, soil amelioration, and the development of genotypes that accumulate greater concentrations of mineral elements lacking in human diets in their edible tissues. Excessive concentrations of harmful mineral elements also compromise crop production and human health. To reduce the entry of these elements into the food chain, strict quality requirements for fertilisers might be enforced, agronomic strategies employed to reduce their phytoavailability, and crop genotypes developed that do not accumulate high concentrations of these elements in edible tissues.

1. Introduction

Food security can be defined as having sufficient, safe, and nutritious food to meet the dietary needs of an active and healthy life [1]. This paper discusses the role of plant mineral nutrition in crop production, the delivery of mineral elements required for human wellbeing, and the prevention of toxic mineral elements entering the human food chain.

Crop production is predicated on the phytoavailability of sufficient quantities of the 14 essential mineral elements required for plant growth and fecundity (Table 1; [2, 3]). These are the macronutrients, nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulphur (S), which are required in large amounts by crops, and the micronutrients chlorine (Cl), boron (B), iron (Fe), manganese (Mn), copper (Cu), zinc (Zn), nickel (Ni), and molybdenum (Mo), which are required in smaller amounts [4]. Deficiency in any one of these elements restricts plant growth and reduces crop yields. In geographical areas of low phytoavailability, these mineral elements are often applied to crops as inorganic or organic fertilisers to increase crop production [2, 3]. However, the application of fertilisers incurs both economic and environmental costs. In some regions, especially those remote from the origin of manufacture, the cost of inorganic fertilisers can constitute a high proportion of total production costs, and vagaries and uncertainties in the price of inorganic fertilisers can prohibit their use [5, 6]. The manufacture of inorganic fertilisers is energy intensive and depletes natural resources, and fertiliser applications that exceed crop requirements can reduce land, water, and air quality through leaching and runoff, eutrophication, and gaseous emissions [7, 8]. Current research in plant mineral nutrition is directed towards developing (1) agronomic strategies that improve the efficiency of fertiliser use by crops and (2) genetic strategies to develop crops with greater acquisition and physiological utilisation of mineral elements [3, 4]. These efforts contribute both to food security and to the economic and environmental sustainability of agriculture.

Humans require sufficient intakes of many mineral elements for their wellbeing [4, 11–13]. In addition to the 14

Table 1: The main chemical forms in which mineral elements are acquired from the soil solution by roots, and the critical leaf concentrations for their sufficiency and toxicity in nontolerant crop plants. The critical concentration for sufficiency is defined as the concentration in a diagnostic tissue that allows a crop to achieve 90% of its maximum yield. The critical concentration for toxicity is defined as the concentration in a diagnostic tissue above which yield is decreased by more than 10%. It should be recognized that critical tissue concentrations depend upon the exact solute composition of the soil solution and can differ greatly both between and within plant species. The latter differences reflect both ancestral habitats and ecological strategies. Data are compiled from references [4, 9, 10].

Element	Form acquired	Critical leaf concentrations ($mg\,g^{-1}$ DM)	
		Sufficiency	Toxicity
Nitrogen (N)	NH_4^+, NO_3^-	15–40	
Potassium (K)	K^+	5–40	>50
Phosphorus (P)	$H_2PO_4^-$	2–5	>10
Calcium (Ca)	Ca^{2+}	0.5–10	>100
Magnesium (Mg)	Mg^{2+}	1.5–3.5	>15
Sulphur (S)	SO_4^{2-}	1.0–5.0	
Chlorine (Cl)	Cl^-	0.1–6.0	4.0–7.0
Boron (B)	$B(OH)_3$	$5–100 \times 10^{-3}$	0.1–1.0
Iron (Fe)	Fe^{2+} Fe^{3+}-chelates	$50–150 \times 10^{-3}$	>0.5
Manganese (Mn)	Mn^{2+} Mn-chelates	$10–20 \times 10^{-3}$	0.2–5.3
Copper (Cu)	Cu^+, Cu^{2+} Cu-chelates	$1–5 \times 10^{-3}$	$15–30 \times 10^{-3}$
Zinc (Zn)	Zn^{2+} Zn-chelates	$15–30 \times 10^{-3}$	$100–300 \times 10^{-3}$
Nickel (Ni)	Ni^{2+} Ni-chelates	0.1×10^{-3}	$20–30 \times 10^{-3}$
Molybdenum (Mo)	MoO_4^{2-}	$0.1–1.0 \times 10^{-3}$	>1
Sodium (Na)	Na^+	—	2–5
Aluminium (Al)	Al^{3+}	—	$40–200 \times 10^{-3}$
Cobalt (Co)	Co^{2+}	—	$10–20 \times 10^{-3}$
Lead (Pb)	Pb^{2+}	—	$10–20 \times 10^{-3}$
Cadmium (Cd)	Cd^{2+} Cd-chelates	—	$5–10 \times 10^{-3}$
Mercury (Hg)	Hg^{2+}	—	$2–5 \times 10^{-3}$
Arsenic (As)	$H_2AsO_4^-$, H_3AsO_3	—	$1–20 \times 10^{-3}$
Chromium (Cr)	Cr^{3+}, CrO_4^{2-}, $Cr_2O_7^{2-}$	—	$1\text{-}2 \times 10^{-3}$

elements that are essential for plants, humans require significant amounts of sodium (Na), selenium (Se), cobalt (Co) and iodine (I) in their diet and possibly small amounts of fluorine (F), lithium (Li), lead (Pb), arsenic (As), vanadium (V), chromium (Cr), and silicon (Si) also. Ultimately, plant products provide humans with the majority of these mineral elements. Unfortunately, the diets of over two-thirds of the world's population lack one or more of these essential mineral elements [13–15]. In particular, over 60% of the world's 6 billion people are Fe deficient, over 30% are Zn deficient, almost 30% are I deficient, and about 15% are Se deficient. In addition, dietary deficiencies of Ca, Mg and Cu occur in many developed and developing countries. Mineral malnutrition is attributed to either crop production on soils with low phytoavailability of mineral elements essential to human nutrition or consumption of staple crops, such as cereals, or phloem-fed tissues, such as fruit, seeds, and tubers, that have inherently low tissue concentrations of certain mineral elements [14, 16], compounded by a lack of fish or animal products in the diet. Soils with low phytoavailability of mineral elements include (1) alkaline and calcareous soils that have low phytoavailabilities of Fe, Zn, and Cu, and comprise 25–30% of all agricultural land [10, 14, 17–21], (2) coarse-textured, calcareous, or strongly acidic soils that have low Mg content [22], (3) midcontinental regions that have low I content [23, 24], and (4) soils derived mostly from igneous rocks that have low Se content [25, 26]. Currently, mineral malnutrition is considered to be amongst the most serious global challenges to humankind and is avoidable [13–15, 27].

The presence of excessive concentrations of potentially harmful mineral elements also compromises both crop production (Table 1) and human health. On acid soils, toxicities of Mn and aluminium (Al) limit crop production [3, 4, 10, 28]. Soil acidity occurs on about 40% of the world's agricultural land [29, 30]. Additionally, Na, B, and Cl

toxicities reduce crop production on sodic or saline soils, which comprise 5–15% of the world's potential agricultural land [31] and toxicities of Mn and Fe can arise in waterlogged or flooded soils [10]. Excessive concentrations of Ni, Co, Cr, and Se can limit growth of plants on soils derived from specific geological formations [10, 32, 33]. In addition, imbalances of Ca, Mg, and K can occur in irrigated agriculture and toxic concentrations of Zn, Cu, Pb, As, cadmium (Cd) and mercury (Hg) have accumulated in agricultural soils in some areas due to human activities [10, 34–36]. Mineral imbalances of Ca, Mg, and K in forage can have serious consequences for the nutrition and health of ruminant animals [14]. Toxic elements contained in produce can accumulate in the food chain with detrimental consequences for animal and human health.

This paper describes how the application of current knowledge of soil science, agronomy, plant physiology, and crop genetics can underpin the production of edible crops that contribute sufficient mineral elements for adequate animal and human nutrition, whilst limiting the entry of toxic elements to the human food chain.

2. Increasing Food Production

The successes of the "Green Revolution" have enabled food production to keep pace with the growth of human populations through the development of semidwarf crops resistant to pests and pathogens, whose yields are maintained through the application of agrochemicals to control weeds, pests, and diseases, mineral fertilisers, and irrigation [3, 37, 38]. It is widely believed that the world currently produces sufficient food for its population, and it is often assumed that food security can be achieved by better distribution and access, driven principally by open markets [39, 40]. In this context, it is often stated that about one-sixth of the world's population are obese, whilst another sixth are starving. The immediate social imperative is, therefore, to redistribute food according to need and, in the future, to maintain food production at rates equal to, or greater than, population growth. The world's population is increasing at a rate of 80 million people a year, and many of these people will live in developing countries [6, 38, 41]. Feeding these people will necessitate significant infrastructural development.

Recent estimates suggest that less than 20% of the increased crop production required in the next two decades could come from the cultivation of new land and about 10% from increased cropping intensity [6, 42]. Thus, food security for the world must be achieved by increasing yields per hectare on the same land area farmed today. It was suggested that average cereal yields needed to increase by about 25% from 3.23 t ha^{-1} in 2005/07 to 4.34 t ha^{-1} in 2050 to feed the world's population [41]. This is a challenging task. The production of food crops is further challenged by increasing demands for animal feeds, fibres, timber, biofuels, landscape amenities, biological conservation, and urban development [6, 38–45]. It is estimated that almost half the world's food production is directly supported by manufactured N-fertilisers and that this reliance will increase as the population of the world grows [8, 12, 46].

2.1. Reducing the Yield Gap. The "yield gap" is the difference between actual and potential crop production. Potential crop production is defined as an idealised state in which an adapted crop variety grows without losses to pests or pathogens and experiences no biophysical limitation other than uncontrollable factors, such as solar radiation, air temperature, and water supply [47, 48]. Yield gaps can range from 15 to 95% of yield potential, depending on the crop grown and the agricultural system employed [38, 47–50]. Irrigated crops often approach 80% of potential yield, whilst rainfed systems deliver a lower percentage of potential yield [47]. Higher inputs realise greater yields and reduce yield gaps [47]. Global aggregated yield gaps are currently estimated to be about 60% for maize, 47% for rice, and 43% for wheat [48]. Agricultural systems can be categorised as either "intensive" or "extensive". Intensive agricultural systems utilise high inputs of fertilisers, agrochemicals, and water, together with effective mechanization, to produce high yields per unit area. Extensive agriculture is associated generally with smallholder farming. It has low inputs of capital and labour and, often, low yields per unit area. Yield gaps are greatest for extensive agricultural systems, which have, therefore, the greatest potential for increased crop production. Extensive agriculture occupies >40% of the world's agricultural land and sustains about 40% of its population [49]. Major contributors to yield gaps include (1) biophysical factors, such as soil texture, pH and mineral composition, drought, flooding, and land topology, (2) biotic factors, such as weed pressures, which can reduce global yields of major crops by 20–40%, and losses to pests and diseases, which can reduce global yields of major crops by 25–50% [51], (3) poor husbandry, such as inferior seed, suboptimal planting rates, inappropriate fertiliser applications, and occurrence of lodging, and (4) socioeconomic factors, such as profit maximization, risk aversion, market influences, lack of capital, infrastructure or labour, and lack of information [38, 39, 47, 48]. Thus, reducing yield gaps will depend on the implementation of improved technologies that address water availability, soil conditions, mineral nutrition, crop protection, and crop husbandry [47].

2.2. Alleviating Constraints on Infertile Soils. Major constraints to crop production occur on alkaline, acid, saline, and sodic soils [4]. These constraints can be addressed by both agronomic measures and by the cultivation of adapted genotypes.

The major constraints to crop production in acid soils are Al and Mn toxicities. Liming, especially with dolomitic lime ($CaMg(CO_3)_2$), is an effective way to raise soil pH to avoid Al and Mn toxicities, and also to avoid Ca and Mg deficiencies [10, 21, 28]. The primary constraint is often Al toxicity, and cultivating Al-excluding or Al-tolerant crops allows agricultural production on acid soils. Plant roots can reduce Al uptake (1) by secretion of organic acids or mucilage from the root to chelate Al in the rhizosphere, (2) by raising rhizosphere pH to reduce the concentration of Al^{3+}, which is the phytotoxic Al species, and (3) by binding Al to cell wall components [28, 52–54]. Aluminium entering plant cells can be rendered nontoxic by sequestration in the vacuole as a

complex with organic acids [28, 52, 53]. Crop genotypes with these attributes can be selected in breeding programmes or created by genetic modification (GM) of elite germplasm [3, 54]. Likewise, there are large differences both between and within plant species in their exclusion and tolerance of Mn, which can be exploited to improve crop production on acid soils [28].

The major constraint to crop production on calcareous or alkaline soils is often the low phytoavailability of Fe, Zn, Mn, or Cu [10, 17, 19, 21, 27, 34]. This can be remedied by supplying these elements as soil or foliar fertilisers. The application of acidifying fertilisers, such as urea, ammonium nitrate, ammonium sulphate, ammonium phosphates, or elemental S, can address soil alkalinity, whilst the introduction of appropriate microorganisms and companion plants, either through intercropping or inclusion in rotations, that increase the phytoavailability of Fe, Zn, Mn, and Cu can increase the yields of crops susceptible to their deficiencies [10, 12, 14, 21]. In addition, since the total concentrations of Fe, Zn, Mn, and Cu in many soils would be sufficient for crop nutrition if they were phytoavailable, cultivating genotypes with greater acquisition or physiological utilisation of these elements can increase crop yields [10, 12, 19, 27, 55]. There is considerable genetic variation both between and within plant species in their growth responses to the phytoavailability of Fe, Zn, Cu, and Mn, in their ability to acquire these mineral elements, and in their physiological utilisation of these elements to produce yield [19, 21, 55–58].

Sodium toxicity is thought to affect 5–15% of potential agricultural land [31]. Crop production on this land can be increased by management practices that reduce the concentration of Na^+ in the soil solution [59]. Traditionally, saline soils are remediated by leaching soluble salts from the soil profile by irrigation with fresh water, and sodic soils are remediated through the application of Ca^{2+}, often as gypsum, followed by flushing the soil with fresh water [59]. These management practices also remove Cl and B (depending on soil pH) from saline and sodic soils. These management strategies can be augmented by growing crops or varieties that have greater exclusion or tolerance of Na, Cl, or B. There is considerable genetic variation both between and within plant species for growth in soils with high Na, Cl and B concentrations that can be utilised for crop selection or breeding [31, 60–62]. In addition, knowledge of plant transport processes has allowed transgenic plants to be created that have greater yields on saline and sodic soils. For example, the overexpression of orthologues of HKT1 that retrieve of Na^+ from the xylem restricts shoot Na concentrations and confers Na tolerance to transgenic plants [31, 63], and increased expression of genes encoding transport proteins that catalyse B efflux from cells (BORs) increases tolerance to high B concentrations in the soil solution [62, 64].

2.3. Optimising Fertiliser Applications for Sustainable Intensification. In many agricultural soils, there is insufficient phytoavailable N, P, or K for the rapid growth of crop plants [3, 8, 65, 66]. To increase crop yields, these elements are, therefore, supplied as inorganic fertilisers, manures, composts, or miscellaneous "waste" materials including industrial biproducts, such as blood and bones, winery, brewery, and distillery residues, residues from sugar production, plasterboard, and paper crumble, and fly ash [8, 67–70]. To increase food production in the future, sustainable intensification will be required. High crop yields might be achieved and sustained through appropriate management of multiple sources of mineral input, both inorganic and organic, to remove nutritional constraints to crop production, supported by suitable amendments to address other soil constraints such as acidity or alkalinity [3, 67].

There are many agronomic strategies to improve efficiencies in the use of inorganic and organic fertilisers by crops. These include the use of (1) fertiliser recommendations informed by field response trials and based on soil or plant analyses [67, 71], (2) model-based decision support systems to inform fertiliser recommendations [72, 73], (3) fertiliser placement and other precision application technologies [66, 67, 73–75], (4) foliar fertilisation through insecticide and herbicide spraying programmes to allow fertiliser applications when crops are growing at maximal rates, and (5) crop residues, composts, or animal manures to improve soil quality [21, 67, 76, 77]. The introduction of legumes into rotations improves their N-economies and can increase crop yields in extensive, N-limited agricultural systems [67, 78].

These agronomic strategies can be complemented by the development of crop varieties that acquire and utilise fertilisers more efficiently to produce a commercial yield. The literature contains many definitions relating to the efficient use of fertilisers in agriculture [79]. The agronomic use efficiency of a mineral element (MUE) supplied in a fertiliser is generally defined as crop dry matter (DM) yield per unit of mineral element available (M_a) in the soil (g DM g^{-1} M_a). This is numerically equivalent to the product of the plant mineral content (M_p) per unit of available mineral element (g M_p g^{-1} M_a), which is often referred to as plant mineral uptake efficiency (MUpE), and the yield per unit plant mineral content (g DM g^{-1} M_p), which is often referred to as the mineral utilisation efficiency (MUtE) of the plant. There is considerable genetic variation, both between and within crop species, in all these measures for mineral elements supplied in fertilisers, including N, P, and K [21, 80–84].

Nitrogen utilisation efficiency (NUtE) often contributes more than N uptake efficiency (NUpE) to agronomic N use efficiency (NUE) when plants are grown with a low N supply [21, 85–87]. Historical improvements in NUtE are attributed to a greater partitioning of dry matter to the grain (i.e., increased harvest index), and NUtE is often positively correlated with yield. In crops, such as cereals and oilseed rape, that require continued N uptake by the root system following anthesis, NUpE also contributes significantly to NUE [87, 88].

In contrast, differences between genotypes in their yield responses to P fertilisation are often correlated with P uptake efficiency (PUpE) but not P utilisation efficiency (PUtE) within the plant [79, 82]. The trait of PUpE has been attributed to improved root architectures, particularly greater production of lateral roots, topsoil foraging characteristics, the production of root hairs, and the exudation of organic acids and phosphatases into the rhizosphere [65, 79, 82, 89, 90].

Chromosomal loci (QTL) influencing aspects of PUE have been reported in rice [91–96], wheat [97, 98], maize [99–101], bean [102–105], soybean [106–108], *Brassica rapa* [109, 110], *Brassica oleracea* [89], and *Brassica napus* [111, 112]. This genetic knowledge will accelerate breeding for PUE in crops.

Plant species vary considerably in their responses to K-fertiliser and in their abilities to acquire and utilise K for growth [21, 113, 114]. Although there is genetic variation in both K uptake efficiency (KUpE) and K utilisation efficiency (KUtE) within crop species [21, 81, 84, 113, 115], agronomic K use efficiency (KUE) is often correlated with KUpE and rarely with KUtE [84]. Greater KUpE has been attributed to: (1) increased exudation of compounds that release more nonexchangeable K^+ into the soil solution, (2) increased K^+ uptake capacity of root cells, which accelerates K^+ diffusion to the root surface, (3) proliferation of roots into the soil volume, which decreases the distance for K^+ diffusion to the root and increases the root surface area available for K^+ uptake, and (4) higher transpiration rates, which accelerates the mass flow of the soil solution to the root surface [114].

3. Biofortification of Edible Crops for Human Nutrition

In principle, two complementary strategies can be employed to increase mineral concentrations in edible crops [11, 12, 14, 15, 27, 116–119]. The first strategy, termed "agronomic" biofortification, employs the use of fertilisers containing the mineral elements lacking in human diets, principally Zn, Cu, Fe, I, Se, Mg, and Ca, in conjunction with (1) appropriate soil amendments, such as composts and manures to increase soil concentrations of essential elements, (2) acidifying fertilisers, such as urea, ammonium nitrate, ammonium sulphate, ammonium phosphates, or elemental S, to rectify soil alkalinity or lime to rectify soil acidity, and (3) appropriate crop rotations, intercropping, or the introduction of beneficial soil microorganisms to increase the phytoavailability of mineral elements [10, 14, 21, 55, 120]. Where mineral elements, such as Fe or Zn, become rapidly unavailable to roots, the use of foliar fertilisers, rather than soil fertilisers, is recommended [3, 10]. The application of N fertilisers, can be used to increase Zn concentrations in leaves and phloem-fed tissues [121–125]. The second strategy, termed "genetic" biofortification, employs crop genotypes with increased abilities to acquire mineral elements and accumulate them in edible tissues. There is sufficient natural genetic variation in the concentrations of mineral elements commonly lacking in human diets in the edible tissues of most crop species to breed for increased concentrations of mineral elements in edible tissues [14, 27, 118, 126] and also scope for targeted GM of crops [14, 125–127].

Agronomic strategies are most effective where appropriate infrastructures for the production, distribution, and application of inorganic fertilisers are available and are the only feasible strategies in regions where soils have insufficient concentrations of mineral elements required for human nutrition to support mineral-dense crops [12, 14, 20, 116]. Several authors have reviewed appropriate methods, infrastructural requirements, and practical benefits for food production, economic sustainability, and human health of agronomic biofortification of edible crops [12, 14, 20, 116]. Examples of the successful use of agronomic strategies include (1) the application of Se-fertilisers to increase dietary Se intakes in Finland, New Zealand, and elsewhere [25, 26, 128], (2) the iodinisation of irrigation water to increase dietary intakes of I in Xinjiang, China [23, 129], and (3) the use of compound fertilisers containing Zn to increase crop production, dietary Zn intakes, and human health in Anatolia, Turkey [20, 116]. Rational approaches to select areas that would benefit most from agronomic biofortification have also been developed [130].

Genetic strategies can be considered in regions where the total concentrations of mineral elements required for human nutrition are sufficient to support mineral-dense crops, but the accumulation of these elements is limited by their phytoavailability and acquisition by plant roots [14]. This strategy is particularly relevant in areas lacking the infrastructures required for fertiliser distribution [14, 15]. It is considered cost effective and beneficial to the 40% of the world's population who rely primarily on their own food for sustenance [14, 15]. It has been observed that there is sufficient genetic variation within germplasm collections of all major crops to breed varieties that accumulate greater concentrations of mineral elements in their edible portions [14, 15, 27, 118, 125]. Such breeding strategies can be facilitated by the development of molecular markers associated with the accumulation of essential mineral elements in edible portions of crop plants. Recent research has, therefore, been directed to the identification of chromosomal loci (QTL) associated with these traits (Table 2). For example, QTL affecting the accumulation of essential mineral elements commonly lacking in human diets in edible portions have been identified in rice [131–136], wheat [131–140], barley [141, 142], maize [143, 144], bean [145–152], soybean [153], brassicas [154–158], and potato [159]. This knowledge will facilitate conventional breeding of mineral-dense crops.

Strategies employing GM of crop plants are also being developed to increase the acquisition of mineral elements essential for human nutrition and their accumulation in edible tissues [14, 125, 160–162]. These strategies are primarily focussed on the biofortification of edible produce with Fe and Zn. In nongraminaceous plants, Fe uptake can be increased by overexpressing genes encoding Fe(III) reductases [163], and in graminaceous plants the acquisition of Fe and Zn can be increased by greater exudation of phytosiderophores [164]. The overexpression of genes encoding transporters catalysing Fe^{2+} or Zn^{2+} influx to root cells, sequestration in the vacuole, or delivery to the xylem have met with some success in the biofortification of roots and leaves of crop plants with Fe and Zn, but rarely in the biofortification of fruit, seeds, or tubers [14, 125, 127]. By contrast, the overexpression of genes encoding nicotianamine synthase (NAS) often leads to increased concentrations of Fe, Zn, and Mn both in leaves and in seeds [14, 125, 161]. In addition, targeted overexpression of genes encoding metal-binding proteins, such as ferritin and lactoferrin, have increased Fe, Zn, and Cu concentrations

Table 2: Studies in which chromosomal loci (QTL) have been identified in crop plants that affect the concentrations of essential mineral elements most commonly lacking in human diets.

Crop species	Tissue	Elements	References
Rice (*Oryza sativa*)	Grain	Fe	Gregorio et al. [131]
	Grain	Fe, Zn, Mn	Stangoulis et al. [132]
	Grain	Fe, Zn, Mn, Cu, Ca	Lu et al. [133]
	Grain	Fe, Zn, Mn, Cu, Ca, Mg	Garcia-Oliveira et al. [134]
	Grain	Fe, Zn, Mn, Cu, Mg, Se	Norton et al. [135]
	Grain	Zn	Zhang et al. [136]
Wheat (*Triticum* spp.)	Grain	Fe, Zn, Mn	Distelfeld et al. [137]
	Grain	Zn	Shi et al. [138]
	Grain	Fe, Zn	Genc et al. [139]
	Grain	Fe, Zn, Mn, Cu, Ca, Mg	Peleg et al. [140]
Barley (*Hordeum vulgare*)	Grain	Zn	Lonergan et al. [141]
	Grain	Zn	Sadeghzadeh et al. [142]
Maize (*Zea mays*)	Kernel	Fe	Lung'aho et al. [143]
	Kernel	Fe, Zn, Mg	Šimić et al. [144]
Bean (*Phaseolus vulgaris*)	Seed	Fe, Zn	Beebe et al. [145]
	Seed	Fe, Zn, Ca	Guzmán-Maldonado et al. [146]
	Seed	Zn	Cichy et al. [147]
	Seed	Fe, Zn, Ca	Gelin et al. [148]
	Seed	Fe, Zn	Blair et al. [149]
	Seed	Fe, Zn	Cichy et al. [150]
	Seed	Fe, Zn	Blair et al. [151]
	Seed	Fe, Zn	Blair et al. [152]
Soybean (*Glycine max*)	Seed	Ca	Zhang et al. [153]
Oilseed Rape (*Brassica napus*)	Seed	Fe, Zn, Mn, Cu, Ca, Mg	Ding et al. [154]
Brassica oleracea	Leaf	Ca, Mg	Broadley et al. [155]
	Leaf	Zn	Broadley et al. [156]
Brassica rapa	Leaf	Fe, Zn, Mn, Mg	Wu et al. [157]
	Leaf	Ca, Mg	Broadley et al. [158]
Potato (*Solanum tuberosum*)	Tuber	Fe, Zn, Cu, Ca, Mg	Subramanian [159]

in rice grain [160, 165, 166] and Fe concentrations in maize seeds [167], lettuce leaves [168], tomato fruits, and potato tubers [169]. In wheat, the expression of a functional NAC transcription factor (*NAM-B1*) increases grain Fe and Zn concentrations by accelerating senescence and increasing the remobilisation of these elements from leaves to developing grain [170]. Successful biofortification of edible produce with Ca has been achieved through the overexpression of genes encoding the vacuolar Ca^{2+}/H^{+}-antiporters AtCAX1 lacking its autoinhibitory domain (*sCAX1*), a modified AtCAX2 (*sCAX2*) or AtCAX4 in appropriate tissues [171–174].

4. Reducing the Entry of Toxic Elements to the Human Food Chain

Some natural soils can contain high concentrations of mineral elements that are potentially toxic to plants and animals [4]. For example, acid soils have excessive Al and Mn phytoavailability, serpentine soils can have excessive Ni, Co or Cr concentrations, and seleniferous soils contain excessive Se concentrations [10, 28, 33, 59]. Industrial activities have also contaminated agricultural soils with, for example, Pb, Cd, Ni, Zn, and Cu from the mining and refining of metal ores [10, 34, 59] and radioisotopes from intentional or accidental discharges [175, 176]. Other human activities, such as the burning of fossil fuels and various wastes, have also contributed to the atmospheric deposition of potentially toxic elements onto agricultural soils, and the application of Cu pesticides in agriculture has increased soil Cu concentrations [10, 34, 177, 178].

Soil amendments, including inorganic fertilisers, manures, sewage sludges, and urban wastes, can also contain high concentrations of potentially toxic mineral elements and radioisotopes [10, 34, 66–68, 178–181], and the recycling of agricultural and municipal wastes can also result in the accumulation of harmful, and persistent, organic compounds [68]. Some manufactured phosphate fertilisers can contain high concentrations of, in particular, Cd, Cr, Hg, Pb and radioisotopes of uranium (U), and radium (Ra), but

Table 3: Statutory maximum annual metal loading rates (kg ha^{-1} y^{-1}) over a ten-year period for agricultural soils in the United Kingdom [190] and the European Communities [191], statutory maximum cumulative metal loading rates (kg ha^{-1}) for agricultural soils in the United States of America [192], and critical soil concentrations (kg ha^{-1}) considered to be phytotoxic calculated assuming a soil bulk density of 1200 kg m^{-3} and a depth of 0.10 m [193].

Element	UK (kg ha^{-1} y^{-1})	EC (kg ha^{-1} y^{-1})	USA (kg ha^{-1})	Critical (kg ha^{-1})
Cd	0.15	0.15	39	6.0
Hg	0.1	0.1	17	3.7
Ni	3	3	420	120
Cu	7.5	12	1500	120
Cr	15	3	—	113
Pb	15	15	300	150
Zn	15	30	2800	390
Mo	0.2	—	—	8.1
Se	0.15	—	100	11
As	0.7	—	41	38

the concentrations of these elements vary widely depending upon the source of rock phosphate [2, 66, 67]. Animal manures and slurries can contain significant quantities of Cd, Cr, Pb, Co, Zn, Mn, Cu, and Mo [67, 178, 182]. Similarly, sewage sludges can contain high concentrations of Pb, Cd, Cr, Se, Co, Ni, Zn, Mn, and Cu, and also human pathogens [67, 178, 181, 183–186]. Composted municipal solid waste is frequently applied at high application rates (e.g., 200 Mg ha^{-1}), which can result in large amounts of Pb, Cd, Cr, As, Hg, Se, Co, Ni, Zn, Mn, Cu and Mo entering soils [67, 68, 185, 187]. Fortunately, the phytoavailability of many of these potentially toxic elements from municipal composts is relatively low [68, 185]. Industrial wastes such as food wastes, paper sludge, and fly ash can also contain significant amounts of potentially toxic elements [178, 188]. In many countries, legislation limits the quantities of heavy metals applied to soils on which edible crops are grown for human consumption (Table 3; [68, 178, 184, 187, 189–193]). It is important that these limits are followed to maintain both crop production and human health.

There are particular concerns about As concentrations in paddy rice, especially in South Asia in countries such as Bangladesh, India, and China [36]. Flooded paddy conditions lead to the mobilisation of arsenite, which is taken up efficiently by rice roots through the silicon transport pathway [36]. Growing rice for longer periods under aerobic soil conditions, by midseason draining of water or cultivation in raised soil beds, has been proposed as an effective way to reduce As uptake by rice, and Si-fertilisers can also be employed to restrict As uptake [36]. In addition to these agronomic strategies, varieties of rice are being identified that accumulate lower concentrations of As, and other potentially toxic elements, in grain and QTL associated with these traits are being identified for breeding safer crops [137, 194, 195]. Similarly, genotypes of other crops that accumulate lower concentrations of potentially toxic mineral elements in their edible portions are being developed through conventional breeding and GM approaches [10, 36, 126, 196].

The continued replenishment of mineral elements in the soil is essential to maintain future food production. Sustainable sources of mineral elements must be sought through recycling through the food chain. Crop residues, animal manures, sewage sludges, municipal composts, and industrial wastes can all contribute to the delivery of the mineral elements required for plant growth. However, their use can also increase inputs of potentially toxic elements and organic pollutants to agricultural soils. Legal limits to their use must be followed to prevent toxicities to plants and animals, and it is generally recommended that they are used in combination with inorganic fertilisers through integrated nutrient management to avoid threats to human health and the wider environment [67]. In particular, animal manures can contribute significantly to the input of potentially toxic elements to agricultural soils [68, 186]. To reduce the entry of potentially toxic elements to the human food chain from this source, feed regimes can be adopted that result in lower concentrations of such elements in animal manures. When municipal composts are applied to agricultural land, these should conform to good quality criteria [67, 68, 185]. The concentrations of potentially toxic elements in some sewage sludges can be unacceptably high [184, 186]. Thus, controls on discharges to sewers and treatment of sewage effluents to remove potentially toxic elements should be actioned [183]. Furthermore, it is not recommended that municipal composts are mixed with sewage sludge, since this practice can increase the phytoavailability of potentially toxic elements [68]. Finally, phytoextraction strategies can be employed to remediate contaminated land, and the plant material generated might be used as biofuels [32, 126, 197].

5. Conclusion

This paper has described how managing plant mineral nutrition might contribute to future food security. It has highlighted roles for both agronomy and plant breeding in delivering sufficient, safe, and nutritious food to meet the dietary needs of an increasing human population. It has noted that the problems of mineral deficiencies and toxicities must be addressed to maximise crop production in both intensive and extensive agricultural systems. The chemical constraints to crop production on alkaline, acid, saline, and sodic soils can be addressed through agronomy or the development of tolerant genotypes. In intensive agricultural systems it is likely that inorganic fertilisers will continue to be required to maintain yields. However, their use might be reduced by agronomic strategies that improve fertiliser use efficiencies, by replacement with organic fertilisers, and by judicious choice of genotypes that acquire and utilise mineral elements better in producing commercial yields. In extensive agricultural systems integrated fertiliser management strategies using biological N_2 fixation, nonacidifying inorganic fertilisers, and organic fertilisers and amendments to develop soil fertility can be usefully adopted. To increase the dietary delivery of mineral elements essential to human wellbeing, agronomic strategies to increase the phytoavailability of these

elements combined with the cultivation of crops that acquire and accumulate greater concentrations of these elements in their edible tissues can be pursued where there is sufficiency of these elements present in the soil to support mineral-dense crops. However, where these essential elements are not present in the soil, the application of fertilisers containing these elements is required to increase their amounts in human diets, if diets remain unchanged. To reduce the entry of toxic elements into the human food chain, strict quality requirements for inorganic and organic fertilisers might be enforced, agronomic strategies could be used to reduce the phytoavailability of these elements, and crop genotypes can be developed that do not accumulate toxic concentrations of mineral elements in their edible tissues. Thus, ongoing interdisciplinary research in plant mineral nutrition, soil science, agronomy, and crop breeding is required for future food security to improve soil quality, optimise fertiliser applications for sustainable crop production, and develop strategies for the biofortification of edible crops with essential mineral elements to address mineral malnutrition in humans and other animals.

Acknowledgment

This paper is based on a talk given by P. J. White at the European Geosciences Union General Assembly in April 2011. The work was supported by the Rural and Environment Science and Analytical Services Division (RESAS) of the Scottish Government through Work package 7.2 (2011–2016).

References

[1] Food and Agriculture Organization of the United Nations [FAO], *The Strategic Framework for FAO: 2000–2015—A Summary*, FAO, Rome, Italy, 2000.

[2] M. Lægreid, O. C. Bøckman, and O. Kaarstad, *Agriculture, Fertilizers and the Environment*, CABI Publishing, Wallingford, UK, 1999.

[3] N. K. Fageria, V. C. Baligar, and C. A. Jones, *Growth and Mineral Nutrition of Field Crops*, CRC Press, Boca Raton, Fla, USA, 3rd edition, 2011.

[4] P. J. White and P. H. Brown, "Plant nutrition for sustainable development and global health," *Annals of Botany*, vol. 105, no. 7, pp. 1073–1080, 2010.

[5] D. I. Gregory and B. L. Bump, "Agriculture and rural development discussion paper 24. Factors affecting supply of fertilizer in Sub-Saharan Africa," The World Bank, Washington, DC, USA, 2005.

[6] C. Nellemann, M. MacDevette, T. Manders et al., "The environmental food crisis. The environment's role in averting future food crises. A UNEP rapid response assessment," Birkeland Trykkeri AS, Norway, 2009.

[7] P. M. Vitousek, R. Naylor, T. Crews et al., "Nutrient imbalances in agricultural development," *Science*, vol. 324, no. 5934, pp. 1519–1520, 2009.

[8] C. J. Dawson and J. Hilton, "Fertiliser availability in a resource-limited world: production and recycling of nitrogen and phosphorus," *Food Policy*, vol. 36, no. 1, supplement, pp. S14–S22, 2011.

[9] P. J. White, "Ion uptake mechanisms of individual cells and roots: short-distance transport," in *Marschner's Mineral Nutrition of Higher Plants*, P. Marschner, Ed., pp. 7–47, Academic Press, London, UK, 3rd edition, 2012.

[10] P. J. White and D. J. Greenwood, "Properties and management of cationic elements for crop growth," in *Russell's Soil Conditions and Plant Growth*, P. J. Gregory and S. Nortcliff, Eds., Wiley-Blackwell, Oxford, UK, 12th edition, 2012.

[11] P. J. White and M. R. Broadley, "Biofortifying crops with essential mineral elements," *Trends in Plant Science*, vol. 10, no. 12, pp. 586–593, 2005.

[12] R. D. Graham, R. M. Welch, D. A. Saunders et al., "Nutritious Subsistence Food Systems," *Advances in Agronomy*, vol. 92, pp. 1–74, 2007.

[13] A. J. Stein, "Global impacts of human mineral malnutrition," *Plant and Soil*, vol. 335, no. 1, pp. 133–154, 2010.

[14] P. J. White and M. R. Broadley, "Biofortification of crops with seven mineral elements often lacking in human diets—iron, zinc, copper, calcium, magnesium, selenium and iodine," *New Phytologist*, vol. 182, no. 1, pp. 49–84, 2009.

[15] H. E. Bouis and R. M. Welch, "Biofortification—a sustainable agricultural strategy for reducing micronutrient malnutrition in the Global South," *Crop Science*, vol. 50, pp. S20–S32, 2010.

[16] A. J. Karley and P. J. White, "Moving cationic minerals to edible tissues: potassium, magnesium, calcium," *Current Opinion in Plant Biology*, vol. 12, no. 3, pp. 291–298, 2009.

[17] E. Frossard, M. Bucher, F. Mächler, A. Mozafar, and R. Hurrell, "Potential for increasing the content and bioavailability of Fe, Zn and Ca in plants for human nutrition," *Journal of the Science of Food and Agriculture*, vol. 80, no. 7, pp. 861–879, 2000.

[18] Z. Rengel, "Genotypic differences in micronutrient use efficiency in crops," *Communications in Soil Science and Plant Analysis*, vol. 32, no. 7-8, pp. 1163–1186, 2001.

[19] M. R. Broadley, P. J. White, J. P. Hammond, I. Zelko, and A. Lux, "Zinc in plants: tansley review," *New Phytologist*, vol. 173, no. 4, pp. 677–702, 2007.

[20] I. Cakmak, "Enrichment of fertilizers with zinc: an excellent investment for humanity and crop production in India," *Journal of Trace Elements in Medicine and Biology*, vol. 23, no. 4, pp. 281–289, 2009.

[21] N. K. Fageria, *The Use of Nutrients in Crop Plants*, CRC Press, Boca Raton, Fla, USA, 2009.

[22] S. R. Wilkinson, R. M. Welch, H. F. Mayland, and D. L. Grunes, "Magnesium in plants: uptake, distribution, function, and utilization by man and animals," *Metal Ions in Biological Systems*, vol. 26, pp. 33–56, 1990.

[23] G. H. Lyons, J. C. R. Stangoulis, and R. D. Graham, "Exploiting micronutrient interaction to optimize biofortification programs: the case for inclusion of selenium and iodine in the HarvestPlus program," *Nutrition Reviews*, vol. 62, no. 6, pp. 247–252, 2004.

[24] J. F. Risher and L. S. Keith, *Iodine and Inorganic Iodides: Human Health Aspects*, WHO Press, Geneva, Switzerland, 2009.

[25] H. Hartikainen, "Biogeochemistry of selenium and its impact on food chain quality and human health," *Journal of Trace Elements in Medicine and Biology*, vol. 18, no. 4, pp. 309–318, 2005.

[26] M. R. Broadley, P. J. White, R. J. Bryson et al., "Biofortification of UK food crops with selenium," *Proceedings of the Nutrition Society*, vol. 65, no. 2, pp. 169–181, 2006.

[27] I. Cakmak, "Enrichment of cereal grains with zinc: agronomic or genetic biofortification?" *Plant and Soil*, vol. 302, no. 1-2, pp. 1–17, 2008.

[28] E. George, W. J. Horst, and E. Neumann, "Adaptation of plants to adverse chemical soil conditions," in *Marschner's Mineral Nutrition of Higher Plants*, P. Marschner, Ed., pp. 409–472, Academic Press, London, UK, 3rd edition, 2012.

[29] H. R. von Uexküll and E. Mutert, "Global extent, development and economic impact of acid soils," *Plant and Soil*, vol. 171, no. 1, pp. 1–15, 1995.

[30] M. E. Sumner and A. D. Noble, "Soil acidification: the world story," in *Handbook of Soil Acidity*, Z. Rengel, Ed., pp. 1–28, Marcel Dekker, New York, NY, USA, 2003.

[31] R. Munns and M. Tester, "Mechanisms of salinity tolerance," *Annual Review of Plant Biology*, vol. 59, pp. 651–681, 2008.

[32] S. N. Whiting, R. D. Reeves, D. G. Richards et al., "Use of plants to manage sites contaminated with metals," in *Plant Nutritional Genomics*, M. R. Broadley and P. J. White, Eds., pp. 287–315, Blackwell Publishing, Oxford, UK, 2005.

[33] P. J. White, M. R. Broadley, H. C. Bowen, and S. E. Johnson, "Selenium and its relationship with sufur," in *Sulfur in Plants—An Ecological Perspective*, M. J. Hawkesford and L. J. de Kok, Eds., pp. 225–252, Springer, Dordrecht, The Netherlands, 2007.

[34] Z. L. He, X. E. Yang, and P. J. Stoffella, "Trace elements in agroecosystems and impacts on the environment," *Journal of Trace Elements in Medicine and Biology*, vol. 19, no. 2-3, pp. 125–140, 2005.

[35] P. Higueras, R. Oyarzun, J. Lillo et al., "The Almadén district (Spain): anatomy of one of the world's largest Hg-contaminated sites," *Science of the Total Environment*, vol. 356, no. 1-3, pp. 112–124, 2006.

[36] F. J. Zhao, S. P. McGrath, and A. A. Meharg, "Arsenic as a food chain contaminant: mechanisms of plant uptake and metabolism and mitigation strategies," *Annual Review of Plant Biology*, vol. 61, pp. 535–559, 2010.

[37] L. T. Evans, "Adapting and improving crops: the endless task," *Philosophical Transactions of the Royal Society B*, vol. 352, no. 1356, pp. 901–906, 1997.

[38] H. C. J. Godfray, J. R. Beddington, I. R. Crute et al., "Food security: the challenge of feeding 9 billion people," *Science*, vol. 327, no. 5967, pp. 812–818, 2010.

[39] N. Koning and M. K. van Ittersum, "Will the world have enough to eat?" *Current Opinion in Environmental Sustainability*, vol. 1, no. 1, pp. 77–82, 2009.

[40] P. J. Gregory and T. S. George, "Feeding nine billion: the challenge to sustainable crop production," *Journal of Experimental Botany*, vol. 62, no. 15, pp. 5233–5239, 2011.

[41] J. Bruinsma, "The resource outlook to 2050: by how much do land, water, and crop yields need to increase by 2050?" in *Proceedings of the FAO Expert Meeting on 'How to Feed the World in 2050*, FAO, Rome, Italy, June 2009.

[42] J. Bruinsma, *World Agriculture: Towards 2015/2030. An FAO Perspective*, Earthscan Publications, London, UK, 2003.

[43] Food and Agriculture Organization of the United Nations [FAO], *The State of Food and Agriculture 2008. Biofuels: Prospects, Risks and Opportunities*, FAO, Rome, Italy, 2008.

[44] P. Pingali, T. Raney, and K. Wiebe, "Biofuels and food security: missing the point," *Review of Agricultural Economics*, vol. 30, no. 3, pp. 506–516, 2008.

[45] P. Smith, P. J. Gregory, D. Van Vuuren et al., "Competition for land," *Philosophical Transactions of the Royal Society B*, vol. 365, no. 1554, pp. 2941–2957, 2010.

[46] J. W. Erisman, M. A. Sutton, J. Galloway, Z. Klimont, and W. Winiwarter, "How a century of ammonia synthesis changed the world," *Nature Geoscience*, vol. 1, no. 10, pp. 636–639, 2008.

[47] D. B. Lobell, K. G. Cassman, and C. B. Field, "Crop yield gaps: their importance, magnitudes, and causes," *Annual Review of Environment and Resources*, vol. 34, pp. 179–204, 2009.

[48] K. Neumann, P. H. Verburg, E. Stehfest, and C. Müller, "The yield gap of global grain production: a spatial analysis," *Agricultural Systems*, vol. 103, no. 5, pp. 316–326, 2010.

[49] The World Bank, *World Development Report 2008. Agriculture for Development*, The World Bank, Washington DC, USA, 2007.

[50] K. W. Jaggard, A. Qi, and S. Ober, "Possible changes to arable crop yields by 2050," *Philosophical Transactions of the Royal Society B*, vol. 365, no. 1554, pp. 2835–2851, 2010.

[51] E. C. Oerke, "Crop losses to pests," *Journal of Agricultural Science*, vol. 144, no. 1, pp. 31–43, 2006.

[52] J. F. Ma, P. R. Ryan, and E. Delhaize, "Aluminium tolerance in plants and the complexing role of organic acids," *Trends in Plant Science*, vol. 6, no. 6, pp. 273–278, 2001.

[53] L. V. Kochian, O. A. Hoekenga, and M. A. Piñeros, "How do crop plants tolerate acid soils? Mechanisms of aluminum tolerance and phosphorous efficiency," *Annual Review of Plant Biology*, vol. 55, pp. 459–493, 2004.

[54] E. Delhaize, B. D. Gruber, and P. R. Ryan, "The roles of organic anion permeases in aluminium resistance and mineral nutrition," *FEBS Letters*, vol. 581, no. 12, pp. 2255–2262, 2007.

[55] R. D. Graham, "Genotype differences in tolerance to manganese deficiency," in *Manganese in Soils and Plants*, R. D. Graham, R. J. Hannam, and N. C. Uren, Eds., pp. 261–276, Kluwer Academic Publishers, Dordrecht, The Netherlands, 1988.

[56] G. Hacisalihoglu and L. V. Kochian, "How do some plants tolerate low levels of soil zinc? Mechanisms of zinc efficiency in crop plants," *New Phytologist*, vol. 159, no. 2, pp. 341–350, 2003.

[57] V. D. Jolley, K. A. Cook, N. C. Hansen, and W. B. Stevens, "Plant physiological responses for genotypic evaluation of iron efficiency in strategy I and strategy II plants—a review," *Journal of Plant Nutrition*, vol. 19, no. 8-9, pp. 1241–1255, 1996.

[58] I. Evans, E. Solberg, and D. M. Huber, "Copper and plant disease," in *Mineral Nutrition and Plant Disease*, L. E. Datnoff, W. H. Elmer, and D. M. Huber, Eds., pp. 177–188, The American Phytopathological Society, St Paul, Minn, USA, 2007.

[59] M. Hodson, "Managing adverse soil chemical environments," in *Russell's Soil Conditions and Plant Growth*, P. J. Gregory and S. Nortcliff, Eds., Wiley-Blackwell, Oxford, UK, 12th edition, 2012.

[60] P. J. White and M. R. Broadley, "Chloride in soils and its uptake and movement within the plant: a review," *Annals of Botany*, vol. 88, no. 6, pp. 967–988, 2001.

[61] D. G. Masters, S. E. Benes, and H. C. Norman, "Biosaline agriculture for forage and livestock production," *Agriculture, Ecosystems and Environment*, vol. 119, no. 3-4, pp. 234–248, 2007.

[62] R. Reid, "Can we really increase yields by making crop plants tolerant to boron toxicity?" *Plant Science*, vol. 178, no. 1, pp. 9–11, 2010.

[63] S. J. Roy, E. J. Tucker, and M. Tester, "Genetic analysis of abiotic stress tolerance in crops," *Current Opinion in Plant Biology*, vol. 14, no. 3, pp. 232–239, 2011.

[64] K. Miwa and T. Fujiwara, "Boron transport in plants: co-ordinated regulation of transporters," *Annals of Botany*, vol. 105, no. 7, pp. 1103–1108, 2010.

[65] J. P. Lynch, "Roots of the second green revolution," *Australian Journal of Botany*, vol. 55, no. 5, pp. 493–512, 2007.

[66] E. A. Kirkby and A. E. Johnson, "Soil and fertilizer phosphorus in relation to crop nutrition," in *The Ecophysiology of Plant-Phosphorus Interactions*, P. J. White and J. P. Hammond, Eds., pp. 177–223, Springer, Dordrecht, The Netherlands, 2008.

[67] R. N. Roy, A. Finck, G. J. Blair, and H. L. S. Tandon, *FAO Fertilizer and Plant Nutrition Bulletin 16. Plant Nutrition for Food Security. A Guide for Integrated Nutrient Management*, FAO, Rome, Italy, 2006.

[68] J. C. Hargreaves, M. S. Adl, and P. R. Warman, "A review of the use of composted municipal solid waste in agriculture," *Agriculture, Ecosystems and Environment*, vol. 123, no. 1-3, pp. 1–14, 2008.

[69] Department for Environment, Food and Rural Affairs [Defra], *Fertiliser Manual (RB209)*, The Stationery Office, London, UK, 8th edition, 2010.

[70] F. J. García Navarro, J. A. Amorós ortiz-Villajos, C. J. Sánchez Jiménez, S. B. Martín-Consuegra, E. M. Cubero, and R. J. Ballesta, "Application of sugar foam to red soils in a semiarid Mediterranean environment," *Environmental Earth Sciences*, vol. 59, no. 3, pp. 603–611, 2009.

[71] J. P. Hammond and P. J. White, "Diagnosing phosphorus deficiency in crop plants," in *The Ecophysiology of Plant-Phosphorus Interactions*, P. J. White and J. P. Hammond, Eds., pp. 225–246, Springer, Dordrecht, The Netherlands, 2008.

[72] K. Zhang, D. J. Greenwood, P. J. White, and I. G. Burns, "A dynamic model for the combined effects of N, P and K fertilizers on yield and mineral composition; description and experimental test," *Plant and Soil*, vol. 298, no. 1-2, pp. 81–98, 2007.

[73] R. Gebbers and V. I. Adamchuk, "Precision agriculture and food security," *Science*, vol. 327, no. 5967, pp. 828–831, 2010.

[74] R. J. Bryson, *Proceedings of The International Fertiliser Society 577. Improvement in Farm and Nutrient Management through Precision Farming*, IFS, York, UK, 2005.

[75] I. G. Burns, J. P. Hammond, and P. J. White, "Precision placement of fertiliser for optimising the early nutrition of vegetable crops—a review of the implications for the yield and quality of crops, and their nutrient use efficiency," *Acta Horticulturae*, vol. 852, pp. 177–188, 2010.

[76] M. Herrero, P. K. Thornton, A. M. Notenbaert et al., "Smart investments in sustainable food production: revisiting mixed crop-livestock systems," *Science*, vol. 327, no. 5967, pp. 822–825, 2010.

[77] P. Hallett and A. G. Bengough, "Managing the soil physical environment for plants," in *Russell's Soil Conditions and Plant Growth*, P. J. Gregory and S. Nortcliff, Eds., Wiley-Blackwell, Oxford, UK, 12th edition, 2012.

[78] G. Hardarson and W. J. Broughton, *Maximising the Use of Biological Nitrogen Fixation in Agriculture*, Kluwer Academic Publishers, Dordrecht, The Netherlands, 2003.

[79] P. J. White, M. R. Broadley, D. J. Greenwood, and J. P. Hammond, *Proceedings of The International Fertiliser Society 568. Genetic Modifications to Improve Phosphorus Acquisition by Roots*, IFS, York, UK, 2005.

[80] B. Hirel, J. Le Gouis, B. Ney, and A. Gallais, "The challenge of improving nitrogen use efficiency in crop plants: towards a more central role for genetic variability and quantitative genetics within integrated approaches," *Journal of Experimental Botany*, vol. 58, no. 9, pp. 2369–2387, 2007.

[81] Z. Rengel and P. M. Damon, "Crops and genotypes differ in efficiency of potassium uptake and use," *Physiologia Plantarum*, vol. 133, no. 4, pp. 624–636, 2008.

[82] P. J. White and J. P. Hammond, "Phosphorus nutrition of terrestrial plants," in *The Ecophysiology of Plant—Phosphorus Interactions*, P. J. White and J. P. Hammond, Eds., pp. 51–81, Springer, Dordrecht, The Netherlands, 2008.

[83] R. Sylvester-Bradley and D. R. Kindred, "Analysing nitrogen responses of cereals to prioritize routes to the improvement of nitrogen use efficiency," *Journal of Experimental Botany*, vol. 60, no. 7, pp. 1939–1951, 2009.

[84] P. J. White, J. P. Hammond, G. J. King et al., "Genetic analysis of potassium use efficiency in *Brassica oleracea*," *Annals of Botany*, vol. 105, no. 7, pp. 1199–1210, 2010.

[85] P. B. Barraclough, J. R. Howarth, J. Jones et al., "Nitrogen efficiency of wheat: genotypic and environmental variation and prospects for improvement," *European Journal of Agronomy*, vol. 33, no. 1, pp. 1–11, 2010.

[86] P. H. Beatty, Y. Anbessa, P. Juskiw, R. T. Carroll, J. Wang, and A. G. Good, "Nitrogen use efficiencies of spring barley grown under varying nitrogen conditions in the field and growth chamber," *Annals of Botany*, vol. 105, no. 7, pp. 1171–1182, 2010.

[87] I. J. Bingham, A. J. Karley, P. J. White, W. T. B. Thomas, and J. R. Russell, "Analysis of improvements in nitrogen use efficiency associated with 75 years of barley breeding," *European Journal of Agronomy*. In press.

[88] P. M. Berry, J. Spink, M. J. Foulkes, and P. J. White, "The physiological basis of genotypic differences in nitrogen use efficiency in oilseed rape (*Brassica napus* L.)," *Field Crops Research*, vol. 119, no. 2-3, pp. 365–373, 2010.

[89] J. P. Hammond, M. R. Broadley, P. J. White et al., "Shoot yield drives phosphorus use efficiency in *Brassica oleracea* and correlates with root architecture traits," *Journal of Experimental Botany*, vol. 60, no. 7, pp. 1953–1968, 2009.

[90] A. Fita, F. Nuez, and B. Picó, "Diversity in root architecture and response to P deficiency in seedlings of *Cucumis melo* L," *Euphytica*, vol. 181, no. 3, pp. 323–339, 2011.

[91] J. J. Ni, P. Wu, D. Senadhira, and N. Huang, "Mapping QTLs for phosphorus deficiency tolerance in rice (*Oryza sativa* L.)," *Theoretical and Applied Genetics*, vol. 97, no. 8, pp. 1361–1369, 1998.

[92] F. Ming, X. Zheng, G. Mi, L. Zhu, and F. Zhang, "Detection and verification of quantitative trait loci affecting tolerance to low phosphorus in rice," *Journal of Plant Nutrition*, vol. 24, no. 9, pp. 1399–1408, 2001.

[93] P. Mu, C. Huang, J. X. Li, L. F. Liu, and Z. C. Li, "Yield trait variation and QTL mapping in a DH population of rice under phosphorus deficiency," *Acta Agronomica Sinica*, vol. 34, no. 7, pp. 1137–1142, 2008.

[94] S. Heuer, X. Lu, J. H. Chin et al., "Comparative sequence analyses of the major quantitative trait locus *phosphorus uptake 1 (Pup1)* reveal a complex genetic structure," *Plant Biotechnology Journal*, vol. 7, no. 5, pp. 456–471, 2009.

[95] J. Li, Y. Xie, A. Dai, L. Liu, and Z. Li, "Root and shoot traits responses to phosphorus deficiency and QTL analysis at seedling stage using introgression lines of rice," *Journal of Genetics and Genomics*, vol. 36, no. 3, pp. 173–183, 2009.

[96] J. H. Chin, R. Gamuyao, C. Dalid et al., "Developing rice with high yield under phosphorus deficiency: Pup1 sequence to application," *Plant Physiology*, vol. 156, no. 3, pp. 1202–1216, 2011.

[97] J. Su, Y. Xiao, M. Li et al., "Mapping QTLs for phosphorus-deficiency tolerance at wheat seedling stage," *Plant and Soil*, vol. 281, no. 1-2, pp. 25–36, 2006.

[98] J. Y. Su, Q. Zheng, H. W. Li et al., "Detection of QTLs for phosphorus use efficiency in relation to agronomic performance of wheat grown under phosphorus sufficient

and limited conditions," *Plant Science*, vol. 176, no. 6, pp. 824–836, 2009.

[99] J. Chen, L. Xu, Y. Cai, and J. Xu, "QTL mapping of phosphorus efficiency and relative biologic characteristics in maize (*Zea mays* L.) at two sites," *Plant and Soil*, vol. 313, no. 1-2, pp. 251–266, 2008.

[100] J. Chen, L. Xu, Y. Cai, and J. Xu, "Identification of QTLs for phosphorus utilization efficiency in maize (*Zea mays* L.) across P levels," *Euphytica*, vol. 167, no. 2, pp. 245–252, 2009.

[101] M. Li, X. Guo, M. Zhang et al., "Mapping QTLs for grain yield and yield components under high and low phosphorus treatments in maize (*Zea mays* L.)," *Plant Science*, vol. 178, no. 5, pp. 454–462, 2010.

[102] H. Liao, X. Yan, G. Rubio, S. E. Beebe, M. W. Blair, and J. P. Lynch, "Genetic mapping of basal root gravitropism and phosphorus acquisition efficiency in common bean," *Functional Plant Biology*, vol. 31, no. 10, pp. 959–970, 2004.

[103] S. E. Beebe, M. Rojas-Pierce, X. Yan et al., "Quantitative trait loci for root architecture traits correlated with phosphorus acquisition in common bean," *Crop Science*, vol. 46, no. 1, pp. 413–423, 2006.

[104] I. E. Ochoa, M. W. Blair, and J. P. Lynch, "QTL analysis of adventitious root formation in common bean under contrasting phosphorus availability," *Crop Science*, vol. 46, no. 4, pp. 1609–1621, 2006.

[105] K. A. Cichy, M. W. Blair, C. H. Galeano Mendoza, S. S. Snapp, and J. D. Kelly, "QTL analysis of root architecture traits and low phosphorus tolerance in an andean bean population," *Crop Science*, vol. 49, no. 1, pp. 59–68, 2009.

[106] Y. D. Li, Y. J. Wang, Y. P. Tong, J. G. Gao, J. S. Zhang, and S. Y. Chen, "QTL mapping of phosphorus deficiency tolerance in soybean (*Glycine max* L. Merr.)," *Euphytica*, vol. 142, no. 1-2, pp. 137–142, 2005.

[107] D. Zhang, H. Cheng, L. Geng et al., "Detection of quantitative trait loci for phosphorus deficiency tolerance at soybean seedling stage," *Euphytica*, vol. 167, no. 3, pp. 313–322, 2009.

[108] Q. Liang, X. Cheng, M. Mei, X. Yan, and H. Liao, "QTL analysis of root traits as related to phosphorus efficiency in soybean," *Annals of Botany*, vol. 106, no. 1, pp. 223–234, 2010.

[109] J. Zhao, D. C. L. Jamar, P. Lou et al., "Quantitative trait loci analysis of phytate and phosphate concentrations in seeds and leaves of *Brassica rapa*," *Plant, Cell and Environment*, vol. 31, no. 7, pp. 887–900, 2008.

[110] J. P. Hammond, S. Mayes, H. C. Bowen et al., "Regulatory hotspots are associated with plant gene expression under varying soil phosphorus supply in *Brassica rapa*," *Plant Physiology*, vol. 156, no. 3, pp. 1230–1241, 2011.

[111] M. Yang, G. Ding, L. Shi, J. Feng, F. Xu, and J. Meng, "Quantitative trait loci for root morphology in response to low phosphorus stress in *Brassica napus*," *Theoretical and Applied Genetics*, vol. 121, no. 1, pp. 181–193, 2010.

[112] M. Yang, G. Ding, L. Shi, F. Xu, and J. Meng, "Detection of QTL for phosphorus efficiency at vegetative stage in *Brassica napus*," *Plant and Soil*, vol. 339, no. 1, pp. 97–111, 2011.

[113] S. P. Trehan, "Nutrient management by exploiting genetic diversity of potato—a review," *Potato Journal*, vol. 32, pp. 1–15, 2005.

[114] P. J. White and A. J. Karley, "Potassium," in *Plant Cell Monographs 17, Cell Biology of Metals and Nutrients*, R. Hell and R.-R. Mendel, Eds., pp. 199–224, Springer, Dordrecht, The Netherlands, 2010.

[115] V. C. Baligar, N. K. Fageria, and Z. L. He, "Nutrient use efficiency in plants," *Communications in Soil Science and Plant Analysis*, vol. 32, no. 7-8, pp. 921–950, 2001.

[116] I. Cakmak, *Proceedings of the International Fertiliser Society 552. Identification and Correction of Widespread Zinc Deficiency in Turkey—A Success Story*, IFS, York, UK, 2004.

[117] T. Johns and P. B. Eyzaguirre, "Biofortification, biodiversity and diet: a search for complementary applications against poverty and malnutrition," *Food Policy*, vol. 32, no. 1, pp. 1–24, 2007.

[118] W. H. Pfeiffer and B. McClafferty, "HarvestPlus: breeding crops for better nutrition," *Crop Science*, vol. 47, pp. S88–S105, 2007.

[119] A. H. Khoshgoftarmanesh, R. Schulin, R. L. Chaney, B. Daneshbakhsh, and M. Afyuni, "Micronutrient-efficient genotypes for crop yield and nutritional quality in sustainable agriculture. A review," *Agronomy for Sustainable Development*, vol. 30, no. 1, pp. 83–107, 2010.

[120] X. He and K. Nara, "Element biofortification: can mycorrhizas potentially offer a more effective and sustainable pathway to curb human malnutrition?" *Trends in Plant Science*, vol. 12, no. 8, pp. 331–333, 2007.

[121] H. L. Hao, Y. Z. Wei, X. E. Yang, Y. Feng, and C. Y. Wu, "Effects of different nitrogen fertilizer levels on Fe, Mn, Cu and Zn concentrations in shoot and grain quality in rice (*Oryza sativa*)," *Rice Science*, vol. 14, no. 4, pp. 289–294, 2007.

[122] R. Shi, Y. Zhang, X. Chen et al., "Influence of long-term nitrogen fertilization on micronutrient density in grain of winter wheat (*Triticum aestivum* L.)," *Journal of Cereal Science*, vol. 51, no. 1, pp. 165–170, 2010.

[123] U. B. Kutman, B. Yildiz, and I. Cakmak, "Improved nitrogen status enhances zinc and iron concentrations both in the whole grain and the endosperm fraction of wheat," *Journal of Cereal Science*, vol. 53, pp. 118–125, 2011.

[124] U. B. Kutman, B. Yildiz, and I. Cakmak, "Effect of nitrogen on uptake, remobilization and partitioning of zinc and iron throughout the development of durum wheat," *Plant and Soil*, vol. 342, no. 1-2, pp. 149–164, 2011.

[125] P. J. White and M. R. Broadley, "Physiological limits to zinc biofortification of edible crops," *Frontiers in Plant Nutrition*, vol. 2, article 80, 2011.

[126] F. J. Zhao and S. P. McGrath, "Biofortification and phytoremediation," *Current Opinion in Plant Biology*, vol. 12, no. 3, pp. 373–380, 2009.

[127] B. M. Waters and R. P. Sankaran, "Moving micronutrients from the soil to the seeds: genes and physiological processes from a biofortification perspective," *Plant Science*, vol. 180, no. 4, pp. 562–574, 2011.

[128] P. Ekholm, H. Reinivuo, P. Mattila et al., "Changes in the mineral and trace element contents of cereals, fruits and vegetables in Finland," *Journal of Food Composition and Analysis*, vol. 20, no. 6, pp. 487–495, 2007.

[129] X. M. Jiang, X. Y. Cao, J. Y. Jiang et al., "Dynamics of environmental supplementation of iodine: four years' experience of iodination of irrigation water in Hotien, Xinjiang, China," *Archives of Environmental Health*, vol. 52, no. 6, pp. 399–408, 1997.

[130] E. Zapata-Caldas, G. Hyman, H. Pachón, F. A. Monserrate, and L. V. Varela, "Identifying candidate sites for crop biofortification in Latin America: case studies in Colombia, Nicaragua and Bolivia," *International Journal of Health Geographics*, vol. 8, no. 1, article 29, 2009.

[131] G. B. Gregorio, D. Senadhira, H. Htut, and R. D. Graham, "Breeding for trace mineral density in rice," *Food and Nutrition Bulletin*, vol. 21, no. 4, pp. 382–386, 2000.

[132] J. C. R. Stangoulis, B. L. Huynh, R. M. Welch, E. Y. Choi, and R. D. Graham, "Quantitative trait loci for phytate in

rice grain and their relationship with grain micronutrient content," *Euphytica*, vol. 154, no. 3, pp. 289–294, 2007.

[133] K. Lu, L. Li, X. Zheng, Z. Zhang, T. Mou, and Z. Hu, "Quantitative trait loci controlling Cu, Ca, Zn, Mn and Fe content in rice grains," *Journal of Genetics*, vol. 87, no. 3, pp. 305–310, 2008.

[134] A. L. Garcia-Oliveira, L. Tan, Y. Fu, and C. Sun, "Genetic identification of quantitative trait loci for contents of mineral nutrients in rice grain," *Journal of Integrative Plant Biology*, vol. 51, no. 1, pp. 84–92, 2009.

[135] G. J. Norton, C. M. Deacon, L. Xiong, S. Huang, A. A. Meharg, and A. H. Price, "Genetic mapping of the rice ionome in leaves and grain: identification of QTLs for 17 elements including arsenic, cadmium, iron and selenium," *Plant and Soil*, vol. 329, no. 1, pp. 139–153, 2010.

[136] X. Zhang, G. Zhang, L. Guo et al., "Identification of quantitative trait loci for Cd and Zn concentrations of brown rice grown in Cd-polluted soils," *Euphytica*, vol. 180, no. 2, pp. 173–179, 2011.

[137] A. Distelfeld, I. Cakmak, Z. Peleg et al., "Multiple QTL-effects of wheat Gpc-B1 locus on grain protein and micronutrient concentrations," *Physiologia Plantarum*, vol. 129, no. 3, pp. 635–643, 2007.

[138] R. Shi, H. Li, Y. Tong, R. Jing, F. Zhang, and C. Zou, "Identification of quantitative trait locus of zinc and phosphorus density in wheat (*Triticum aestivum* L.) grain," *Plant and Soil*, vol. 306, no. 1-2, pp. 95–104, 2008.

[139] Y. Genc, A. P. Verbyla, A. A. Torun et al., "Quantitative trait loci analysis of zinc efficiency and grain zinc concentration in wheat using whole genome average interval mapping," *Plant and Soil*, vol. 314, no. 1-2, pp. 49–66, 2009.

[140] Z. Peleg, I. Cakmak, L. Ozturk et al., "Quantitative trait loci conferring grain mineral nutrient concentrations in durum wheat × wild emmer wheat RIL population," *Theoretical and Applied Genetics*, vol. 119, no. 2, pp. 353–369, 2009.

[141] P. F. Lonergan, M. A. Pallotta, M. Lorimer, J. G. Paull, S. J. Barker, and R. D. Graham, "Multiple genetic loci for zinc uptake and distribution in barley (*Hordeum vulgare*)," *New Phytologist*, vol. 184, no. 1, pp. 168–179, 2009.

[142] B. Sadeghzadeh, Z. Rengel, C. Li, and H. Yang, "Molecular marker linked to a chromosome region regulating seed Zn accumulation in barley," *Molecular Breeding*, vol. 25, no. 1, pp. 167–177, 2009.

[143] M. G. Lung'aho, A. M. Mwaniki, S. J. Szalma et al., "Genetic and physiological analysis of iron biofortification in Maize Kernels," *PLoS ONE*, vol. 6, no. 6, article e20429, 2011.

[144] D. Ŝimić, S. Mladenović Drinić, Z. Zdunić et al., "Quantitative trait loci for biofortification traits in maize grain," *Journal of Heredity*, vol. 103, no. 1, pp. 47–54, 2012.

[145] S. Beebe, A. V. Gonzalez, and J. Rengifo, "Research on trace minerals in the common bean," *Food and Nutrition Bulletin*, vol. 21, no. 4, pp. 387–391, 2000.

[146] S. H. Guzmán-Maldonado, O. Martínez, J. A. Acosta-Gallegos, F. Guevara-Lara, and O. Paredes-López, "Putative quantitative trait loci for physical and chemical components of common bean," *Crop Science*, vol. 43, no. 3, pp. 1029–1035, 2003.

[147] K. A. Cichy, S. Forster, K. F. Grafton, and G. L. Hosfield, "Inheritance of seed zinc accumulation in navy bean," *Crop Science*, vol. 45, no. 3, pp. 864–870, 2005.

[148] J. R. Gelin, S. Forster, K. F. Grafton, P. E. McClean, and G. A. Rojas-Cifuentes, "Analysis of seed zinc and other minerals in a recombinant inbred population of navy bean (*Phaseolus vulgaris* L.)," *Crop Science*, vol. 47, no. 4, pp. 1361–1366, 2007.

[149] M. W. Blair, C. Astudillo, M. A. Grusak, R. Graham, and S. E. Beebe, "Inheritance of seed iron and zinc concentrations in common bean (*Phaseolus vulgaris* L.)," *Molecular Breeding*, vol. 23, no. 2, pp. 197–207, 2009.

[150] K. A. Cichy, G. V. Caldas, S. S. Snapp, and M. W. Blair, "QTL analysis of seed iron, zinc, and phosphorus levels in an andean bean population," *Crop Science*, vol. 49, no. 5, pp. 1742–1750, 2009.

[151] M. W. Blair, J. I. Medina, C. Astudillo et al., "QTL for seed iron and zinc concentration and content in a Mesoamerican common bean (*Phaseolus vulgaris* L.) population," *Theoretical and Applied Genetics*, vol. 121, no. 6, pp. 1059–1070, 2010.

[152] M. W. Blair, C. Astudillo, J. Rengifo, S. E. Beebe, and R. Graham, "QTL analyses for seed iron and zinc concentrations in an intra-genepool population of Andean common beans (*Phaseolus vulgaris* L.)," *Theoretical and Applied Genetics*, vol. 122, no. 3, pp. 511–521, 2011.

[153] B. Zhang, P. Chen, A. Shi, A. Hou, T. Ishibashi, and D. Wang, "Putative quantitative trait loci associated with calcium content in soybean seed," *Journal of Heredity*, vol. 100, no. 2, pp. 263–269, 2009.

[154] G. Ding, M. Yang, Y. Hu et al., "Quantitative trait loci affecting seed mineral concentrations in *Brassica napus* grown with contrasting phosphorus supplies," *Annals of Botany*, vol. 105, no. 7, pp. 1221–1234, 2010.

[155] M. R. Broadley, J. P. Hammond, G. J. King et al., "Shoot calcium and magnesium concentrations differ between subtaxa, are highly heritable, and associate with potentially pleiotropic loci in *Brassica oleracea*," *Plant Physiology*, vol. 146, no. 4, pp. 1707–1720, 2008.

[156] M. R. Broadley, S. Ó. Lochlainn, J. P. Hammond et al., "Shoot zinc (Zn) concentration varies widely within *Brassica oleracea* L. and is affected by soil Zn and phosphorus (P) levels," *Journal of Horticultural Science and Biotechnology*, vol. 85, no. 5, pp. 375–380, 2010.

[157] J. Wu, Y. X. Yuan, X. W. Zhang et al., "Mapping QTLs for mineral accumulation and shoot dry biomass under different Zn nutritional conditions in Chinese cabbage (*Brassica rapa* L. ssp. pekinensis)," *Plant and Soil*, vol. 310, no. 1-2, pp. 25–40, 2008.

[158] M. R. Broadley, J. P. Hammond, G. J. King et al., "Biofortifying *Brassica* with calcium (Ca) and magnesium (Mg)," in *Proceedings of the 16th International Plant Nutrition Colloquium*, Paper 1256, 2009.

[159] N. K. Subramanian, *Genetics of mineral accumulation in potato tubers*, Ph.D. thesis, University of Nottingham, UK, 2012.

[160] P. Lucca, S. Poletti, and C. Sautter, "Genetic engineering approaches to enrich rice with iron and vitamin A," *Physiologia Plantarum*, vol. 126, no. 3, pp. 291–303, 2006.

[161] A. A.T. Johnson, B. Kyriacou, D. L. Callahan et al., "Constitutive overexpression of the OsNAS gene family reveals single-gene strategies for effective iron- and zinc-biofortification of rice endosperm," *PLoS ONE*, vol. 6, no. 9, article e24476, 2011.

[162] R. Sayre, J. R. Beeching, E. B. Cahoon et al., "The biocassava plus program: biofortification of cassava for sub-Saharan Africa," *Annual Review of Plant Biology*, vol. 62, pp. 251–272, 2011.

[163] M. Vasconcelos, K. Datta, N. Oliva et al., "Enhanced iron and zinc accumulation in transgenic rice with the ferritin gene," *Plant Science*, vol. 164, no. 3, pp. 371–378, 2003.

[164] M. Suzuki, K. C. Morikawa, H. Nakanishi et al., "Transgenic rice lines that include barley genes have increased tolerance to low iron availability in a calcareous paddy soil," *Soil Science and Plant Nutrition*, vol. 54, no. 1, pp. 77–85, 2008.

[165] S. Nandi, Y. A. Suzuki, J. Huang et al., "Expression of human lactoferrin in transgenic rice grains for the application in infant formula," *Plant Science*, vol. 163, no. 4, pp. 713–722, 2002.

[166] M. Vasconcelos, H. Eckert, V. Arahana, G. Graef, M. A. Grusak, and T. Clemente, "Molecular and phenotypic characterization of transgenic soybean expressing the Arabidopsis ferric chelate reductase gene, FRO2," *Planta*, vol. 224, no. 5, pp. 1116–1128, 2006.

[167] G. Drakakaki, S. Marcel, R. P. Glahn et al., "Endosperm-specific co-expression of recombinant soybean ferritin and *Aspergillus phytase* in maize results in significant increases in the levels of bioavailable iron," *Plant Molecular Biology*, vol. 59, no. 6, pp. 869–880, 2005.

[168] F. Goto, T. Yoshihara, and H. Saiki, "Iron accumulation and enhanced growth in transgenic lettuce plants expressing the iron- binding protein ferritin," *Theoretical and Applied Genetics*, vol. 100, no. 5, pp. 658–664, 2000.

[169] D. K. X. Chong and W. H. R. Langridge, "Expression of full-length bioactive antimicrobial human lactoferrin in potato plants," *Transgenic Research*, vol. 9, no. 1, pp. 71–78, 2000.

[170] C. Uauy, A. Distelfeld, T. Fahima, A. Blechl, and J. Dubcovsky, "A NAC gene regulating senescence improves grain protein, zinc, and iron content in wheat," *Science*, vol. 314, no. 5803, pp. 1298–1301, 2006.

[171] S. Park, N. H. Cheng, J. K. Pittman et al., "Increased calcium levels and prolonged shelf life in tomatoes expressing Arabidopsis H^+/Ca^{2+} transporters," *Plant Physiology*, vol. 139, no. 3, pp. 1194–1206, 2005.

[172] C. K. Kim, J. S. Han, H. S. Lee et al., "Expression of an *Arabidopsis CAX2* variant in potato tubers increases calcium levels with no accumulation of manganese," *Plant Cell Reports*, vol. 25, no. 11, pp. 1226–1232, 2006.

[173] J. Morris, K. M. Hawthorne, T. Hotze, S. A. Abrams, and K. D. Hirschi, "Nutritional impact of elevated calcium transport activity in carrots," *Proceedings of the National Academy of Sciences of the United States of America*, vol. 105, no. 5, pp. 1431–1435, 2008.

[174] S. Park, M. P. Elless, J. Park et al., "Sensory analysis of calcium-biofortified lettuce," *Plant Biotechnology Journal*, vol. 7, no. 1, pp. 106–117, 2009.

[175] P. J. White and M. R. Broadley, "Mechanisms of caesium uptake by plants," *New Phytologist*, vol. 147, no. 2, pp. 241–256, 2000.

[176] S. V. Fesenko, R. M. Alexakhin, M. I. Balonov et al., "An extended critical review of twenty years of countermeasures used in agriculture after the Chernobyl accident," *Science of the Total Environment*, vol. 383, no. 1–3, pp. 1–24, 2007.

[177] E. I. B. Chopin, B. Marin, R. Mkoungafoko et al., "Factors affecting distribution and mobility of trace elements (Cu, Pb, Zn) in a perennial grapevine (*Vitis vinifera* L.) in the Champagne region of France," *Environmental Pollution*, vol. 156, no. 3, pp. 1092–1098, 2008.

[178] F. A. Nicholson, S. R. Smith, B. J. Alloway, C. Carlton-Smith, and B. J. Chambers, "Quantifying heavy metal inputs to agricultural soils in England and Wales," *Water and Environment Journal*, vol. 20, no. 2, pp. 87–95, 2006.

[179] R. M. Welch, W. H. Allaway, W. A. House, and J. Kubota, "Geographic distribution of trace element problems," in *Micronutrients in Agriculture*, J. J. Mortvedt, F. R. Cox, L. M. Shuman, and R. M. Welch, Eds., pp. 31–57, Soil Science Society of America, Madison, Wis, USA, 2nd edition, 1991.

[180] R. L. Chaney, "Zinc phytotoxicity," in *Zinc in Soil and Plants*, A. D. Robson, Ed., pp. 135–150, Kluwer Academic Publishers, Dordrecht, The Netherlands, 1993.

[181] G. Gascó and M. C. Lobo, "Composition of a Spanish sewage sludge and effects on treated soil and olive trees," *Waste Management*, vol. 27, no. 11, pp. 1494–1500, 2007.

[182] M. D. Webber and S. S. Singh, "Contamination of agricultural soils," in *The Health of our Soils—Toward Sustainable Agriculture in Canada*, D. F. Acton and L. J. Gregorich, Eds., pp. 87–96, Centre for Land and Biological Resources Research, Ottawa, Canada, 1995.

[183] S. Babel and D. del Mundo Dacera, "Heavy metal removal from contaminated sludge for land application: a review," *Waste Management*, vol. 26, no. 9, pp. 988–1004, 2006.

[184] O. Hanay, H. Hasar, N. N. Kocer, and S. Aslan, "Evaluation for agricultural usage with speciation of heavy metals in a municipal sewage sludge," *Bulletin of Environmental Contamination and Toxicology*, vol. 81, no. 1, pp. 42–46, 2008.

[185] S. R. Smith, "A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge," *Environment International*, vol. 35, no. 1, pp. 142–156, 2009.

[186] G. Murtaza, A. Ghafoor, M. Qadir, G. Owens, M. A. Aziz, and M. H. Zia, "Disposal and use of sewage on agricultural lands in Pakistan: a review," *Pedosphere*, vol. 20, no. 1, pp. 23–34, 2010.

[187] J. Barth, F. Amlinger, E. Favoino et al., "Compost production and use in the EU," ORBIT e.V. / European Compost Network ECN, Weimar, Germany, 2008.

[188] A. K. Gupta, R. P. Singh, M. H. Ibrahim, and B.-K. Lee, "Fly ash for agriculture: implications for soil properties, nutrients, heavy metals, plant growth and pest control," *Sustainable Agriculture Reviews*, vol. 8, pp. 269–286, 2012.

[189] K. Mengel, E. A. Kirkby, H. Kosegarten, and T. Appel, *Principles of Plant Nutrition*, Kluwer Academic Publishers, Dordrecht, The Netherlands, 2001.

[190] U.K. Department of the Environment [DoE], *Code of Practice for Agriculture Use of Sewage Sludge*, DoE Publications, London, UK, 1996.

[191] Commission of the European Communities, "Council Directive of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture (86/278/EEC)," *Official Journal of the European Communities*, vol. 181, pp. 6–12, 1986.

[192] U.S. Environmental Protection Agency [USEPA], "Biosolids technology fact sheet: land application of biosolids (EPA 832-F-00-064)," USEPA, Washington, DC, USA, 2000.

[193] A. Kabata-Pendias and H. Pendias, *Trace Elements in Soils and Plants*, CRC Press, Boca Raton, Fla, USA, 1992.

[194] S. Ishikawa, N. Ae, and M. Yano, "Chromosomal regions with quantitative trait loci controlling cadmium concentration in brown rice (*Oryza sativa*)," *New Phytologist*, vol. 168, no. 2, pp. 345–350, 2005.

[195] J. Zhang, Y. G. Zhu, D. L. Zeng, W. D. Cheng, Q. Qian, and G. L. Duan, "Mapping quantitative trait loci associated with arsenic accumulation in rice (*Oryza sativa*)," *New Phytologist*, vol. 177, no. 2, pp. 350–355, 2008.

[196] P. J. White, K. Swarup, A. J. Escobar-Gutiérrez, H. C. Bowen, N. J. Willey, and M. R. Broadley, "Selecting plants to minimise radiocaesium in the food chain," *Plant and Soil*, vol. 249, no. 1, pp. 177–186, 2003.

[197] R. L. Chaney, J. S. Angle, C. L. Broadhurst, C. A. Peters, R. V. Tappero, and D. L. Sparks, "Improved understanding of hyperaccumulation yields commercial phytoextraction and phytomining technologies," *Journal of Environmental Quality*, vol. 36, no. 5, pp. 1429–1433, 2007.