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LCA FOR FOOD PRODUCTS

Assessing agricultural soil acidification and nutrient management in life cycle assessment

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Abstract
Purpose This paper describes part of the first detailed environmental life cycle assessment (LCA) of Australian red meat (beef and sheep meat) production. The study was intended to assist the methodological development of life cycle impact assessment by examining the feasibility of new indicators for natural resource management (NRM) issues relevant to soil management in agricultural LCA. This paper is intended to describe the NRM indicators directly related to agricultural soil chemistry.
Materials and methods Three nutrient management indicators—nitrogen (N), phosphorus (P) and potassium (K) balances—were estimated on the basis of 1 kg of hot standard carcass weight (HSCW) for three grazing properties in Australia. We also examined a soil acidification indicator based on the effects of agricultural practices.
Results and discussion The N balance for the grazing properties varied from a loss of 28 g N/kg HSCW to an accumulation of 170 g N/kg HSCW. For comparison, the N content of cattle is about 24 g/kg liveweight. The main contributors to these changes were the growth of N-fixing pastures (or lack thereof) and the application of fertilisers. The P and the K balances showed similar results, varying from a 3.9-g loss to a 19-g accumulation of P and a 4-g loss to a 95-g accumulation of K per kilogram HSCW. Decisions about pasture management were also reflected in the results of the soil acidification indicator. We also identified that soil erosion at the grazing properties is a significant component of nutrient losses.
Conclusions The results suggest that reducing the leaching of soil N might be the best way to balance the N budget without causing acidification. The NRM indicators developed can be benchmarked against other production systems as the application of these indicators progresses.

Keywords LCA · Nutrient management · Red meat · Soil acidification potential

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1 Introduction

Australia is the second largest exporter of beef and sheep meat in the world (MLA 2010). The red meat industry, like many other primary industries, is under increasing pressure to document and justify its impact on the environment. In this context, the importance of quantitative, strategic environmental data in general and life cycle assessment (LCA) for meat products in particular is recognised worldwide with LCAs having been conducted on pork

(Basset-Mens and van der Werf 2005; Eriksson 2005), lamb (Schlich and Fleissner 2005; Williams et al. 2006) and beef (Cederberg and Stadig 2003; Ogino et al. 2007; Verge et al. 2008). Related LCAs have examined leather (Mila i Canals et al. 2002) and dairy products (Eide 2002; Hospido et al. 2003; Lundie et al. 2003; Casey and Holden 2005).

To optimise environmental outcomes and target management interventions, managers and policy makers need performance information which takes a holistic life cycle perspective and is based on best practice data acquisition and analysis. Great attention is currently drawn to greenhouse gas emissions ("carbon footprinting"), energy consumption and water use in the meat industry, and further research is focussed on potential environmental issues related to chemicals (Khan et al. 2008). On the other hand, issues of natural resource management (NRM) related to the loss of nutrients in soils and soil acidification are of increasing concern for actors within the red meat industry in Australia.

NRM matters are typically not well represented in agricultural LCAs, and the standard suite of LCA impact categories is not designed to reflect local impacts. A majority of the LCAs listed above have considered "acidification potential". Yet consistent with historical concerns about acid rain and damage to both the built environment and ecosystems like the Black Forest, they have taken the perspective of estimating the potential for airborne emissions to affect downwind ecosystems. However, while this kind of acidification effect is not significant in all environments, acidification due to agricultural practices is recognised as a major degradation issue worldwide, and business as usual in Australia was predicted to cause between 14 and 39 million hectares of agricultural land to fall below pH 5.5 over 10 years (NLWRA 2001). This is an aspect of current LCA practice that limits its relevance to land managers concerned with depletion of the principal resource they manage—agricultural soil. Other LCA indicators may be of greater interest to them if LCIA can be extended to also answer local questions of significance to these stakeholders. In this study, an acid/base balance model and nutrient balances are used to evaluate soil acidification under different red meat production systems.

2 Materials and methods

2.1 System description and indicators

The red meat production systems under consideration are (1) a sheep meat supply chain in Western Australia (WA) which also produces grain and wool, (2) an organic beef supply chain in Victoria (VIC) and (3) a beef/sheep meat supply chain in New South Wales (NSW) which also produces grain on irrigated land. To reflect a wide range of prevailing weather conditions, data were collected for the 2002 (which was relatively dry) and 2004 (which had more typical rainfall) calendar years for each supply chain. The functional unit is the delivery of 1 kg of hot standard carcass weight (HSCW) at the exit of the meat processing works. More detail on the farm models and LCA results for more typical indicators including energy consumption, carbon footprint and water use have been described earlier (Peters et al. 2010a, b).

This paper focuses on nutrient management (nitrogen (N), phosphorus (P) and potassium (K)) and soil acidification potential indicators. All of these are specifically intended for application at the farm site for soil management, so in these cases we do not assess elemental flows that do not connect with the site. This focus reflects the reality that agriculturalists want tools that help them manage their own natural resources in the first place. For example, K flows in power station wastes are unlikely to have a significant influence on farm soil characteristics unless they are deliberately included in soil amendments (in which case they would be included in this analysis). Therefore developing an LCA approach within this limited context is a step forward for the agricultural system because it can allow predictive management of the manager's natural resources, to complement current retrospective methods based on soil sampling. This can be used in decision-making to compliment other LCA indicators such as those previously published in this study, including water use (Peters et al. 2010a) and greenhouse emissions (Peters et al. 2010b). Nutrient management and soil acidification indicators could in principle be expanded to include other agricultural activities within the background system (e.g. grain production offsite) but this is beyond the scope of this study and beyond the management control of most graziers dealing with retail intermediaries for such supplies.

For each grazing property, we used mass balance principles to estimate annual N, P and K inputs (incoming livestock, fertiliser, feedstuffs and other nutrient inputs) and outputs (outgoing livestock, wool, harvested product (e.g. hay, grain) and environmental losses). Input and output masses were, where possible, calculated from the farm records kept for each property for each year. Inputs and outputs less readily quantified in situ (N fixation by legume pastures, N leaching through the soil profile) were estimated based on literature data.

The sign of all the balances was aligned with the idea of negative life cycle impact indicators in general use in LCA. Thus just as a larger global warming potential was worse for the atmosphere, a larger acidification potential was worse for the soil. This in turn was driven by N losses, so we arbitrarily assigned a positive sign to N losses from the property. Of course, on a particular farm where excess N was present in soils, the last step in this logic may be paradoxical.

2.2 Livestock nutrient flows

The nutrient flows in incoming and outgoing cattle were estimated using the BEEF-BAL programme (DPI&F 2003). We used a generic chemical composition of livestock, introducing two possible error sources: (1) differences in the body composition of store animals (with a low body-fat content) compared to finished animals (typically with a higher body-fat content), and (2) variations in body composition between animals of different sexes and ages. Since this research was performed at the property scale, these errors are likely to be small compared to errors elsewhere in the nutrient balance.

Data for the chemical composition of sheep (CNMSP 2007) varied less than 5% from the beef cattle estimate so we assumed they were equivalent. We assumed that clean wool (70% of greasy wool weight) is 100% protein and therefore contains approximately 16% N, with negligible P and K.

2.3 Other nutrient inputs

Chemical analyses for synthetic fertilisers (Incitec 2005a, b, c) and organic fertilisers and soil amendments (Nutri-tech 2006) were collected from manufacturers. Nutrient inputs in feed were estimated by combining feeding data with standard figures for dry matter content and nutrient composition.

N fixation is the most significant non-traded nutrient input. It was not possible to directly assess the mass of N fixed by legumes on the supply chain properties so we relied on literature data for N fixation by legume pastures in Australia. The reported rate of N fixation varies widely (Sanford et al. 1995; Unkovich et al. 1997; Dear et al. 1999; Riffkin et al. 1999; McKenzie et al. 2003). It is influenced by many factors, primarily the legume species, pasture growth rate and the percentage of legume in the pasture sward. These factors were investigated for each supply chain and matched as closely as possible to the reported conditions in the available literature. Actual annual N fixed could vary from 10–190 kg N/ha/year or more. We selected an intermediate figure: clover-based pastures fixed 92 kg N/ha/year under normal rainfall or 46 kg N/ha/year when rainfall was limited in the survey year (Unkovich et al. 1997).

The parameters used to calculate the nutrient inputs for the farms in the three supply chains are summarised in Table 1. Data on N, P and K represent averages from the literature shown in the table.

2.4 Other nutrient outputs

Nutrient outputs have two main forms: export of produce and losses to the environment. Exports associated with animals for slaughter were estimated using the BEEF-BAL model described above. Where wool was produced, exports could be accounted for directly, but environmental losses were allocated on the basis of economic value as explained in Peters et al. (2010a, b). Nutrient loss is a major environmental concern, particularly with respect to N and P transfer to surface and groundwater resources. We estimated suspended particulate erosion, dissolved nutrient losses in overland flows, leaching and gaseous loses. Other nutrient loss pathways (e.g. fire) were considered minor in these southern meat supply chains and were excluded.

Particulate nutrient losses are the product of the soil erosion rate and the soil nutrient content. We used broad scale erosion estimates from the National Land and Water Resources Audit which relate only to waterborne erosion (NLWRA 2001). These estimates took into account the erosion gully density and the annual hillslope erosion rate and were enhanced by a qualitative assessment of site specific factors (i.e. livestock trampling or reduction in vegetative cover) on each property, as described further in Peters et al. (2010a, b). Wind erosion may also be a factor but consistent data were unavailable on this (NLWRA 2001). Total soil nutrient levels were estimated based on property management and production. The nutrient content of the eroded fraction of soil can differ from the nutrient content of the in situ soil, but no data on this were available so we assumed that they were equal.

Dissolved nutrient losses in runoff have been researched at the property scale by only five studies in Australia (Barlow et al. 2005) and published estimates vary widely (Table S1 in the Electronic Supplementary Material). Dissolved nutrient losses in runoff are influenced by soil fertility and the nutrient concentration at the soil surface. Intensively grazed areas have a higher nutrient turnover and higher deposition of manure on the soil surface and hence higher nutrient levels in runoff. We matched the systems under study as closely as possible to the published studies and conservatively estimated these losses. At the VIC site, we estimated losses of 3 kg N/ha/year (Ridley et al. 2003) and 2.5 kg P/ha/year (Barlow et al. 2005) for *intensively grazed areas*. At the NSW site, we estimated losses of 3 kg N/ha/year and 0.45 kg P/ha/year because of the lower intensity of the grazing system (Ridley et al. 2003). At the WA site, we estimated losses of 0.2 kg N/ha/year (Ridley et al. 2003) and 0.1 kg P/ha/year (Costin 1980) in response to the low soil fertility and lower grazing intensity at this site. Nutrient losses from *cropped areas* were also relevant at the NSW (6 kg N/ha/year and 2 kg P/ha/year) and WA (0.2 kg N/ha/year and 0.1 kg P/ha/year) properties.

Nutrient losses due to leaching can lead to groundwater contamination and soil acidification. The greatest concern is usually associated with N leaching as nitrate (NO_3^-), although K is also mobile in the soil solution. We were unable to estimate K losses due to limited data. We estimated nitrate leaching using published data, taking care

Table 1 Summary of nutrient input data

Nutrient inputs	Description	Value	Reference
Livestock	Sheep/cattle	N=2.4% of liveweight P=0.7% of liveweight K=0.2% of liveweight	QDPI&F (2005)
Fertiliser	Urea $(NH_4)_3PO_4$ $(NH_4)_2HPO_4$ Single superphosphate Pivot 15 Organic and K humates Seachange kelp mix Muriate of potash $(NH_4)_2SO_4$, K_2SO_4	Dependant on fertiliser type—taken from manufacturers' reported nutrient levels	Incitec (2005a); Incitec 2005b; 2005c
Feed and feed supplements	Pasture hay Legume hay Lupins Canola meal Canola oil Minerals	N/P/K=1.3:0.4:0.2 (%) N/P/K=2.1:0.4:1.0 (%) N/P/K=4.6:0.3:0.8 (%) N/P/K=5.8:1.0:1.2 (%) N/P/K=0.0:0.3:0.4 (%) Dependant on type	QDPI&F (2005)
Legume N fixation	Mixed clover/grass pastures	46 kg/ha or 92 kg/ha Estimate dependant on density of legume within the sward and the annual rainfall in the survey year	Sandford et al. (1995) Unkovich et al. (1997) Dear et al. (1999) Riffkin et al. (1999) McKenzie et al. (2003)

to match the grazing properties to the literature conditions as these data are highly variable (3.7 to 82.0 kg N/ha/year) depending on experimental technique and location effects (Ridley et al. 1990; Eckard et al. 2004). Factors we considered include the soil type, annual rainfall and rainfall pattern, pasture production, N inputs to the system and other indicators of nitrate leaching (e.g. accelerated soil acidification). We also compared the leaching rate with the overall nutrient balance to check our assumptions. Even a small difference in nitrate leaching may significantly affect other parameters, particularly soil acidification.

N can also be lost from agricultural systems through volatilisation and denitrification (NH_3, N_2, N_2O). Based on published data, total gaseous N losses from the three systems were estimated at 20–27 kg/ha/year depending on the level of N input to the system (Dalal et al. 2003; Eckard et al. 2003). Assessment of losses from the WA and NSW systems was relatively difficult as they have been researched less than VIC systems. In the absence of other data we assumed that N_2 emissions were equal for all properties. Because of the drier climate in the NSW and WA systems, we assumed lower N_2O losses (0.2 kg/ha/year after Dalal et al. (2003)). These may be overestimates depending on specific management practices on farm, and are included in the LCI for indicative purposes only.

The parameters used to calculate the nutrient outputs for the farms in the three supply chains are summarised in Table 2. Data are presented as a range covering the three properties and two years of data collection.

2.5 Soil acidification potential

To estimate the soil acidification potential we used an acid/base balance model. We used published data for the acidification rates resulting from product removal, N fertiliser acidification and grazed pasture acidification potential on improved or extensively managed pastures. The model took account of the annual rainfall, soil type and the likely leaching potential, proportion of legume pasture, estimated annual dry matter production and grazing management system.

The drivers of acidification considered are (1) use of N fertilisers, (2) movement of manure within the system by grazing animals, (3) use of legume-based pastures, (4) removal of agricultural products resulting in a net export of alkalinity and (5) management that promotes a build up of soil organic matter. Soil organic matter can act as an acidifying substrate and as a buffer against pH change, resulting in different effects over time and between soil types. These are balanced by the inputs of alkali into the system (livestock imports, some fertilisers, lime and other soil ameliorants).

Table 2 Summary of nutrient output data

Nutrient outputs	Description	Value	Reference
Export of produce	Sheep/cattle	N=2.4% of liveweight P=0.7% of liveweight K=0.2% of liveweight	QDPI&F (2005)
	Wool	N=16% of clean weight	
Nutrient loss with overland flow	Improved pasture	N=0.2–3.0 kg/ha/year P=0.1–2.5 kg/ha/year	See Supporting Information for a range of references
	Extensive pasture	N=0.1–0.3 kg/ha/year P=0.02–0.1 kg/ha/year	
	Cropping	N=0.2–6.0 kg/ha/year P=0.1–2.0 kg/ha/year	
Nutrient loss with soil erosion	Soil loss rates by sheet and rill erosion and, gully erosion (used as input for nutrient losses)	Sheet and rill erosion=0–5 t/ha/year Gully erosion=0–0.2 t/ha/year	(NLWRA 2001)
	Soil loss estimate multiplied by nutrient concentration within soil	N=0.0–0.1 kg/ha/year P=0.0–0.1 kg/ha/year	NLWRA (2001) and on farm soil analysis data
Leaching	N leaching dependant on soil type, rainfall and N mass balance	N=4–34 kg/ha/year (10–40% of N inputs from fertiliser and legumes)	Ridley et al. (1990) Anderson et al. (1998) Eckhard et al. (Ridley et al. 1990; Anderson et al. 1998; Eckard et al. 2004)
Gaseous losses	N_2, NO_x, NH_3	N=20–27 kg/ha/year	(Dalal et al. 2003; Eckard et al. 2003)

Transformation of N compounds in the soil occurs through a sequence of reactions, starting with conversion of organic N or urea to NH_4^+ through the process of ammonification (Eqs. 1a and 1b) (Bolan et al. 1991). This process results in the net consumption of H^+ ions (or release of OH^- ions) which increases pH.

$$RNH_2 + H_2O + H^+ \rightarrow R - OH + NH_4^+ \tag{1a}$$

$$(NH_2)_2CO + 3H_2O \rightarrow 2NH_4^+ + 2OH^- + CO_2 \tag{1b}$$

The next process is the conversion of NH_4^+ to NO_3^- via the nitrification processes (Eq. 2). The combined ammonification–nitrification process for organic N leads to a net increase of one mole of H^+ for every mole of N transformed. Oxidation of ammonia-based fertilisers can generate two net moles of H^+ per mole of N.

$$NH_4^+ + 2O_2 \rightarrow NO_3^- + H_2O + 2H^+ \tag{2}$$

$$2NO_3^- + 2H^+ \rightarrow N_2 + 2.5O_2 + H_2O \tag{3a}$$

$$2NH_4^+ + 2O_2 \rightarrow 2NO_2^- + 8H^+ + 6e^- \rightarrow N_2 + 4H_2O \tag{3b}$$

Provided that NO_3^- generated through the nitrification process (or conversion of NH_4^+ via nitrite (NO_2^-)) is completely converted to N_2 through the denitrification process (Eqs. 3a and 3b respectively), there is no net proton gain from the decomposition of organic N (Helyar 1976; Bolan et al. 1991). However, this is rarely the case in practice (Bolan et al. 1991), and N is lost from agricultural systems via several main pathways including product removal, volatilisation (Eq 4) and leaching of nitrate.

$$NH_4^+ + OH^- \rightarrow NH_{3(g)} + H_2O \tag{4}$$

When nitrate leaches beyond the root zone of plants with charge-balancing basic cations (Ca^{2+}, Mg^{2+}, K^+ or Na^+) instead of H^+ ions (generated from oxidation reactions, i.e. reactions 1a and 1b), there is, in the absence of counter-measures, a permanent net acidification effect in the soil. Nitrate leaching is one of the primary causes of soil acidification and is accounted for in both the nutrient management and fertiliser application components of the soil acidification model. For each kilomole of H^+ remaining in the soil following nitrate leaching it is assumed that 50 kg of $CaCO_3$ is required to neutralise the acidifying effect (Slattery et al. 1991).

The first driver of acidification considered in the model is the applications of fertiliser N and particularly

ammonium-based fertilisers (Bolan et al. 1991; Moody 2005). Acidification rates (expressed as the amount of lime required to neutralise the effect) resulting from the application of some N fertilisers are presented in Table 3.

These values are reported for Queensland, but the processes are expected to be similar in other regions. The major variable between regions affecting acidification is expected to be the amount of nitrate leached from the system. The model uses the leaching rate from the N balance. This was checked by calculating the kilomoles of N leached and the associated kilomoles of H^+ ions left in the soil matrix following the theory outlined by Bolan et al. (1991). This theory was also used to estimate the acidifying effect of N leaching under legume pastures.

The second driver of acidification considered in the model is the grazing behaviour of the stock. Uneven manure deposition across grazing areas is attributed to the typical behaviour patterns of grazing animals, which graze over a large proportion of a paddock but select a small area on which to rest. These resting areas or "stock camps" receive higher manure deposition, potentially resulting in a deposit of 34% of manure and urine (and therefore nutrients from the paddock) on the stock camp area (Hilder 1964, cited in Slattery et al. 1991). While the addition of anions in manure (particularly calcium) can result in decreased acidification on the stock camp area, the higher rate of N and organic matter addition increases acidification on stock camp areas (Cayley et al. 2002), and the result of anion transfer to stock camping areas is a "grazing effect" which acidifies the rest of the paddock where livestock graze.

The acidification potential as a result of grazing behaviour in sheep varies across regions from 9 to 25 kg $CaCO_3$/ha/year (Slattery et al. 1991; Cayley et al. 2002). These data were used in the model where relevant to the supply chains. No data were available for the effect of cattle grazing on acidification. However anecdotal evidence suggests that cattle also use stock camps in extensive grazing properties. Considering the lack of research for this effect with cattle production, the mean value from the sheep research referenced above was used where cattle displayed some camping behaviour on the supply chain properties. The VIC property was assumed to avoid the grazing effect as cell grazing (small fields) was the dominant pasture management practice. For the NSW and WA properties, a value of 10 kg $CaCO_3$/ha/year was selected as an approximation from the data presented in the literature.

The third driver of acidification considered in the model is nitrate leaching for legume pasture. Nitrate leaching below annual and perennial-legume-based pastures has been studied by several research groups (Ridley et al. 2001; Eckard et al. 2004). They reported a range of leaching rates for different pastures, soil types and rainfall patterns. The current research attempted to match the published leaching rates as closely as possible to the case study properties in the supply chains to determine the likely acidification potential.

The fourth driver of acidification considered in the model is product transfer. The transfer of agricultural products off-farm results in a net export of alkalinity that can cause acidification. The values quantified by Moody (2005), NLWRA (2001) and Slattery et al. (1991) which underpin the acid/base balance used in the LCI model are summarised in Table 4. Where hay or grain was produced on-farm and then fed to livestock, the acidification effect from removing this product was still measured. This is because internal transfer of alkalinity may still produce acidification of grazing property land despite the produce not leaving the property.

A fifth possible driver of acidification is soil organic matter. Increased soil organic matter concentrations are generally considered beneficial for soil health, structure and fertility. Increased soil organic matter levels under improved pastures can also contribute to soil acidification, though there is a high degree of variability in the literature as to the extent of this effect (Crawford et al. 1994; Cayley et al. 2002; Sandars et al. 2003). The acidifying process is driven by the dissociation of organic acids from the additional organic matter. Acidification may also result from additions of feedlot manure and effluent to pastures (Bouwman and Van Der Hoek 1997). Manure can have a variable effect depending on its nutrient and organic matter content (Schoenau 2005). Some authors report decreasing

Table 3 Lime needed to neutralise fertiliser impact vs. NO_3^- leaching (after Moody 2005)

Fertiliser	Lime requirement in kg $CaCO_3$/kg N applied		
	Percentage of N applied leached (as nitrate)		
	0%	50%	100%
Ammonium sulphate (AS)	3.6	5.4	7.1
Monoammonium phosphate (MAP)	3.6	5.4	7.1
Diammonium phosphate (DAP)	1.8	3.6	5.4
Nitram	0	1.8	3.6
Urea	0	1.8	3.6

Table 4 Alkalinity of exported agricultural produce

Product	$CaCO_3$ equivalent to 1000 kg product	$CaCO_3$ requirement for representative yields (kg/ha)	Reference
Wheat	9	18 (2 t/ha yield)	Slattery et al. (1991)
Barley	8	16 (2 t/ha yield)	Slattery et al. (1991)
Lupins	20	20 (1 t/ha yield)	Slattery et al. (1991)
Grass hay	25	125 (5 t/ha yield)	NLWRA (2001)
Grass hay	30	150 (5 t/ha yield)	Moody (2005)
Clover hay	40	200 (5 t/ha yield)	NLWRA (2001)
Lucerne hay	60	300 (5 t/ha yield)	Slattery et al. (1991)
Legume hay	50	250 (5 t/ha yield)	Moody (2005)
Sheep meat (liveweight)	17	6 (10×kg lambs)	Slattery et al. (1991)
Wool	14	0.6 (5 kg/sheep×8 sheep)	Slattery et al. (1991)

pH due to manure application (Chang et al. 1990), while others (e.g. Whalen et al. 2000) report an increase in pH following manure application on two acid soils under laboratory conditions. Owing to the lack of rigorous data to quantify acidification due to increased organic matter in isolation from the confounding interactions in pasture systems, it was assumed that this effect is smaller than other acidifying processes and could be omitted.

The overall impact of agricultural production on acidification was quantified using the indicator 1 kg of calcium carbonate ($CaCO_3$) equivalent. This indicator represents the amount of a substance, relative to calcium carbonate, required to neutralise the acidifying effect of a process. Compared to the SO_2-equivalent it is a more useful measure for agriculturalists attempting to balance soil acidity in the field. We note that neither indicator represents the actual change in pH that may be observed from a management practice. This is because a range of factors affect pH change, particularly the buffering capacity of the soil.

The parameters used to calculate soil acidification balances for the farms in the three supply chains are summarised in Table 5. Data are presented as a range covering the three properties and 2 years of data collection.

3 Results and discussion

These results should be read in conjunction with our previous reports on other indicators that work across the broader system (Peters et al. 2010a, b). As illustrated in Table 6, with the exception of the NSW grazing property in 2002, all the results for N were negative, suggesting that N accumulated in the grazing property soils in these years. The difference between the 2002 and 2004 results for the NSW grazing property was the result of our assumption that the rate of N fixation by subterranean clover was halved in 2002 because of the very dry conditions on that property in that year. This affected both the fixation and the leaching estimates on the other side of the ledger. The WA property had the highest N accumulation because of the relatively large inputs from both N fixation and fertiliser. In all six cases, the dominant N input was the result of N fixation by the clover-based pastures, with a smaller amount input to the property via fertilisers.

The largest N losses in the NSW and WA supply chains were the result of gaseous losses from pastures, while leaching losses were most important for the VIC property with its much higher rainfall and light soils. Because of the higher rainfall and sandy loam soils at the VIC site, leaching formed a significant component of the N losses at this site (30–34 kg/ha/year), compared to those for NSW and WA (4–15 kg/ha/year) where significantly lower rainfall was experienced in the years considered. For the NSW property, removals in livestock product exports, then losses in overland flow and erosion were next most important in 2002, a dry year. This was not the case in 2004, a wet year, when N losses through leaching were very important, followed by removals in livestock exports, with losses through overland flow and erosion still important. For the WA property, leaching losses were more important than removals through livestock product exports. For the VIC property, gaseous losses are the next most important factor after leaching losses, followed by removals through the export of livestock with minimal removal through overland flow and soil erosion.

These total results are exemplified in greater detail by the results for the NSW property shown in Fig. 1. The data in this figure are scaled to the maximum total N outflows in the 2004 NSW property data. Clearly, the flows of P and K were significantly smaller. The figure indicates the significance of the gaseous loss of N on denitrification and N fixing in leguminous pastures. Compared with the fixation process the introduction of artificial fertiliser appears relatively insignificant. Nevertheless a higher degree of uncertainty was associated with the fixation data as this is based on literature values typical for the region and pasture

Table 5 Summary of acidification potential data

Acidifying process	**Description**	**$CaCO_3$ required for neutralisation**	**References**
N leaching from fertiliser usage	Depends on percentage of N leached (10–40% from nutrient balance)	0–6 kg/ha/year averaged across the whole property	Estimates based on Moody (2005) and Slattery et al. (1991)
Animal grazing behaviour	Caused by transfer of alkalinity within paddocks to livestock camping areas	0 kg/ha/year for the VIC property 10 kg/ha/year for NSW and WA. Averaged across the grazing area	See Cayley et al. (2002)
Net product removal	Caused by removal of alkalinity with plant and animal products	7–14 kg/ha/year averaged across the whole property	See Table 4
N leaching from legume pasture	Depends on percentage of N leached (10–40% from nutrient balance).	VIC=106–121 kg/ha/year NSW=13–51 kg/ha/year WA=24–45 kg/ha/year Data averaged across the whole property	Bolan et al. (1991) Anderson et al. (1998) Ridley et al. (2001) Eckhard et al. (2004)
Alkalinity additions	**Description**	**$CaCO_3$ added (kg/ha/year)**	**References**
Lime and soil additives	Lime added to ameliorate low soil pH	0 kg/ha/year—NSW and WA supply chains. 1,177 kg/ha/year with lime application on VIC property in 2002. Values averaged across the whole property.	Process data
Net product inputs (hay, grain, livestock)	Main imports are livestock and hay	2–21 kg/ha/year averaged across the whole property	See Table 4 for composition data.

type, whereas the other N inputs were based on primary data collected at the properties.

The NSW and WA grazing properties gained more P than they lost in both years. In these cases, the main inputs were fertilisers, and the main output was the P in the animals themselves. Only the VIC supply chain exhibited a net loss of P—roughly equally shared between the animal products and dissolved losses in runoff. This reflects a decision by the property manager to allow the P concentration of the property's soils to decrease from the high concentrations it had reached under the previous owner's management. Nevertheless, all properties lost P via erosion, typically in the order of grams per kilogram HSCW. This could possibly have consequences downstream—freshwater ecosystem eutrophication is frequently associated with excess P, and the growth of blue-green algae is favoured under conditions where the molar ratio of N/P<15 (Paerl 2008).

The K flows in these grazing properties were generally smaller than the P flows except at the VIC property. The other balances show significant differences between the three grazing properties. The WA and VIC properties added significant quantities of K through fertiliser. In particular, large quantities of K_2SO_4 (26 t over 2 years) were spread on the VIC property to correct a small deficit of K and a large deficit of sulphur. For both these grazing properties, exports through livestock products are likely to represent the only significant losses from the system in both years. For the NSW property, no K fertiliser was spread in either year. Nevertheless, some K losses through soil erosion and dissolution in runoff would be expected.

Table 6 Results of NRM indicators per kg of HSCW

Grazing property	Year	Nitrogen (kg N/kg HSCW)	Phosphorus (kg P/kg HSCW)	Potassium (kg K/kg HSCW)	Soil acidification (kg CaCO3-eq/kg HSCW)
WA	2002	−0.120	−0.0180	−0.0130	0.23
	2004	−0.170	−0.0190	−0.0091	0.16
VIC	2002	−0.021	0.0051	−0.0210	−1.20
	2004	−0.058	0.0039	−0.0950	0.28
NSW	2002	0.028	−0.0170	0.0025	0.14
	2004	−0.076	−0.0085	0.0040	0.27

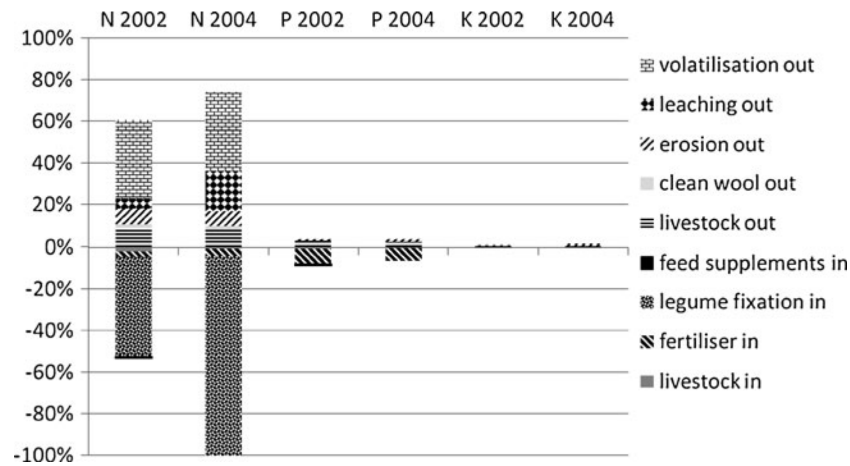

Fig. 1 Nutrient balances at the NSW property in 2002 and 2004

Figure 2 shows the acidification potential results for all six data sets, scaled to the NSW 2004 total. Note that this does not show the full extent of the most divergent result—the 2002 negative acidification potential at the VIC property. This was actually 350% of the NSW 2004 total. As noted above, in 2002 the property manager performed a deliberate and very significant base cation addition, consisting of lime and basalt rock dust equivalent to 1,200 kg $CaCO_3$/ha/year. In acidification potential terms, this was more than ten times the effect of livestock export and N leaching from legume pastures. All the other results show some soil acidification potential which was predominantly the consequence of N leaching from legume pastures and grazing effects. The average of the results, excluding the VIC 2002 outlier, is 0.22 kg $CaCO_3$-eq/kg HSCW. This acidifying effect seems to be a consistent downside to an otherwise effective process for balancing the N budget of the properties.

While we are unaware of other LCA analysts reporting the three macronutrient balances as we have done, other authors mentioned in this paper who studied beef and pork production have generally reported acidification potentials in the range 0.02–0.50 kg SO_2^-equivalent/kg meat. While one may be tempted to compare these data with our results based on their hydrogen ion equivalency, there are some important differences between previous approaches and this work, including the differences in system boundaries, our inclusion of grazing effects and the export of produce as an acidifying process. Where others set out to examine agriculture's impact on the rest of the environment, in this "gate-to-gate" study we were interested in dealing with local natural resource management issues and considered the agricultural property to be the impacted environment. In integrated agricultural corporations and where chain-of-custody information about grain supplies is available, it would be feasible to extend this work to background agricultural production systems. At that point one would be able to compare the classical indicator of regional acidification potential with a total system agricultural soil indicator and explore scenarios in which tradeoffs between local and regional impacts might be presented in an overall sustainability framework (e.g., Lundie et al. 2006).

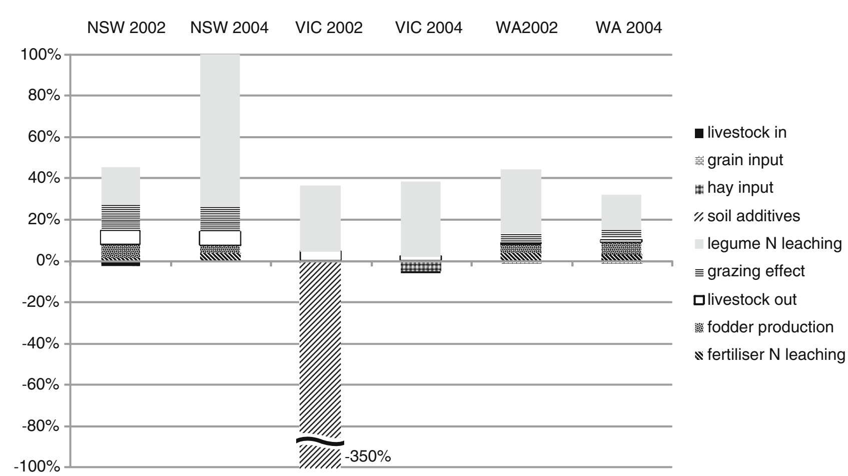

Fig. 2 Acidification potential balance for three properties and 2 years

4 Conclusions

While we have reported on greenhouse emissions, water use and other typical LCA indicators previously, this article is focussed on one issue and should be read in conjunction with the previous papers on this study (Peters et al. 2010a, b) to gain a more complete LCA picture of the systems and a broader range of life cycle indicator results. By using the same functional unit and temporal boundaries in a study of agricultural soil acidification and regular LCA indicators, the potential to make (qualified) contributions to the discussion around balancing local and global impacts is created.

We have constructed a nutrient balance and estimated the soil acidification potential of on-farm activities on the basis of detailed process data acquisition for three Australian properties producing red meat. Sensitivity analysis is difficult given the current availability of data for this work. The results are responsive to variations in agricultural practice and can display an order-of-magnitude range when annual data are compared. The principal uncertainty associated with these results relates to the need to use literature estimates of the N fixed by leguminous pastures due to the infeasibility of cheaply monitoring this flux on the scales required. Quantifying mass transfer of this N can be difficult even with experimental monitoring, and the literature generally represents measurements made at a specific site over a relatively short time frame, reducing the degree of confidence in these results. Further improvement would require more research in the field of gaseous emissions from intensive and extensive grazing enterprises over time across different systems. The main limitation of our results for environmental management is the absence of consideration of the local system's resilience (e.g. soil nutrient stores and pH buffering capacity). However, this limitation is common to all LCA midpoint indicators.

While many agricultural practices currently manage soil nutrients retrospectively by monitoring nutrient concentrations in soil or pasture crops, the nutrient balance approach could potentially be used as a predictive farm management tool. It would be interesting to apply the approach outlined in this paper to a European or north-American agricultural enterprise and identify the extent to which the acidification potential associated with acid rain might play a significant role in comparison to the agricultural practices examined here. It would also be interesting to examine agricultural products other than red meat.

As with many other environmental pressure indicators, applying countermeasures to mitigate the potential environmental impact is technically feasible; the question is whether they are applied in practice. Our results suggest that the typical scale of the pH-increasing processes that can reduce acidification without resorting to importing mineral products is limited. Finding ways to reduce the leaching of soil N might be the best way to balance the N budget without causing acidification.

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