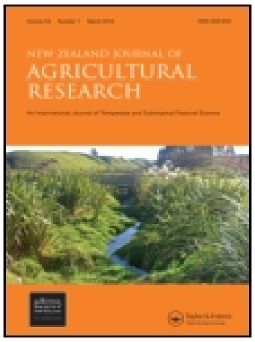

New Zealand Journal of Agricultural Research

ISSN: 0028-8233 (Print) 1175-8775 (Online) Journal homepage: https://www.tandfonline.com/loi/tnza20

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To cite this article: L. M. Condron , K. C. Cameron , H. J. Di , T. J. Clough , E. A. Forbes , R. G. McLaren & R. G. Silva (2000) A comparison of soil and environmental quality under organic and conventional farming systems in New Zealand, New Zealand Journal of Agricultural Research, 43:4, 443-466, DOI: 10.1080/00288233.2000.9513442

To link to this article: https://doi.org/10.1080/00288233.2000.9513442

Published online: 17 Mar 2010.

New Zealand Journal of Agricultural Research, 2000, Vol. 43: 443–466
0028–8233/00/4304–0443 $7.00/0

A comparison of soil and environmental quality under organic and conventional farming systems in New Zealand

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Abstract Organic farming in its various forms is seen by many as a sustainable alternative to conventional farming. This review considers and compares aspects of soil and environmental quality associated with organic and conventional farming systems under New Zealand conditions. The sustainability parameters considered include soil quality, nutrient dynamics, nutrient budgets, trace elements, and pesticides. The review used information from appropriate comparative studies conducted in New Zealand and overseas. However, because of the shortage of data on nutrient dynamics under organic systems in New Zealand, we also used a nutrient balance model (OVERSEER) and a nitrogen leaching estimation model to assess the comparative sustainability of typical model systems. Interpretation of the measured data coupled with the results of the modelling exercise suggests that organic farming carried out according to the Bio-Gro New Zealand production standards can be sustainable if sufficient amounts of nutrient are returned to match removal and losses. Biodynamic farming may be unsustainable because nutrients removed in farm produce are not adequately replaced. Soil organic matter content and biological activity is generally higher under both types of organic system than under conventional systems. Trace element availability and use may limit the sustainability of organic systems if no attempt is made to address natural deficiencies common in New Zealand soils. The reduced use of pesticides may be beneficial for the wider environment. The main conclusion is that a concerted research effort is urgently required to address various soil and environmental quality issues associated with the large-scale adoption of organic farming practices in New Zealand.

Keywords soil quality; organic farming; Bio-Gro; conventional farming; nutrient cycling; trace elements; heavy metals; pesticides; sustainability; biodynamics; environment

INTRODUCTION

This review has been written at a stage of New Zealand agriculture when farmers using conventional methods (essentially the practices of the last 50 years based on the use of synthetic fertilisers and agrochemicals to maximise production) are being challenged on a number of fronts. Plant and animal breeding methods are being refined by molecular biology techniques. Farming systems, on the other hand, are being broadened by a range of alternatives presented under the "organic" and "biodynamics" banners. The pace of change has left many within the agricultural industry grappling with their future options and attempting to understand new concepts and definitions in order to make sense of the debate.

The objective of the paper is, therefore, to attempt to conduct a comparison of soil and environmental quality under organic and conventional New Zealand farming systems that will answer some of the questions about the sustainability of organic farming. Within that context the integration of scientific data reported from studies on organic and biodynamic farms has not been easy

A00013
Received 6 March 2000; accepted 27 July 2000

since the practices on such farms are not necessarily similar. As a result, every attempt has been made to identify sources of data and so caution the reader about these differences.

Organic farming is being increasingly promoted worldwide as a sustainable alternative to conventional farming. Some countries, such as Germany and France, subsidise their farmers to convert to organic farming in order to become more sustainable. Organic agriculture now represents up to 10% of the food production systems in Switzerland and Austria, while in the USA, France, and Japan organic production is increasing at over 20% per annum (FAO 1999). Most estimates project annual increases in the USA market of 15–25% until c. 2005 (Richman 1999).

Organic farming has been defined as "an approach to farming that seeks to create an integrated, sustainable and humane agricultural system" (MAF 1994, p. 2), while MacKerron et al. (2000, p. 60) considered that "there is no single definition of 'organic' farming as the term applies to a movement rather than a single policy". In New Zealand there has been a dramatic escalation of organic food production since 1990. Bio-Gro certified organic licensees have increased from 172 in 1990 to 289 (c. 9000 ha) in 1997 according to a MAF (1997) report. In addition, two large corporates, Heinz-Wattie's Australasia Ltd and the New Zealand Kiwifruit Marketing Board (NZKMB), have entered the market and pulled a number of other entities in their wake as they formed the backbone of an organic food exporting industry (MAF 1997). More recently, Heinz-Wattie's Australasia Ltd and Lincoln University agreed in 1999 to establish an organic production farm of 58 ha at Lincoln to provide growers with information that would assist more farm conversions to organics.

Bio-Gro New Zealand was established in 1983 and is the leading organic certification agency in New Zealand. Bio-Gro New Zealand is an accredited member of the International Federation of Organic Agriculture Movements (IFOAM) which is based in Germany. What makes organic farming unique is that under the Bio-Gro certification programme synthetic inputs are avoided, or largely excluded, and an improvement in soil quality through the use of crop rotations is a central objective. Fertiliser materials are permitted, but the Bio-Gro standards (Bio-Gro New Zealand 1998, p. 27) clearly state that "the underlying principle is that all fertilisers must be worked on by soil or compost organisms before the nutrients are plant assimilable". In addition to appropriate composts and plant materials, nutrients may be added to an organic system in the form of various minerals and organic compounds (Table 1).

The key characteristics of organic farming are: protecting the long-term fertility and quality of the soil; providing nutrients in natural and organic fertilisers; nitrogen self-sufficiency through legumes; weed, disease, and pest control through crop rotations, natural predators, diversity, organic manuring, and limited biological and chemical intervention; extensive management of livestock; minimising the impact on the wider environment.

Biodynamics is a systems approach whereby the farm is viewed as a living whole in which the various farm activities are interrelated. The

Table 1 Fertiliser usage under Bio-Gro New Zealand production standards, from Bio-Gro New Zealand (1998).

Permitted fertilisers	
Gypsum (calcium sulphate)	Approved forms of sulphur, pelletised with bentonite
Dolomite (magnesium limestone)	Approved forms of seaweed and fish products with no additions other than potable water
Approved forms of elemental sulphur	Liquid brews made from clean material and potable water only
Glauconite (greensands)	Unrefined, unadulterated rock or sea salt
Feldspar	
Approved forms of rock phosphate	
Limestones	
Rock minerals	
Restricted fertilisers	
Approved forms of trace elements	Wood ash from untreated wood
Langbeinite rock (potassium magnesium sulphate)	Rock minerals, e.g., Borax
Magnesium sulphate (Epsom salts)	Blood, bone, and feather products, paunch, and dairy whey, if they meet other requirements
Potassium sulphate (regular use may preclude full Bio-Gro status)	

Biodynamic Farming and Gardening Association (New Zealand) Inc. originated in the late 1930s. It employs a full-time advisor, holds frequent regional field days, and monitors the awarding of the "Demeter" trademark (granted in 1986) for certified produce grown by biodynamic methods. The nature and extent of permitted nutrient inputs are more restricted under biodynamic systems compared with organic systems. One of the fundamental aims of biodynamic farming is to attain "a self-sustaining system where inputs of almost all kinds tend to gravitate to nil" (Biodynamic Farming and Gardening Association 1989). Nonetheless, nutrient input can occur in biodynamic systems via the import of materials for composting such as manure and applications of selected liquid manures (e.g., seaweed, fish, nettle (*Urtica dioica*)).

The objective of sustainability is an overarching tenet of organic farming. By the absence of soluble fertilisers and pesticide inputs, certified organic production aims to minimise the potential adverse affects of farming on the environment, both on- and off-farm (Bio-Gro New Zealand 1998). In spite of both an exceptional growth rate and the use of an existing prescription for organic farming across international boundaries, Lipson (1997) concluded that there was a surprising lack of research regarding a variety of aspects of organic agriculture. There appears to be very little reliable research conducted in New Zealand (Manhire 2000) to provide a direct comparison of the sustainability of organic farming versus conventional farming (Arden-Clark & Hodges 1988; Nguyen et al. 1995) at the farm level or at the agroecosystem level (Lewandowski et al. 1999), or to examine the changes that occur during and after the transition period.

It is essential that we compare New Zealand organic farming against our conventional farming systems, rather than assume that overseas comparisons are valid in New Zealand. Unlike Western Europe we do not have large numbers of housed animals to provide bulk manure for organic or biodynamic farming. Meanwhile, our traditional mixed cropping and clover-based pasture systems are more similar to the organic farming concept than they are to the monoculture farming practices of the USA and Europe.

This paper reviews our current knowledge of the differences in soil and environmental quality under organic and conventional New Zealand farming systems by examining the following key parameters: soil quality; nutrient supply, cycling, and losses; nutrient budgets, essential trace elements, and heavy metal contamination; pesticides; and yield and production issues.

SOIL QUALITY

The concepts of soil quality and soil health have been extensively discussed elsewhere by Doran (1996) and Gregorich (1996). In this section we look at the biological, chemical, and physical parameters of the soil that scientists measure in order to assess soil quality. In one of the few detailed New Zealand studies reported to date, Reganold et al. (1993) compared soil properties on pairs of biodynamic and conventional systems over a 4-year period. The key results of this study are shown in Table 2.

Soil organic matter content was found to be significantly higher on most of the biodynamic farms than on their conventional counterparts (Table 2). The greater amounts of organic matter in the biodynamically farmed soil contributed to the better soil structure observed on these farms. The bulk density of the surface 20 cm of soil was lower on the biodynamic farms and soil penetration resistance was also significantly lower.

Soil respiration, mineralisable nitrogen (N), and the ratio of mineralisable N to organic carbon (C) were significantly higher in the biodynamic farm indicating that there was a higher level of microbiological activity in this soil than in the conventionally farmed soil. Earthworm numbers were found to be significantly higher on the biodynamically farmed soil than in the conventionally farmed soil (175 earthworms m^{-2} compared with 21 earthworms m^{-2}, respectively).

Although there was a large amount of variation in soil chemical properties, cation exchange capacity, and total N between farms, the levels were generally higher in the biodynamically farmed soil, and extractable phosphorus (P), extractable sulphur (S), and soil pH were lower (Table 2).

Murata & Goh (1997) also provided a detailed comparison of the effects of crop rotations on soil organic matter in a pair of biodynamic and conventional mixed cropping farms in Canterbury. Total C, total N, microbial biomass C, and microbial biomass N were found to be higher in the cropping phase of the biodynamic system than in the conventional system (Table 3).

Although there are clearly insufficient New Zealand-based results to draw definite conclusions about organic farming systems, the trends towards improved soil quality indicators under biodynamic farming systems (especially biological indicators) are paralleled by key overseas research reports on organic farms.

Liebig & Doran (1999) compared soil quality indicators on five paired organic and conventional farms in Nebraska. Organic C, total N, microbial biomass C, and microbial biomass N were consistently higher across all organic farms compared with conventional farms (Table 4). Averaged across all locations, there was 22% more organic C and 20% more total N under organic versus conventional farms. The higher levels of microbial biomass C and N on organic farms indicated that there was a more active microbial population and this was supported by the higher soil respiration rates observed. Topsoil depth, available water holding capacity, and earthworm numbers were generally greater on the organic farms, while soil bulk density was generally lower. Trends in extractable P were not consistent and soil pH was generally higher on the organic farms (Table 4).

Table 2 Mean values of aggregated soil data from organic and conventional farms in the North Island of New Zealand from Reganold et al. (1993). Data were analysed with ANOVA so that the variation due to different enterprises or soils was absorbed or removed. Topsoil thickness includes surface and subsurface (A) horizons. *, $P < 0.01$.

Soil property	Organic farms	Conventional farms
pH	6.10	6.29*
Carbon (%)	4.84*	4.27
Bulk density (Mg m^{-3})	1.07	1.15*
Penetration resistance (0 to 20 cm) (MPa)	2.84	3.18*
Respiration (µl O_2 h^{-1} g^{-1})	73.7*	55.4
Mineralisable N (mg kg^{-1})	140.0*	105.9
Ratio of mineralisable N to C (mg g^{-1})	2.99*	2.59
Topsoil thickness (cm)	22.8*	20.6
CEC (cmol kg^{-1})	21.5*	19.6
Total N (mg kg^{-1})	4840*	4260
Total P (mg kg^{-1})	1560	1640
Extractable P (mg kg^{-1})	45.7	66.2*
Extractable S (mg kg^{-1})	10.5	21.5*
Extractable Ca (cmol kg^{-1})	12.8	13.5
Extractable Mg (cmol kg^{-1})	1.71	1.68
Extractable K (cmol kg^{-1})	0.97	1.00

Table 3 Effects of conventional and biodynamic arable and pastoral rotation stages on soil total carbon (TC), total nitrogen (TN), C:N ratio, microbial biomass carbon (BC), microbial biomass nitrogen (BN), and BC:BN, BC:TC, and BN:TN ratios on farms in Canterbury, New Zealand, from Murata & Goh (1997). Soil samples are from the present rotation stage with previous rotation history (e.g., C_1P_3 = first year crop after 3 year pasture). LSD, least significant difference.

Crop	TC (%)	TN (%)	C:N (%)	BC (µg g^{-1})	BN (µg g^{-1})
Conventional crop					
C_1P_3	3.04	0.30	10.1	195	50
Biodynamic crop					
C_1P_3	3.90	0.38	10.2	525	107
Conventional pasture					
P_2C_3	3.39	0.33	10.3	587	63
Biodynamic pasture					
P_2C_3	3.16	0.31	10.3	374	75
LSD ($P \leq 0.05$)	0.09	0.01	0.03	88	22

Liebig & Doran (1999) and others (e.g., Gerhardt 1997) have concluded that the capacity of organic farms to improve soil quality is mainly due to the use of more diverse crop sequences, inclusion of ley crops (i.e., a short period of pasture or another restorative crop in a rotation), applications of organic amendments, and less frequent tillage.

Higher microbial pool sizes and greater microbial activity rates in organic versus conventional farm soils have also been reported in Europe. For example, in one of the most comprehensive, long-term (14-year) comparison studies, Mäder et al. (1999) reported a significantly greater soil microbial biomass C and enzyme activity in biodynamic and organically farmed soils compared with conventional farms using mineral fertilisers (Table 5).

Earthworm numbers have also been found to be higher in organic systems than conventional systems (Mäder et al. 1999) (Fig. 1), and similar results have been reported by others who have attributed this to the regular use of animal and green manures on organic farms (Gerhardt 1997). Mulches that are regularly used on organic farms also promote earthworm activities more than do incorporated crop residues or burned stubble on conventional farms (Doran & Werner 1990; Gerhardt 1997).

Mäder et al. (1999) reported similar amounts of organic C and total N in organic compared with conventional systems but lower levels of extractable P and potassium (K) in the organic systems due to the lower nutrient inputs (Table 6).

In a paired-farm comparison study in Canterbury (New Zealand), Goh & Nguyen (1991) reported lower Olsen P (e.g., 12 versus 22 mg P kg^{-1}) and phosphate-extractable S (e.g., 4 versus 11 mg S kg^{-1}) in soil on a biodynamic farm that received no fertiliser inputs than a conventional farm that received fertiliser. Both farms were on the same property and had the same superphosphate topdressing history for 20 years. The biodynamic farm was developed 7 years before the measurements were made and both farms included pastoral as well as cropping phases.

In a survey of nine paired farms throughout New Zealand, Haystead (1987) reported that where organic farmers used adequate levels of rock P and elemental S the soil fertility levels were not significantly different from their conventional counterparts.

With regard to wider issues relating to environmental quality, van der Weerden et al. (1999) compared nitrous oxide emissions from organic versus conventional farming systems after cultivation, and provided some interesting discussion on the effect of fertiliser (conventional) versus soil cultivation (organic) on the soil mineral N content and associated gaseous emissions.

Table 4 Effects of conventional and organic practices on selected soil quality indicators for the top 30.5 cm on farms in Nebraska, USA, from Liebig & Doran (1999). Management systems are matched by soil type. Microbial quotient = (Microbial biomass C ÷ Organic C) × 100. Least significant differences (LSD) at $P \leq 0.1$ for comparison are given for each indicator.

Location	Soil bulk density	Available water holding capacity	Soil pH	Extractable P	Organic C	Total N	Microbial biomass C	Microbial biomass N	Microbial quotient
Farm type	$Mg\ m^{-3}$	$m^3\ m^{-3}$		$kg\ ha^{-1}$					%
Giltner									
Conventional	1.34	0.15	5.93	25	53359	4681	651	32	1.22
Organic	1.26	0.17	6.25	57	55607	4778	816	60	1.46
LSD	0.02*	0.01*	0.38	18*	2773	342	210	15*	0.31
Valley									
Conventional	1.35	0.15	4.96	89	42337	2849	464	26	1.10
Organic	1.08	0.21	6.46	66	57622	3967	775	92	1.35
LSD	0.09*	0.01*	0.17*	14*	3192*	266	138*	29*	0.33
Deweese									
Conventional	1.22	0.18	6.12	133	55567	4720	654	47	1.17
Organic	1.18	0.20	6.59	197	70278	6255	1192	105	1.70
LSD	0.07	0.02	0.25*	18*	5068*	616*	351*	21*	0.48

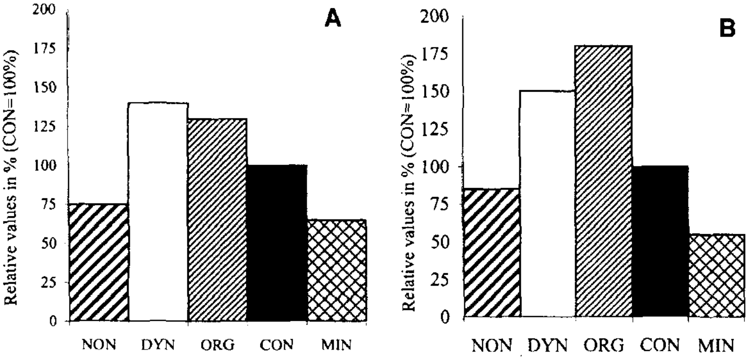

Fig. 1 Biomass (**A**) and abundance (**B**) of earthworms in the control (NON), biodynamic (DYN), organic (ORG), and mineral (MIN) systems, relative to the conventional system (CON = 100%). From Mäder et al. (1999) using data collected in Switzerland from Pfiffner et al. (1993).

NUTRIENT DYNAMICS AND NUTRIENT BUDGETS

Many New Zealand soils are low in native P and S. Since the early 1900s regular additions of these nutrients in the form of superphosphate (together with essential trace elements such as molybdenum (Mo), which is needed by clovers for N_2 fixation) have been required to establish and maintain productive N_2-fixing grass-clover pasture in permanent grassland and mixed cropping systems (Kemp et al. 2000). A significant proportion of the nutrients applied in fertiliser and associated fixed N is retained in the soil in mineral and/or organic forms. These residual forms of nutrients make a

Table 5 Soil microbial properties of five systems in a field experiment near Basle, Switzerland, after two winter wheat crop rotations, adapted from Mäder et al. (1993). NON, control; DYN, biodynamic; ORG, organic; CON, conventional; MIN, mineral. Significant differences are indicated by different letters in the same row ($P = 0.05$).

Soil property	NON	DYN	ORG	CON	MIN
Microbial biomass (mg C_{mic} kg^{-1})	361a	603d	528c	443b	359a
C_{mic} C_{org}^{-1} ratio (g kg^{-1})	24a	34c	32c	27b	25a
Respiration (μg CO_2 C 15 d^{-1} kg^{-1})	258a	324a	302a	295a	273a
Dehydrogenase (mg TPF 6 h^{-1} kg^{-1})	42a	106d	85c	59b	46a
Catalase (g H_2O_2 h^{-1} kg^{-1})	3.6a	6.05c	5.4bc	4.4ab	4.0a
Protease (mg tyrosine equivalents 2 h^{-1} kg^{-1})	233a	810d	613c	476b	378b
Alkaline phosphatase (mg phenol 16 h^{-1} kg^{-1})	112a	1607d	973c	531b	416ab
Saccharase (g red. sugar 3 h^{-1} kg^{-1})	1.18a	2.29d	1.97c	1.58b	1.49ab

Table 6 Soil chemical properties of five systems in a field experiment near Basle, Switzerland after two crop rotations of [1]winter wheat (Mäder et al. 1993) and [2]winter wheat/beetroots (Oberson et al. 1993) NON, DYN, ORG, CON, MIN as in Table 5. DL, extracted with calcium lactate; F_m, mean flux of phosphate ions between solid and liquid phase of soil as determined by ^{32}P isotopic exchange method (Oberson et al. 1993). Significant differences are indicated by different letters in the same row ($P = 0.05$).

Soil property	NON	DYN	ORG	CON	MIN
[1]pH (H_2O)	6.2a	6.9c	6.5b	6.2a	6.1a
[1]Total organic carbon (g C_{org} kg^{-1})	14.8a	17.7c	16.4b	16.1b	14.5a
[1]Total nitrogen (g N kg^{-1})	1.34a	1.69c	1.50b	1.47ab	1.41ab
[1]Extractable P (DL) (mg P kg^{-1})	11a	33c	25b	38d	24b
[2]F_m (mg P min^{-1} kg^{-1})	6a	45b	21a	24a	17a
[1]Extractable K (DL) mg K kg^{-1})	48a	61b	58b	101d	73c
[1]Extractable Ca (HCl/H_2SO_4) (g Ca kg^{-1})	1.81a	2.47b	1.96a	1.84a	1.78a
[2]Extractable Mg (DL) (mg Mg kg^{-1})	68a	101bc	116c	95b	89b

significant contribution to ongoing plant requirements following conversion of land from conventional to organic farming. Therefore, these residuals must be included in any consideration and/or assessment of the long-term sustainability and sustainable management of organic and biodynamic systems in New Zealand.

In one of the few comparative studies conducted in New Zealand, Nguyen et al. (1995) provided nutrient budgets for one pair of conventional and biodynamic and two pairs of conventional and organic mixed cropping farms in Canterbury. The major inputs of N, P, and S as fertiliser and biological sources and the major removals of these nutrients in harvested products (grain, meat, and wool) were calculated for each year of the rotation and a total budget for the full rotation presented. Nutrient budgets were generally balanced or positive for the conventional mixed cropping farms (Table 7). Nitrogen budgets were positive on all the organic farms, primarily due to the greater amount of N-fixation during the 3–4-year pasture phase, compared with the 1–2-year pasture phase on the conventional farms. Phosphorus and S budgets were positive on one organic farm (Table 7) because it used sufficient inputs of rock P fertiliser, elemental S fertiliser, and compost. Phosphorus budgets were negative on the biodynamic and one of the organic farms, and the S budget was negative on the biodynamic farm (Table 8). These authors concluded that the conventional mixed cropping systems currently practised in Canterbury, which involve arable crops in rotation with grazed ryegrass (*Lolium perenne*)/clover pasture or clover seed crops, are sustainable in terms of nutrient supply and removal, whilst the biodynamic farming system studied require greater nutrient inputs (especially P and S) to remain sustainable.

Since the use of soluble mineral fertilisers is prohibited under Bio-Gro standards for organic farming, careful consideration needs to be given to the quantities of permitted sparingly soluble fertilisers (e.g., RPR, elemental S) required to replace non-renewable nutrients (principally P and S) removed in produce and by physical and biological transfer, and maintain overall plant (and animal) production at economically viable levels by maintaining soil biological and chemical fertility (including N_2 fixation).

It is generally assumed that organically farmed systems will have lower nutrient losses (especially N and P) compared with conventionally farmed systems that rely on soluble fertiliser inputs to achieve the desired level of plant and/or animal

Table 7 Budgets of nitrogen, phosphorus, and sulphur inputs and outputs (kg ha^{-1}) in conventional and organic farms in Canterbury, New Zealand, from Nguyen et al. (1995).

	N			P		S	
	Inputs						
Farming systems and rotation	Fertiliser	Fixation	Outputs	Inputs	Outputs	Inputs	Outputs
Conventional							
Wheat							
Grain	58		122		23		9
Burnt stubble			17		3		4
Ryegrass-clover	50	150		25		30	
Clover seeds		150	18		3		1
Peas	19	75	197	13	23	9	9
Beans	38	150	300	25	36	18	19
Animal products			41		6		7
Individual Total	165	525	695	63	93	57	49
Combined Total	690						
Organic							
Wheat			113		15		8
Red beans	28	75	62	14	9	34	4
Peas		75	165	13	19	37	9
Pasture (4 year)		600		52		149	
Animal products			58		8		9
Individual Total	28	750	398	79	51	220	30
Combined Total	778						

production. However, as Watson & Younie (1995) have pointed out, this assumption is often based on subjective judgement because very few reliable comparisons of nutrient fluxes and budgets have been made between the two systems.

Watson & Younie (1995) calculated N balances for two organically and conventionally managed beef production farmlets in Scotland. The main difference between the farmlets was the use of 270 kg N ha^{-1} yr^{-1} on the conventional system and reliance on clover for N supply in the organic system. The organic farmlet was stocked at 2.1 s.u. ha^{-1} and the conventional farmlet at 2.7 s.u. ha^{-1}, and thus the N balance was calculated per stock unit rather than per hectare (Table 9). The "unaccounted-for N" in Table 9 represents immobilisation as well as leaching. The data in Table 9 illustrate the relatively lower efficiency of high inputs of N fertiliser to grassland and the potentially greater risk of N leaching under the conventional system compared with the organic system. The results also indicate that it is the quantity of N in the system rather than the source of N (i.e., biological fixation or fertiliser) which determines the likely environmental impact.

Research at one of the main organic farming research centres in the United Kingdom (Elm Farm) has indicated that the ploughing of grass-clover leys and clover crop residues can cause large leaching losses of N (Fig. 2). As longer term leys are used in organic farming than in conventional farming systems this may indicate that organic farming could increase environmental pollution. Nevertheless, Stopes & Philipps (1994) also showed that when averaged over a 3-year period during the rotation this loss represented approximately 17 kg N ha^{-1} yr^{-1} compared with the average loss from conventional farms of approximately 42 kg N ha^{-1} yr^{-1} over the same 3-year period.

Nitrogen, phosphorus, potassium, and sulphur requirements

The requirement and fate of nutrients in both conventional and organic agricultural systems depend on many variables such as land use, soil type, climate, grazing intensity, yields, and previous fertiliser history. Losses of nutrients from farming systems are determined by the amount of nutrients removed in products, leaching and atmospheric losses, and the immobilisation of nutrients into the soil. The magnitude of inputs of nutrients will depend on the fertilisers applied, nutrients brought onto the farm in other forms (e.g., supplementary feed), atmos-

Table 8 Budgets of phosphorus and sulphur inputs and outputs (kg ha^{-1}) in organic and biodynamic farms in Canterbury, New Zealand from Nguyen et al. (1995).

	P		S	
Crop	Inputs	Outputs	Inputs	Outputs
Organic				
Wheat	0.04	10	0.15	5
Peas	0.04	10	0.15	5
Barley	0.04	9	0.15	5
Oats	0.04	8	0.15	5
Oats	0.04	8	0.15	4
Buckwheat	0.04	8	0.15	4
Pasture (3 year)	21		32	
Animal products		7		8
Total	**21**	**54**	**33**	**31**
Biodynamic				
Wheat (clover oversown)	<0.01	15	<0.001	7
Peas				
Grain	<0.01	11	0.001	6
Sold straw		2		3
GF oats	<0.01		<0.001	
Buckwheat	<0.01	5	<0.001	3
Barley	<0.01	10	<0.001	4
Pasture (4 year)	<0.01		<0.001	
Animal products		11		13
Total	**<0.01**	**54**	**<0.001**	**36**

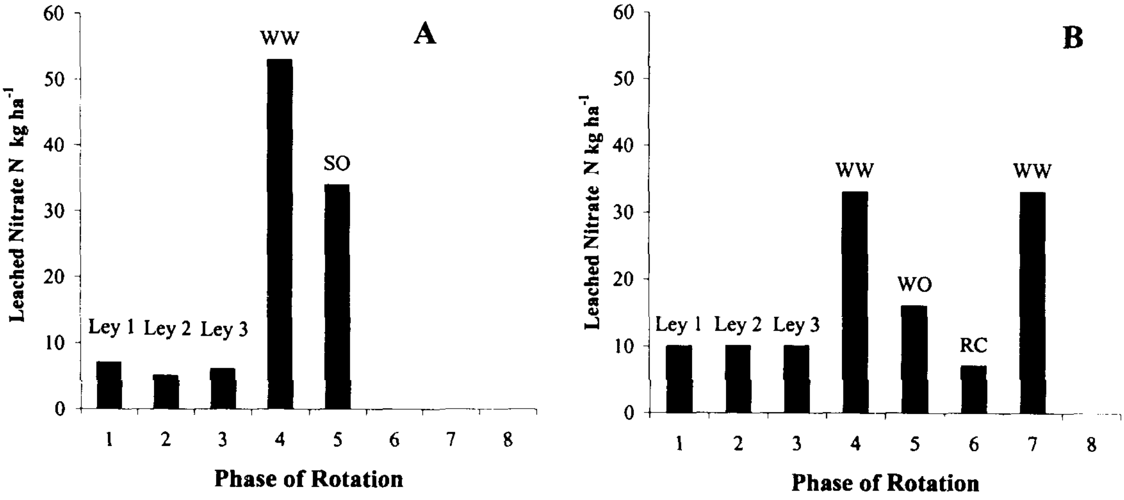

Fig. 2 Nitrate leached (kg N ha^{-1}) from each rotational phase on 2 organic farms in England 1988–1992, from Stopes & Philipps (1994). **A**, 5-year rotation; **B**, 7-year rotation. WW, Winter wheat; SW, Spring wheat; PO, Potatoes; SO, Spring Oats; RC, Red clover; WO, Winter oats. Ley, grass/clover.

pheric inputs, or nutrients released slowly from the soil. Soil type can vary greatly depending on parent material and soil formation processes, which in turn affect the soil's natural fertility. Generally, a farmer aiming for high yields must rely on more than the natural fertility of the soil. However, one aim of sustainable farming systems is to balance these inputs and outputs of nutrients otherwise soil nutrient reserves may be diminished and yields will drop accordingly. If some depletion occurs, then ultimately a new balance may be established. The fertiliser added to achieve the nutrient balance is termed maintenance, or sufficient, fertiliser, i.e., the fertiliser required to replace nutrient outputs from the system. Achieving the balance is not a simple matter and Fig. 3 shows the maintenance P

Table 9 Nitrogen balances for organically (O) and conventionally (C) managed beef production systems at Aberdeen, Scotland, (mean of two production cycles) expressed as kg N ha^{-1} and kg N per stock unit (kg N $s.u.^{-1}$), from Watson & Younie (1995).

kg N ha^{-1}	O	C	kg N $s.u.^{-1}$	O	C
Inputs			**Inputs**		
Atmospheric inputs	12	12	Atmospheric inputs	6	4
Non-symbiotic N fixed	5	5	Non-symbiotic N fixed	2	2
Fertiliser	0	270	Fertiliser	0	100
Symbiotic N fixed	146	0	Symbiotic N fixed	70	0
Bedding	7	9	Bedding	3	3
Purchased feed	46	61	Purchased feed	22	23
Livestock	5	7	Livestock	3	3
Total	**221**	**364**	**Total**	**106**	**135**
Outputs			**Outputs**		
Gaseous loss	60	84	Gaseous loss	29	31
Run-off	6	6	Run-off	3	2
Livestock	52	51	Livestock	25	19
Silage	0	7	Silage	0	3
Unaccounted-for N including leaching	103	216	Unaccounted-for N including leaching	49	80
Total	**221**	**364**	**Total**	**105**	**135**

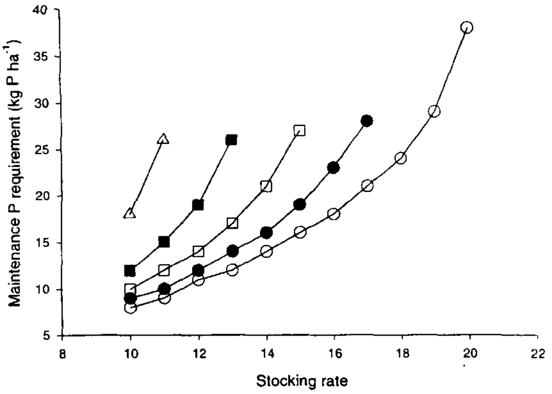

Fig. 3 Effect of stocking rate (s.u. ha^{-1}; where 1 stock unit corresponds to one 55-kg ewe with 1.0 lambs weaned, consuming annual DM of 550 kg) and pasture utilisation (Δ, PU50 = 50%; ■, PU60; □, PU70; ●, PU80; O, PU90) on P maintenance requirements in New Zealand (from Cornforth & Sinclair 1982, table 98).

requirement as it is affected by stocking rate and pasture utilisation (adapted from Cornforth & Sinclair 1982). While the sources of nutrients may differ between conventional and organic farms, the principles of nutrient supply are the same.

Nutrient budget models and simulations

In the absence of data comparing organic farming systems and conventional systems in New Zealand we have used two recently published nutrient budget models, OVERSEER™ for P, K, and S (Ledgard et al. 1999), and a more precise model, the nitrogen from leaching estimation (NLE) model (Di & Cameron 2000), for N leaching losses from dairy pasture systems.

OVERSEER™ model

The OVERSEER™ model is an empirical model that calculates average annual flows of N, P, K, and S in pastoral, cropping, and horticultural systems. The model contains a database with information on nutrient concentrations of fertilisers, animals, products, and crop residues. The equations within the model are based on studies carried out in New Zealand. For example, in pastoral systems, the soil organic matter is assumed to be in a steady state. Transfer of excreta to lanes, sheds, or camp sites is also built into the model. In arable crops, soil organic matter is assumed to change over time as affected by cultivation. The effects of timing of cultivation and fertiliser application and pre-crop management on the fate of nutrients are also taken into account based on past research.

Knowledge of inputs and outputs enables the balance of nutrients for a given agricultural system to be assessed. The nutrient budget model OVERSEER™ is supplied with various scenarios and the subsequent nutrient balance is determined, as shown in the next three sections for dairying, winter wheat, and apple growing. We have used base data from MAF Farm Monitoring Reports (MAF 1999) for the dairying and winter wheat examples, and from Wells (1992) for the apple orchard example, for these calculations. Some of the data used in the simulations were estimated, and the reliability of the budget will thus be determined by the accuracy of these estimations.

Dairying

A typical conventional dairy farm in Canterbury (MAF 1999) has an effective area of 155 ha, carries 440 dairy cows and 100 yearling heifers, and produces 139 050 kg milk solids at a stocking rate of 2.8 cows ha^{-1}.

Assumingthat the farm is 20 km from the coast, has a rainfall of 700 mm, and 500 mm of irrigation is applied. Soil type is sedimentary with an initial Olsen P of 30 mg kg^{-1}, medium K soil reserves with a K soil test of 10 mg kg^{-1}, and an organic-S test of 15 mg kg^{-1}. All supplements are made and consumed on the farm.

Based on this information, the OVERSEER™ model predicts an annual site nutrient requirement for P and K at 13 and 64 kg ha^{-1}, respectively, for the conventional farm (Fig. 4A). The current version of the model is not designed for S predictions for organic dairy systems so no comparison is made for S.

There is a scarcity of published information on organic dairy systems in New Zealand. However, the European literature suggests that organic conversions experience both a drop in milk production, due to changes in feeding regimes, and a decrease in stocking rates (Dalgaard et al. 1998; Olesen et al. 1999; Leslie 2000). If we assume that the typical Canterbury dairy farm above is converted to organic production, and production per cow remains constant, even at a stocking rate of 1 cow ha^{-1} there is a need for P and K at 6 and 22 kg ha^{-1}, respectively (Fig. 4A).

It is possible that production per cow may also decrease. Assuming a 30% decrease in per cow production, with a stocking rate of 1 cow ha^{-1}, then there is still a need for added nutrients at

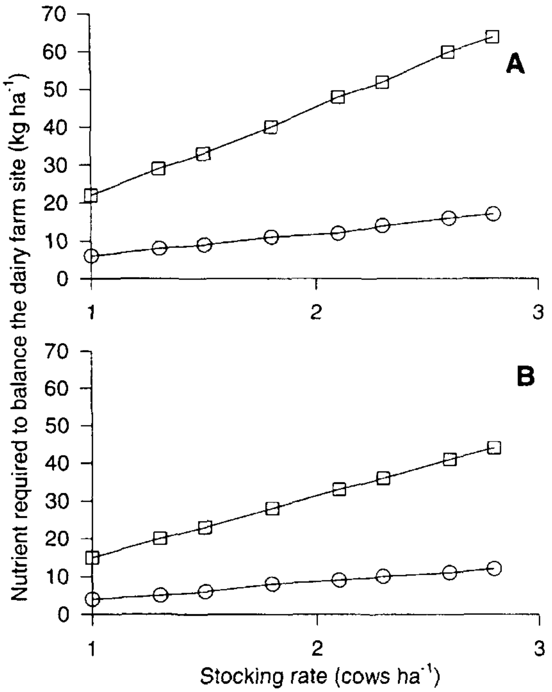

Fig. 4 Effect of stocking rate on the site P and K nutrient requirements at (**A**) 316 kg milk solids cow^{-1} and at (**B**) a lower production rate of 221 kg milk solids cow^{-1}, using OVERSEER. ○, P; □, K.

rates for P and K at 4 and 15 kg ha^{-1}, respectively (Fig. 4B).

Some of these nutrient requirements may be met by importing supplements onto the organic dairy farm (e.g., barley or hay). In practice, the transformations, fate, and availability of such imported nutrients would need to be considered on a case by case basis. However, it is clear that to ensure a nutrient balance in organic dairy systems the use of acceptable fertilisers (Table 1) will be required.

Winter wheat

A conventional winter wheat crop in Canterbury (MAF 1999) is 20 km from the coast, receiving 700 mm rainfall and no irrigation. The soil is sedimentary, Olsen P is 30 kg ha^{-1}, K test is 10 kg ha^{-1}, and no fertiliser was applied in the previous year. Cultivation is done in April, crop residues are burnt, and it is 2 years since the soil was in pasture.

For a yield of 5 t ha^{-1}, the P, K, S, and N required to complete the nutrient balance is 21, 38, 11, and 175 kg ha^{-1}, respectively, under conventional farming. In an organic situation the yields may be lower (Nguyen et al. 1995). If the yield was 3 t ha^{-1} then the P, K, S, and N required to complete the nutrient balance would be 13, 24, 7, and 139 kg ha^{-1}, respectively (Fig. 5). The burning of crop residues is not permitted in an organic situation and hence more nutrients will be retained in the soil. At a yield of 3 t ha^{-1} with the crop residues retained on the site, by ploughing in rather than burning, the soil P, K, and S requirements are further reduced to 12, 16, and 6 kg ha^{-1}, respectively. The N requirement is slightly higher at 140 kg ha^{-1} due to increased leaching and atmospheric losses from the site. These nutrients must then come from other inputs (Table 1) if the site is to sustain its level of fertility.

Grass/clover pasture is an important feature of organic crop rotations for restoring the soil N status. Returning straw to the soil also increases the organic matter pool from which N can potentially be mineralised, or into which N can be immobilised. As the period since the site was in pasture increases, the soil N status decreases due to organic matter mineralisation and subsequent leaching and atmospheric losses of N from the crop site. Maintaining a yield of 5 t ha^{-1} and comparing the organic and conventional systems described above, the leaching losses from the organic system (2 to 5 years since soil was in pasture) were on average 18% greater than the conventional system, despite higher immobilisation also occurring. This is because there was more organic matter in the system as a result of organic matter returns in the straw from which N could be mineralised and potentially leached.

Apple growing

In an apple orchard, nutrient inputs may be supplied via fertiliser, irrigation water, rainfall, sprays, and atmospheric deposits. While cycling within the orchard occurs with leaf fall, sward mowing, and return of prunings, the losses are constituted by leaching, gaseous emissions, pruning removal, and removal of the fruit crop (Haynes 1988).

Data from a semi-intensive (635 trees ha^{-1}) apple-growing system (the accepted standard in New Zealand) from Wells (1992) were used to develop an OVERSEER™ nutrient balance. Assumptions made include the property being 30 km from the coast, with 700 mm rainfall, 200 mm irrigation, an Olsen P of 30 mg kg^{-1}, and a soil K test of 10 mg kg^{-1}.

The requirements for P and K in the conventional and organic systems are similar at all

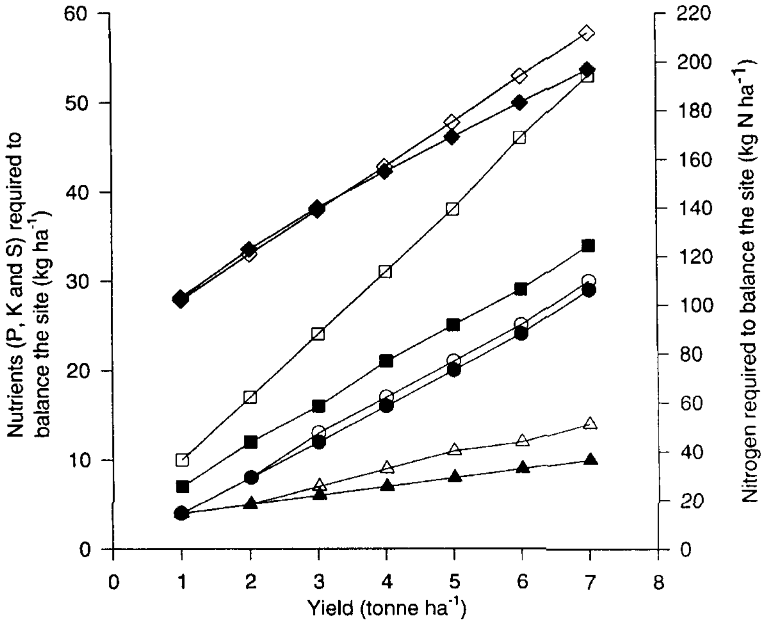

Fig. 5 Effect of increasing wheat yield on nutrient requirements for conventional (burnt straw, open symbols) and organic (straw residues incorporated, solid symbols) systems, using OVERSEER. ◇, N; ○, P; □, K; Δ, S.

production rates. Two of the main differences which can influence nutrient transformations are the presence or absence of a sward under the tree canopy, and the removal or mulching of prunings. A dominant effect of a sward is to increase both N fixation (assuming a legume is included in the sward) and the amount of N cycling in the system. Potassium is the most abundant nutrient in the apple fruit, and as the level of crop increases so too will the demand for K and K fertilisers (Haynes 1988). The nutrients required to balance the site budget of a semi-intensive apple orchard during its development up to the seventh harvest are shown in Fig. 6. The assumption is made that the organic system (with a pasture sward and prunings mulched) will produce at a rate equal to the conventional system (herbicide strip and prunings removed), and again we used data from Wells (1992). For intensive cropping (1905 trees ha^{-1}), nutrient requirements will obviously increase further. Negative N values indicate a surplus of N for crop requirements. The organic orchardist must consider how to use the permitted fertiliser options to replace nutrient losses for a particular system.

P, K, and S fertiliser options for conventional and organic farming

Forms of fertiliser available to supply P, K, and S are many and varied for the conventional farmer. One of the most commonly used conventional P fertilisers is superphosphate, which also contains S as gypsum. Various nutrient additions can be made to superphosphate. For example, K can be added so that all three elements can be applied in one application. Fertiliser options are more limited in organic farming.

As described earlier, a major difference in the philosophies between conventional and organic farming is the principle of not using soluble fertilisers on the organic farm. The effects of the different approaches can be seen if we look at the need for P and S. In many instances conventional farms would use superphosphate as a source of soluble P and S, while on an organic farm RPR and elemental S may be chosen.

Two major differences between superphosphate and RPRs are the availability of the P they contain and the amount of S in the two materials. Most of the P in superphosphate is water soluble (>90%), while in RPRs the P is only slowly available and it may take several years before all the P is plant available, depending on soil type and environmental conditions (Sinclair & Dyson 1988). The availability of P in RPRs is also influenced by soil acidity and rainfall with release of P being greater at pH <6.0 and rainfall >800 mm. Superphosphate and RPRs typically contain 12 and 1% S, respectively (Edmeades et al. 1990). The S in superphosphate is

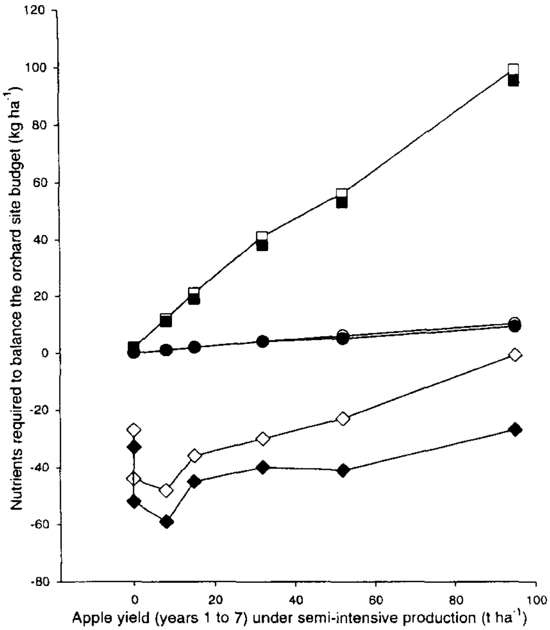

Fig. 6 Apple yield under semi-intensive production during the first 7 harvests. Yields increase as apple trees develop and mature. Data for conventional systems from Wells 1992, open symbols. Simulated organic data, solid symbols. Analysis performed using OVERSEER. ◇, N; ○, P; □, K.

readily available to plants in a sulphate form whereas elemental S must first be oxidised to sulphate by soil micro-organisms. Sulphur can be added to RPRs in the form of elemental S but must be finely ground to be effective.

Nitrogen leaching estimation model and simulations

The loss of N from agricultural systems is the major environmental quality issue in conventional farming. The recent development of the NLE model to predict losses of N has provided us with a valuable tool to assess the merits of different farming systems.

The NLE model

The NLE model of Di & Cameron (2000) is a semi-empirical model for estimating nitrate-N (NO_3^--N) leaching losses and critical N application rates (in order not to exceed the drinking water standard of 11.3 mg NO_3-N l^{-1}) for dairy pasture systems. It is based on the assumption that the amount of annual N leaching loss is determined by the amount of potentially leachable N in the soil (including mineral and mineralisable N) and the amount of drainage. The amount of potentially leachable N in the soil is determined by the major N cycling processes (all in kg N ha^{-1} yr^{-1}):

$$N_{PL} = N_{AD} + N_F + N_B + N_M + N_A - N_P - N_V - N_D \quad (1)$$

where N_{PL} = potentially leachable N;
N_{AD} = atmospheric deposition;
N_F = annual fertiliser or effluent N application rate;
N_B = annual biological N_2-fixation;
N_M = annual net N mineralisation;
N_A = annual animal N returns (mainly urine) to the pasture;
N_P = annual pasture N uptake;
N_V = volatilisation losses after the application of N fertiliser or effluent;
N_D = denitrification loss.

A quadratic relationship was used to relate the amount of N leached with the amount of potentially leachable N in the soil. It was derived from lysimeter studies conducted over a number of years. The model takes into account the effect of urine-N returns by the grazing animal on NO_3^--N leaching, by calculating a weighted average of N leaching losses from the grazed paddock on the basis of the fractional areas occupied by urine and non-urine areas:

$$N_L = N_{L1} \times P_1 + N_{L2} \times P_2 \quad (2)$$

where N_L is the NO_3^--N leaching losses from the grazed paddock, N_{L1} and N_{L2} are the leaching losses from the urine and non-urine patch areas, respectively, and P_1 and P_2 are the proportions of areas occupied by urine and non-urine patches, respectively. The value of P_1 varies with stocking rate, and, on average, it is probably between 0.2 and 0.3, and P_2 equals $(1-P_1)$. Nitrogen is released slowly from manure and it is included in the mineralisation component.

Simulations with the NLE model

Most of the studies that have been carried out so far are for conventional systems. The simulations shown in Table 10 were based on the data reported by Di et al. (1998a, 1998b, 1999), Silva et al. (1999), and Ledgard et al. (1996) for conventional dairy systems. For Scenarios 1 and 2 described in Table 10, the leaching losses are 19 and 46 kg N ha^{-1} yr^{-1}, respectively.

The NLE model predicts that the organic system is likely to result in lower N leaching losses than the comparable conventional system. At this time it is not possible to be more specific because there is a lack of information of N dynamics in organic dairy farming. The N dynamics data used here for the organic farm were simulations based on general information and the Bio-Gro standards (Bio-Gro New Zealand 1998). The features of the organic farm system that was constructed are lower stocking rates, zero N fertiliser inputs, and higher biological N_2 fixation than on a conventional farm. Two scenarios were simulated for both conventional and organic systems with different stocking rates (Table 10).

For the conditions shown in Table 10, the lower leaching losses in the organic system were in part because of the lower N input in the form of N fertilisers, and in part because of the lower stocking rate causing fewer numbers of urine patches. It has previously been shown that urine returns by the grazing animal can have a significant effect on NO_3^--N leaching on grazed dairy pastures (Silva et al. 1999). The lower stocking rates in the organic farm would reduce the areas covered by urine patches, thus resulting in lower leaching losses.

The modelled leaching losses are in line with those reported for conventional dairy systems in New Zealand (Ledgard et al. 1996; Silva et al. 1999), but the predictions for organic systems cannot be tested due to the lack of measured data. There is a clear need to determine N dynamics and leaching losses on organic dairy systems if a more comprehensive comparison is to be carried out on the environmental impacts of NO_3^--N leaching from conventional and organic dairy systems.

TRACE ELEMENTS

Essential elements

There is a dearth of reported studies comparing the trace element status in conventional and organic farms. As a result, this section of the review summarises our knowledge of trace element availability, and plant and animal requirements, in conventional farming systems and assesses the implications of farming organically on trace element-deficient soils within the Bio-Gro prescription.

Trace element deficiencies in soils have been a longstanding and widespread problem in conventional agriculture in New Zealand. As an example, cobalt (Co) deficiency was first encountered in 1893 and it is estimated that 0.5 million ha of farmland is Co deficient and another 2 million ha has latent or incipient Co deficiency in both the North and South Islands (Fraser 1984). Deficiencies sometimes occur singly, but quite often multiple deficiencies occur within a soil type (e.g., Co and selenium (Se) in North Island pumice soils (Watkinson 1992)). Low trace element availability manifests in both animal health problems (Co, copper (Cu), iron (Fe), iodine (I), manganese (Mn), Se, and zinc (Zn)) and sub-optimal plant growth (boron (B), Cu, Fe, Mn, Mo, and Zn). Methods of preventing deficiencies have been developed for

Table 10 NLE-modelled nitrate-N leaching losses in an organic compared with a conventional New Zealand dairy farm. Simulations are made for two artificially constituted scenarios. Annual drainage is assumed to be 300 mm.

	Conventional		Organic	
Scenario	1	2	1	2
Stocking rate (s.u. ha^{-1})	2	3	1	1.5
Urea applied (kg N ha^{-1} yr^{-1})	100	200	0	0
Dairy effluent applied (kg N ha^{-1} yr^{-1})	10	20	5	7.5
Biological N fixation (kg N ha^{-1} yr^{-1})				
urine patch area	10	10	10	10
non-urine area	150	100	240	220
N mineralised (kg N ha^{-1} yr^{-1})	150	200	150	180
Pasture N uptake (kg N ha^{-1} yr^{-1})				
urine patch area	600	600	600	600
non-urine patch area	250	300	200	230
Proportional urine area (%)	20	30	10	15
Modelled leaching losses (kg N ha^{-1} yr^{-1})	**19**	**46**	**9**	**12**

animals, and Grace (1994) has documented current recommendations for Se, Co, Cu, and I in grazing livestock, while Sherrell (1983a, 1983b, 1983c) has made recommendations for B in grasses and pasture legumes, and Mo in pasture legumes (Sherrell 1984). Brassica plants are very sensitive to Mo deficiency (During 1984). For the successful organic farmer a knowledge of the trace element status of the soil and the requirements of crops and animals is essential.

Over the years agricultural scientists have evaluated different ways of preventing trace element deficiencies and developed protocols that recognise the effects of the following factors that are equally relevant to organic systems:

- decreasing plant availability of added trace elements over time due to reaction with soil colloids (Khan et al. 1999);
- the effect of lime on soil pH and trace element availability (During 1984);
- seasonal fluctuations in trace element content of pasture (Cornforth 1984);
- sward composition (McNaught 1938);
- diet selection and digestibility (Rattray & Clark 1984);
- soil temperature effects (Tiller 1988);
- type and maturity of the animal (Grace 1994);
- interactions with other elements in the digestive system (Phillippo et al. 1987);
- soil ingestion by the animal (Andrews et al. 1958), although the extensive grazing approach in organic farming should reduce the effect of this factor.

This body of work has led to trace elements being administered in conventional systems in a number of ways. Selenium can be given to sheep and cattle in a drench, or as a vaccine, at intervals of between 1 and 6 months. Topdressing pasture with 10 g Se ha^{-1} is also an effective method of preventing Se deficiency. Cobalt can be administered to the animal as either an injection of vitamin B_{12}, or as a Co drench. However, topdressing pasture with 350 g ha^{-1} cobalt sulphate ($CoSO_4.7H_2O$) is still considered the most effective way of preventing Co deficiency in grazing animals. Copper can also be administered to animals either in a drench or as an injection. Copper sulphate may be applied to pasture or lucerne, but the effectiveness of this approach is dependent on the level of Mo and S in the animal's diet. Iodine, on the other hand, is effective only when administered directly to the animal; topdressing of pasture is not recommended (Grace 1994), although research is on-going (Smith et al. 1999).

Bio-Gro regulations do not prevent the administration of trace element supplements where they are required, but only "permitted, or Bio-Gro New Zealand approved restricted" materials may be administered at specified rates. The actual rates are not mentioned in the regulations but "heavy metals" (presumably including essential trace elements) must not exceed specified levels in the soil, manures, and composts. If organic farmers are to avoid using drenches or injections, and reject topdressing pasture with soluble salts, then their alternatives are applying organic fertilisers, such as animal manure and seaweed, modifying the pasture composition to include species rich in trace elements, or administering homoeopathic remedies to animals.

Verkleij (1992) reviewed the use of seaweed extracts in agriculture and horticulture, quoting the trace element content of seaweeds as: B (15–910 mg kg^{-1}), Cu (0.6–80 mg kg^{-1}), Fe (90–1500 mg kg^{-1}), Mn (4–240 mg kg^{-1}), and Zn (2–680 mg kg^{-1}). Seaweed extracts are sprayed directly onto the leaves. The trace elements present are in an organic form (mostly chelated) rather than the inorganic form used in conventional fertilisers that are applied to the soil. Verkleij (1992) concluded that because elements such as Fe, Mn, and Zn are in chelated form their effectiveness is greater. Recommendations for seaweed-based products suggest spraying on the crop several times a growing season to increase their effect. A complicating factor in assessing the effectiveness of applying trace elements in seaweed extracts is the presence of plant hormones in the product. Verkleij (1992) also concluded that the ability of crops to respond to seaweed application seems to depend on the type of soil, on the type of crop, and on its growth stage.

There is little research information on the availability of trace elements in animal manures or seaweed to pasture plants in New Zealand organic farms. However, the effective lifetime of organic fertilisers as soil supplements is likely to be an important issue.

The organic farmer may selectively grow plant species or cultivars to counter low trace element availability in conventional grazing species. A 10-year study by Sherrell et al. (1990) found that lucerne grown on pumice soil had a long-term average Co content only half that of white clover. However, there is evidence that other deep-rooting plants have higher levels of trace elements than ryegrass and white clover (*Trifolium repens*). Forage herbs, such as chicory (*Cichorium intybus*) and ribwort plantain (*Plantago lanceolata*) that are higher in Cu and Co than grasses (Mitchell 1945; Younie & Umrani 1997), may be used as herbal strips in a paddock to enhance the animals' intake of trace elements. Alternatively, strategic supplementary grazing on pure species crops may be appropriate at certain times of the year (Waugh et al. 1998).

Potentially, trace element toxicities in animals as well as deficiencies may occur in New Zealand agriculture. Grace (1994) has discussed the sensitivity of young animals to Se, while Smith (1973) reported on the effects of high Mo in pasture on the copper status of livestock on Northland soils.

Trace element deficiencies occur in the human population in New Zealand as well as in livestock. Deficiencies of I, Fe, Se, and Zn may occur in the young as a result of deficiencies in the diet. An organic farm that is well managed in relation to trace element supply has the potential to produce plant and animal products that are appropriate for the human diet.

Heavy metal contamination

In recent years, the uptake of cadmium (Cd) by plants in conventional agricultural systems has been the centre of intensive investigation by New Zealand agricultural scientists. This has been triggered by concerns about its presence in export meat products. Most of the Cd that has entered the farming system in New Zealand has been introduced as a component of the fertiliser, superphosphate. Roberts et al. (1994) reported a range of 0.04–1.53 mg Cd kg^{-1} soil for New Zealand pastoral soils with a history of superphosphate topdressing. As a consequence of these findings and elevated Cd levels in meat products, rock phosphates with lower Cd content (Baechle & Wolstein 1984; Loganathan et al. 1995) are now being used by the New Zealand fertiliser industry in an effort to minimise further contamination of farmland. Many countries have introduced regulations to limit the Cd content of P fertilisers (Mortvedt 1996).

Bio-Gro production standards specify an upper limit of 2 mg Cd kg^{-1} soil (Bio-Gro New Zealand 1998) as background contamination for organic farms. Meeting this requirement should not be a major obstacle for conversion to organic farming. All 193 sites surveyed by Roberts et al. (1994) on 8 soil types would have met the Bio-Gro standard.

As already described, organic farms may use RPRs as a source of P. Low Cd RPR such as Jordan PR has 5 mg Cd kg^{-1}, while Sechura PR has a Cd content of 9–12 mg Cd kg^{-1}. Loganathan et al. (1997) found that Jordan PR applied at a rate of 30 kg P ha^{-1} yr^{-1} did not have a significant effect on the Cd content of grass even after 10 years of annual application. Bolan (1999) calculated that at an application rate of 20 kg P ha^{-1} Sechura PR would take 800 to 1000 years to raise the soil Cd level to the maximum Bio-Gro limit. If more PR is used to maintain sufficient plant-available P in some organic systems, then this time period will be shorter. In comparison, Loganathan et al. (1997) observed a seasonal effect in the Cd content of the grass that was far more significant and that was attributed mainly to changes in growth rate. Tiller (1988) previously reported that an increase in soil temperature from 10°C to 18°C gives rise to a fourfold increase in Cd uptake by subterranean clover (*Trifolium subterraneum*).

A second, more demanding, Bio-Gro quality standard requires that the food produced on an organic farm must not exceed 0.1 mg Cd kg^{-1} (Bio-Gro New Zealand 1998). Roberts et al. (1994) reported that 22–28% of sheep and 14–20% of cattle between 1988 and 1991 had kidney Cd contents greater than the maximum permissible level of 1 mg Cd kg^{-1} set by the New Zealand Department of Health for meat and offal for human consumption. The culmination of this research effort has been the development of a computer model by Loganathan et al. (1999) to predict kidney and liver Cd concentrations in grazing animals in conventional farming. Its applicability to organic systems has yet to be demonstrated.

In organic systems, plants grown to ensure that animals have an adequate intake of essential trace elements may have to be screened for their Cd content. Some weed species have an ability to selectively accumulate Cd (Stünzi 1998). Roberts et al. (1994) reported that the elevated Cd levels in pastoral soils were reflected in the Cd content of weed species, to a lesser extent for grass species, and not at all in legume species. The main weeds in the Roberts et al. (1994) study were plantains (*Plantago* spp.), docks (*Rumex* spp.), buttercup (*Ranunculus* spp.), and dandelion (*Taraxacum* spp).

Gray et al. (1999a) reported that increasing the soil pH from 5.5 to 7.0 significantly decreased Cd concentrations in clover, ryegrass, and, to a lesser extent, wheat (*Triticum aestivum*). The extent of any increase in phytoavailability of Cd to these plants with a reduction in soil pH in New Zealand soils is not known. However, as a result of applying elemental S and reducing soil pH from 6.0 to 3.5 the Cd content in the shoots of mustard (*Sinapis alba* cv. Jara) was increased (Tichy et al. 1997).

The leaching of Cd into groundwater is not an issue within the normal soil pH range of 5.5 to 7.0. Loganathan & Hedley (1997) found that under pasture, after 10 years of topdressing, no significant amounts of fertiliser Cd moved below 120 mm. Some movement of Cd down the soil profile under irrigation was inferred by Gray et al. (1999b). The Cd in the 0–30-mm depth is less plant-available because of the greater organic matter content and higher pH of the soil. In a study of the amount of Cd desorbed from 29 New Zealand soils, Gray et al. (1999c) found that 85% of the variation was related to pH, organic matter, and total Cd. It can be concluded that Cd is not as strongly bound in the soil as Cu, but its movement down the profile is restricted if the soil has adequate organic matter and a near neutral pH.

In arable farming, Gray & McLaren (1999) and Roberts et al. (1995) found considerable variation in Cd concentrations in New Zealand wheat grain samples. Part of this variation was attributed to cultivar differences and partly to the Cd content of the soil. Roberts et al. (1995) reported an average Cd concentration of 70 $\mu g\ kg^{-1}$ fresh weight (FW) for wheat grown in mid Canterbury, while Gray & McLaren (1999) reported a mean of 54 $\mu g\ kg^{-1}$ FW from their survey of 33 winter and 18 spring cultivars grown throughout New Zealand's arable regions. None of these cultivars was the product of genetic engineering, so the results of these studies are relevant to the choices to be made in organic farming. Gray et al. (1999d) also found large differences in Cd content between species such as maize (*Zea mays*), barley (*Hordeum vulgare*), white clover, lucerne (*Medicago sativa*), and ryegrass grown on 10 different New Zealand soils. However, detailed survey results, similar to those carried out for wheat, are not available in the literature and the filling of this gap in our knowledge would aid our understanding of Cd availability in organic systems in New Zealand.

PESTICIDES

Conventional farming

It is estimated that currently about 3300 t of active ingredients are used in agricultural, horticultural, and forestry production systems in New Zealand (Holland & Rahman 1999). These pesticides comprise about 68% herbicides, 24% fungicides, and 8% insecticides. About 34% of the pesticides are used for horticultural purposes (e.g., orchards, vegetables), 25% for pastoral agriculture, 16% for cropping systems, and 25% for forestry. Pesticide use in pastoral agriculture is mainly for weed control. Herbicides are also the main forms of pesticides in cereal and arable crops. The highest pesticide use occurs in orchard crops and fresh vegetables, e.g., apples, lettuce, brassicas, potatoes, and onions. Cereal and herbage seed production also makes extensive use of pesticides.

While the use of pesticides is often economically justifiable, with $3–4 returns in reduced crop losses for every NZ$1 spent on pesticides (Holland & Rahman 1999), there has been increasing public concern on adverse environmental and health effects. As pesticides are toxic by design, when released into the environment they may not only kill the target organism but also adversely affect other beneficial organisms living in the vicinity. Pesticides may also dissipate into other ecosystems, e.g., air, surface water, and groundwater (Close 1996). Pesticide residues in food for human consumption are also of major public concern. The environmental impacts of the persistent organochlorine insecticides (e.g., DDT) which were used in the 1950s and 1960s have been widely recognised and their use was stopped in the 1970s (Ministry for the Environment 1998; Cameron et al. 2000). The effects of other pesticides on soil organisms, wildlife, and aquatic life have also been recorded (e.g., Madhun & Freed 1990). Residues of a range of pesticides have been detected in diet surveys (e.g., Vannoort et al. 1995). While possible impacts on human health by long-term exposure to low concentrations of pesticides in the environment and in food are still a topic of intense public debate and scientific research, health authorities in New Zealand and overseas have developed safety guidelines to limit residue concentrations in drinking water and in food.

Organic farming

In organic farming systems, the problems associated with the pesticide residues in the environment as discussed above can be hugely reduced, because pest and disease control is achieved not by relying on synthetic pesticides, but by a range of alternative integrated management practices (Table 11) (Bio-Gro New Zealand 1998). The virtual absence of pesticide use allows more plants and invertebrates to flourish (van Elsen 2000). Soil micro-flora and -fauna are also enhanced by organic practices (Liebig & Doran 1999). Therefore, the better performance in many of the soil biological quality indicators in the organic system compared with the conventional system, as reviewed in the sections above, may be partly attributable to reduced effects of pesticides in organic systems. A recent study in New Zealand found that organically farmed carrot fields had significantly higher numbers of beneficial arthropods compared with conventionally managed fields (Berry et al. 1996). Similar findings have been reported in other countries (e.g., Moreby et al. 1994). The lack of pesticides in the organic production system will obviously reduce the effect on surrounding environments, e.g., air and water, and will reduce residue concentrations in food.

Over the past decade, there has been a significant increase in organic production in New Zealand, particularly in kiwifruit, honey, vegetables, and apples. For example, more than 500 ha of kiwifruit were organic in 1999, representing 7% of the total kiwifruit area. However, the organic conversions so far have only had a small effect on total pesticide use nationally, although the reductions are significant in kiwifruit (Holland & Rahman 1999).

It must be noted that some of the chemicals permitted for use in organic systems may also have adverse effects on the environment (Table 11). For example, Cu compounds used for disease control can have a negative effect on soil organisms if used at high concentrations. Rotenone is toxic to fish and pyrethrum will kill beneficial organisms in addition to its target (Conacher & Conacher 1998). The fine dust of diatomaceous earth is a lung irritant and needs to be handled with care. The use of genetically modified pest-resistant crops is also controversial and their environmental effects are largely unknown at this stage.

Farmers considering a conversion from conventional to organic farming should also be aware that parts of New Zealand are contaminated with pesticides from past use. For example, DDT residue concentrations are still high in some Canterbury soils as a result of DDT use in the 1950s and 1960s. The presence of pesticide residues in the produce at significant concentrations may complicate the certification process. Spray-drift from neighbouring conventional farms also has the potential to contaminate the soil and produce on organic farms. Studies in New Zealand kiwifruit orchards have shown that, on average, 79% of the pesticides sprayed end up on target crops, 19% end up on the ground or shelter, and about 2% leaves the orchard as drift (Holland et al. 1997). The distance for 50% decline in pesticide deposition from drift was about 26 m. Drift losses are particularly high for volatile pesticides and on farms without good shelter. In addition, if imported manures or composts are used on organic farms, it is important to ensure that industrial wastes do not contaminate these organic wastes which may contain persistent organochlorine pesticides from past uses.

Weed and pest control is sometimes achieved by frequent cultivation of the soil. This practice may

Table 11 Management measures or materials for pest and disease control in organic farming systems which are accepted by Bio-Gro New Zealand (Bio-Gro New Zealand 1998).

Approved management measures or materials
Biological controls (parasites, predators, or disease organisms)
Diatomaceous earth
Gas saturation using argon carbon dioxide or nitrogen
Approved forms of herbal sprays, homoeopathic preparations
Hydrogen peroxide
Mechanical controls (traps, barriers, lures)
Natural purgatives (e.g., salt water)
Pheromones (but not directly on plants)
Approved forms of soft soaps (potassium-based soaps)
Sulphur burning
Approved forms of vegetable oils
Waterglass (sodium silicate)
Citric acid (for water stain)
Bentonite
Restricted materials and practices (may be used but only in accordance with the principles specified in the Standards,with progressive annual reductions in their use, until alternatives are approved)
Bordeaux, Burgundy mixtures
Copper hydroxide
Lime sulphur
Sulphur preparations
Potassium permanganate
Borax
Bicarbonate of soda
Mineral oils
Rodenticides (in covered containers only): Brodifacoum, Bromadiolene, Warfarin
Quassia
Ryania
Pyrethrum cinerafolium
Rotenone (derris)
Neem
Stockholm tar

also have implications on soil structure as cultivation can damage soil aggregates. However, the use of greater quantities of organic and/or green manures might offset some of this effect, as soil organic matter from the manure would be beneficial for soil structural stability. It is also possible that more frequent cultivation under organic farming may increase soil organic N mineralisation and N loss by leaching, but this is as yet unknown. In conclusion, effective pest and weed control is probably the biggest challenge for economical organic farming. There is a need for more research on integrated approaches to farm management for pest control with minimum use of synthetic pesticides. It is also important to ensure that these alternative pest management practices do not have other adverse effects on soil and environmental quality.

YIELD AND PRODUCTION

Recognising that the focus of this review is soil and environmental quality, we have not included detailed coverage of yield and production effects. Organic production is initially about quality, but yield and production levels are not insignificant issues in determining economic viability of organic and biodynamic farming, and key issues in the literature are therefore included for completeness.

For example, O'Connor et al. (1990) and Lambert et al. (1990) clearly demonstrated that on hill-country soils in the North Island of New Zealand pasture and animal production declined from the second year following the cessation of superphosphate inputs. This has negative implications for conversion to biodynamic systems.

Average crop yields of two organic systems were markedly lower (17% less) than conventional systems (Mäder et al. 1999), which was mainly attributed to the lower nutrient input and lower plant protection achieved. Generally, differences in crop yield between organic and conventional systems were larger for crops with a relatively short vegetation period, such as potatoes, than for crops with a relatively long vegetation period, such as cereals and grass clover. Newton (1995) reported data taken from Lampkin (1990) and Harper (1983) that showed a lower yield for wheat, barley, oats, and potatoes on organic farms than on conventional farms. However, different crops respond in different ways to organic fertilisers (Kell 1992).

At a financial level, Reganold et al. (1993) found that biodynamic farms had lower total gross margins than did their conventional neighbours, although most of the difference was put down to the smaller size and greater enterprise diversity of the biodynamic farms.

In summary, it is generally accepted that on conversion to organic farming, yield and production may decline, but there are more recent data relating to dairy farms from the Institute of Grassland and Environmental Research (1998) in the United Kingdom showing that an initial drop in production can be followed by an improvement after three years of organic farming.

CONCLUSIONS

There is scientific evidence that organic farming can improve soil biological properties and soil physical properties related to the build-up of organic matter. However, there is insufficient information and knowledge to draw more comprehensive conclusions about the effects of organic, relative to conventional, farming systems on soil and environmental quality in New Zealand. The research effort overseas, especially in Europe, will provide much useful guidance, but the unique character of New Zealand's soils, climate, and farming systems will necessitate on-site investigations.

Organic farming under a well-managed Bio-Gro organic system with an adequate supply of nutrients is probably sustainable over the long-term, but there are concerns about the supplies of N and of trace elements. There needs to be a greater research effort to establish whether these nutrients can be supplied in adequate amounts under organic farming without depleting the soil reserves. Biodynamic farming appears to be less sustainable than Bio-Gro or conventional farming, because nutrients removed from the soil are not being adequately replaced.

The continued development of effective long-term nutrient management regimes for organic (and indeed conventional) farming systems in New Zealand that deliver optimum levels of plant production, is dependent upon appropriate assessment of nutrient input requirements. This, in turn, raises the issue of how to assess soil nutrient status. Many conventional soil-testing procedures, and their interpretation with regard to fertiliser recommendations, may be inappropriate for use in organic farming given obvious differences in the nature of fertiliser materials allowed under organic compared with conventional farming. Clearly, further work is required to consider and evaluate soil testing procedures which are appropriate for use in determining the particular fertiliser needs of various organic farming systems in New Zealand.

The conclusions reached here relate to soil and environmental quality and we recognise that there are significant production, economic, and social issues relating to the choice of production system that are beyond the scope of this paper. Within our field of expertise, we have concluded that research is urgently required to provide the necessary information that will assist farmers in making informed choices about possible conversion to organic farming, enable farm advisers and industry personnel to provide correct advice, assist policy makers with legislation, and provide a defence of New Zealand's "clean green" marketing strategy.

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