Why restore marginal cropland to permanent pasture?
Land resource and environmental issues

D. MARK SILBURN, J. BRETT ROBINSON AND DAVID M. FREEBAIRN
Natural Resources & Water (Qld), Agricultural Production Systems Research Unit, Toowoomba, Queensland, Australia

Abstract

Soil physical and chemical degradation associated with cropping has resulted in reduced economic and environmental performance. This degradation of soil resources occurs slowly, but insidiously, and the effects may be hidden by variability in climatic and economic cycles. Permanent pasture or pasture-ley systems are important options available as alternative enterprises or as part of an ameliorative strategy. Addition of a pasture phase in a cropping system needs to explicitly recognise the soil properties that need to be improved, which may include surface or subsurface structure, nitrogen availability, organic carbon accumulation, or ephemeral soil surface conditions (cover, fracture of restrictive layers, surface storage). Benefits to soil physical condition developed under pasture are fragile and easily destroyed by tillage, wheel traffic and animal traffic and grazing. The benefits are best retained, when cropping is reintroduced, by using minimum tillage, stubble retention and controlled traffic, and by optimising fertility and crop productivity.

Pasture systems are inherently more efficient in using rainfall (they result in less runoff and deep drainage) than cropping, but the economics are dependent on relative prices of commodities and climatic conditions at a given time. In areas of high salinity risk, pastures have an important role in reducing deep drainage, although the placement of different land uses needs to consider salt stores and water pathways. The change in deep drainage when cropping is replaced with pasture is greatest on lighter-textured or shallow soils with lower plant available water capacity, giving considerably more reduction in the volume of recharge per unit area of pasture established.

Introduction

Cropping can become unsustainable on poorer soils, due to erosion, low water use efficiency, excessive deep drainage, restricted productivity and decline in productivity due to erosion, structural degradation and nutrient rundown. Well managed pastures can markedly improve soil physical properties, largely by increasing the organic matter content of the topsoil. Pasture legumes, particularly lucerne, can provide significant quantities of nitrogen for subsequent crops or grass pastures. Thus, soils that are fragile or somewhat marginal for cropping may be rehabilitated using pastures and farmed using pasture ley-crop rotations, with cropping more likely during wetter climate cycles. However, many poorer ‘cropping’ soils are best suited to permanent pasture. In this paper, we discuss the role of pastures in cropland in the northern grain-cropping zone, which includes southern inland and central Queensland and northern NSW (Freebairn et al. 2006). We will discuss environmental outcomes (runoff, soil erosion, salinity risk) and effects of pastures on soil properties, where there has been significant new work since the Fourth Australian Tropical Pastures Conference in 1990.

Suitable pasture grass and legume species for the region are discussed by Bellotti et al. (1991) and Lloyd et al. (1991). Bellotti et al. (1991) also discuss the search for reliable methods for pasture establishment on marginal cropping lands in the subtropics. Establishment is still proving difficult some 15 years on, with a considerable proportion of failures (L. Eyre, QMDC, personal communication).
Casual road surveys in the cropping areas of Queensland indicate significant areas of previously cropped land have been returned to pasture, e.g. indicated by the presence of contour banks in pasture lands. However, records or mapping of these changes in land use are difficult to find. In the Condamine catchment, at least 76 000 ha of land cropped in 1951 was returned to pasture by 1999 (Biggs 2007). In the inland Burnett, some 20% of previously cropped Red Ferrosols was returned to pastures (primarily for beef cattle) since the mid-1980s (Bell et al. 1997). Galletly (1985) refers to some 10 000 ha in the Lockyer Valley that had lost up to 0.5 m of soil and was retired from cropping. Anecdotal evidence indicates that considerable areas of more fragile soils have been converted from cropping to pasture in recent decades, e.g. on sodic, duplex soils in the Bauhinia area in central Queensland (study area of Stevens et al. 2006), on shallow soils on sloping lands in the south Burnett, on Red Kandosols and Chromosols in the western Moonie catchment and in mulga areas south and west of St George.

The reasons for these changes in land use include:

- cropping on soils that were unsuited to cultivation, from a land capability point of view
- rundown in soil properties resulting in lower land capability under continuous cropping. This includes loss of soil depth and fertility by soil erosion (Loch and Silburn 1997), decline in organic carbon (Dalal and Mayer 1986a) and nitrogen (Dalal and Mayer 1986b), and decline in soil structure and infiltration (Bell et al. 1997; Connolly et al. 1997).
- extended periods of below average rainfall
- farms that were too small to remain viable for cropping
- competition for labour; cropping has high peaks in labour demand
- changes in profitability of grains relative to cattle and sheep over time
- lifestyle choices and personal preferences and skills.

In reality, landholders will largely base their choice of cropping or pasture on the last 2 factors. However, land capability and rundown ultimately affect profitability, and the effects emerge over time. To quote a farmer for whom the first author once worked near Talwood: ‘we don’t crop that paddock because it doesn’t do as well anymore’. In addition, personal preferences and skills change over time. Thus, these changes in land use are not necessarily permanent, with generational and ownership changes often changing enterprise mixes. In much of the northern grain-cropping zone, land is taken in and out of cropping depending on the patterns of rainfall and changes in relative profitability of cropping and animal production enterprises. Cropping has sometimes been used to recover the costs of land development, to increase soil phosphorus or to control regrowth of woody vegetation.

Issues that have become more apparent in cropping in some areas in recent decades are poor water use efficiency and salinity risk. Some soils have such low water holding capacity that they are quite ‘leaky’ even in drier areas (e.g. <600 mm/yr rainfall). Deep drainage is high under fallow-cropping despite the low rainfall. This is due to the ‘lumpiness’ of rainfall — large falls or wet weeks, months or years (Yee Yet and Silburn 2003; Owens et al. 2007). Thus, leaching losses of nitrate nitrogen may be greater than previously thought.

Environmental outcomes

Runoff and soil erosion

Runoff, soil erosion and suspended sediment loads are generally lower from well managed pastures than from cropping (Tables 1 and 2), although results depend on management for both land uses. Soil erosion and suspended sediment loads can be reduced considerably by retaining surface cover in cropping systems (Freebairn and Wockner 1986). Similarly, runoff and erosion increase considerably in pastures when cover is below 30–40%.

Stevens et al. (2006) measured runoff and erosion for: (a) 2 years before and 4 years after conversion from cropping to pasture; and (b) a permanent pasture, on sodic duplex soils (near Bauhinia) in the Dawson Valley, central Queensland. Rainfall in all years (400–600 mm/yr) was below average (684 mm/yr). Runoff was 20–24% of rainfall during the cropping period, compared with 6–14% from pasture in the same period. Soil erosion was 6 times greater from cropping than from pasture. This is because cover was about 20% on the cropped area and 80% on the pasture. Runoff, as a proportion of rainfall, from these soils was considerably higher than from most soils used...
Restoring marginal cropland to permanent pasture

for cropping (see for example Tables 1 and 2). The data are for drier-than-average years; runoff would be even greater in wet years. Severe soil erosion under cropping has removed the surface layer, which contained most of the nutrients, exposed the sodic subsoil and reduced the productivity of these soils. Cropping has now mostly been replaced with pasture on these sodic soils. It is questionable whether they should ever have been cropped, especially when the loss of surface structure and nutrients is considered, and the unlikely restoration of these soils in the foreseeable future.

Many poorer soils, with intrinsically low infiltration rates, lose a high proportion of rainfall as runoff. For example, 25% of rainfall became runoff under bare fallow cropping on an Alfisol in the semi-arid tropics in India (Cogle et al. 1996). This soil has a sandy loam surface, is susceptible to crusting and hard-setting, and is similar to some lighter-textured soils that are cropped in the northern grain-cropping zone. Cropping has now mostly been replaced with pasture on these sodic soils. It is questionable whether they should ever have been cropped, especially when the loss of surface structure and nutrients is considered, and the unlikely restoration of these soils in the foreseeable future.

Many poorer soils, with intrinsically low infiltration rates, lose a high proportion of rainfall as runoff. For example, 25% of rainfall became runoff under bare fallow cropping on an Alfisol in the semi-arid tropics in India (Cogle et al. 1996). This soil has a sandy loam surface, is susceptible to crusting and hard-setting, and is similar to some lighter-textured soils that are cropped in the northern grain-cropping zone. Runoff was 9% of rainfall under buffel (Cenchrus ciliaris) and 3% under perennial Verano stylo (Stylosanthes hamata). Runoff was also reduced by adding stubble cover in the cropping system, to 5% of rain, but deep drainage increased to 18% of rain. The high runoff potential and low plant available water capacity (PAWC) (115 mm in 0.8m) severely limit the water use efficiency that can be achieved with cropping on such soils. Note that most of the annual rainfall in the semi-arid tropics in India, and in the semi-arid tropics in northern Australia (i.e., the wet-dry tropics), falls in a 3–4 month period; soil may be saturated when further rain falls, and runoff and drainage are inevitable.

## Nutrients in runoff

Comparative data for nutrient runoff from cropland and pastures in Queensland are rare. However, Stevens et al. (2006) measured total kjeldahl nitrogen (TKN) and total phosphorus (TP) in runoff, in the erosion study described above. Annual TKN loss in runoff was 5–13 kg/ha from cropping, 7 times greater than from perennial pasture. Annual TP loss was 1–3.4 kg/ha from cropping, 11 times greater than from pasture. Little is known about the comparative losses of dissolved nutrients. Dissolved nutrients and the bio-available component of nutrients in sediment pose a greater risk to aquatic ecosystems and are also less likely to settle-out than are nutrients attached to sediment. Late spring to mid-summer was the critical period for occurrence of runoff, soil loss and TKN and TP losses in the study of Stevens et al. (2006). Thus, good levels of cover need to be maintained in spring and summer to minimise runoff, erosion and nutrient losses.

## Effects of erosion and nutrient loss on subsequent productivity

Where severe soil erosion and nutrient decline have occurred during a cropping phase, the productivity of subsequent pastures will be reduced. Chilcott et al. (2004) simulated the effects of loss of water holding capacity and surface soil nitrogen supply on pasture production, for 9 land types in the western Downs and southern Dawson (situated between Miles, Wandoan and Roma). GRASP (Littleboy and McKeon 1997), a native pasture growth and soil water balance model developed for semi-arid and tropical grasslands, was used. Stocking

### Table 1. Mean annual rainfall, runoff and sediment losses from tilled and pasture catchments near Greenmount, eastern Darling Downs, Nov 1986–May 1990. Pasture established Oct 1986; purple pigeon grass and snail medic; mown for hay, rarely grazed. (From the study site of Freebairn and Wockner 1986).

<table>
<thead>
<tr>
<th>Catchment condition</th>
<th>Wheat, bare fallow</th>
<th>Pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rainfall (mm/yr)</td>
<td>700</td>
<td>700</td>
</tr>
<tr>
<td>Cover (%)</td>
<td>6</td>
<td>99</td>
</tr>
<tr>
<td>Runoff (mm/yr)</td>
<td>66 (9.5%)</td>
<td>36 (5%)</td>
</tr>
<tr>
<td>Total soil loss (t/ha)</td>
<td>40</td>
<td>1.2</td>
</tr>
<tr>
<td>Suspended sediment loss (t/ha)</td>
<td>4</td>
<td>0.2</td>
</tr>
</tbody>
</table>

### Table 2. Mean annual rainfall, runoff and suspended sediment losses from tilled and pasture catchments near Wallumbilla, Jan 1, 1990–Dec 31, 1995. Pasture established Mar 12, 1990; buffel grass and snail medic; grazed. (From Freebairn et al. in press).

<table>
<thead>
<tr>
<th>Catchment condition</th>
<th>Traditional tillage</th>
<th>Pasture</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rainfall (mm/yr)</td>
<td>495</td>
<td>495</td>
</tr>
<tr>
<td>Mean cover (%) (range)</td>
<td>25 (5–50)</td>
<td>70 (50–80)</td>
</tr>
<tr>
<td>Runoff (mm/yr)</td>
<td>33 (6.6%)</td>
<td>9.6 (2.3%)</td>
</tr>
<tr>
<td>Number of runoff events</td>
<td>12</td>
<td>11</td>
</tr>
<tr>
<td>Suspended sediment loss (t/ha)</td>
<td>2.25</td>
<td>0.24</td>
</tr>
</tbody>
</table>
rates were adjusted annually to a set rate of utilisation of pasture on offer.

Compared with no soil loss, loss of 50 mm of soil caused only a small reduction in long-term pasture production (e.g. 5%) on the best 4 land types (bragalow-belah, softwood scrub and alluvial soils) and a small increase in pasture utilisation rate. However, on poorer land types (poplar box and other forest soils), this level of soil loss reduced pasture production by 8–24%. While this is not an extreme reduction, it resulted in the pasture utilisation rates increasing from <30% to 40–50%, if stocking rate was maintained at the original level, compared with a sustainable level of 20–25% (grazers’ consensus for safe carrying capacity). Utilisation above 30% causes degradation of grass basal area and reduced soil cover. Greater depths of erosion caused further reductions in pasture productivity and increase in utilisation rates, with the less fertile land types most affected. If stocking rates are not adjusted to account for the true land capability, high utilisation rates lead to further degradation and erosion, leading to a downward spiral of accelerating degradation and loss of productivity.

Salinity risk

Deep drainage, or water movement below the root zone, is the major component of salinity risk that can be managed (Grundy et al. 2007). Most other factors are more-or-less inherent features of the landscape. Increased deep drainage can cause increased lateral flow or increase groundwater recharge and raise groundwater levels. Salinity or waterlogging occurs where lateral flow or groundwater discharges in lower slopes or into streams. Long-term average deep drainage generally increases with increasing rainfall, reduced perenniality (‘green days’ per year), for instance with annual cropping instead of pasture, and with reduced soil PAWC. Soil PAWC is lower with shallower-rooted vegetation, for instance with annual crops instead of pasture or woodland, thus drainage is greater. Deep drainage typically increases in the order: woodland < pasture < cropping < irrigation, because of the differences in ‘green days’ and rooting depth for these land uses. These trends have been demonstrated using soil water balance modelling (Yee Yet and Silburn 2003; Owens et al. 2007) and field measurements of deep drainage using soil chloride in the Queensland Murray-Darling Basin (Tolmie et al. 2003) and the Fitzroy (Radford, B.J., Silburn, D.M. and Forster, B.A., unpublished data).

Managing existing salinity or prevention of future salinity typically involves reducing deep drainage in the contributing area of the groundwater system, upslope of the salinity occurrence. This area can be of the order of square kilometres and may involve a number of land owners, soils and land uses. Management often evolves into a mosaic of options: tree planting, conversion of cropland to pasture, increased water use in cropland using opportunity cropping, and improved water use efficiency in irrigation. This leads to difficult choices of where in the landscape to invest in changes in management. Where can we get the most ‘bang for the buck’?

To explore part of this question, we looked at whether soil type was important in determining investment in land use change. We estimated deep drainage for pasture and wheat cropping on soils with a range of water holding capacities, using the Howleaky? daily soil water balance model (Rattray et al. 2004). Climate data from the north-east Darling Downs (Oakey, 633 mm/yr rainfall) and the Maranoa (Roma, 598 mm/yr rainfall) for 1960–2006 were used. Both areas have a range of soils, including soils marginal for cropping, and mixed farming including livestock, and returning cropland to pasture may be a viable option. Soil water holding capacities (PAWC) were measured at sites in the region, after ponding and allowing the soil to drain, to measure the bulk density and drained upper limit (field capacity) profiles, and after a crop (finished under a rainout shelter) to obtain the crop lower limit profile (wilting point), as described by Dalgliesh and Foale (1998) and Christodoulou et al. (2001). For simplicity, pastures and crops were given the same rooting depths for each soil, whereas pastures may in fact have deeper rooting depth than crops in some cases and would then have somewhat lower deep drainage. Differences in runoff potential, soil permeability and drainable porosity of the soils are also accounted for in the modelling.
(a) Model sensibility testing

Deep drainage rates for the Vertosol (198 mm PAWC) in the Maranoa correspond well with estimates of deep drainage at the Wallumbilla catchment study of Freebairn et al. (in press) where the PAWC was measured. For this site, Tolmie et al. (2003) found average annual deep drainage of 9–12 mm/yr for wheat cropping, compared with 10 mm/yr modelled, and 0.9 mm/yr for a permanent pasture, compared with 0.4 mm/yr modelled (Figure 1). Modelled runoff for wheat cropping in the Maranoa was 6.2% of rainfall compared with 6.6% measured at Wallumbilla catchment study (Freebairn et al. in press). Runoff from pasture was about half that from wheat-fallow cropping, in line with measured results described above (Tables 1 and 2).

Red Kandosols and Chromosols in the region typically have low soil chloride under native vegetation, indicating that enough deep drainage occurs naturally to prevent a build up of rainfall-derived chloride in the soil profile. For example, Young and McLeod (2001) measured chloride (Cl⁻) concentrations <20 mg/kg in a Red Chromosol in the Manilla district in northern NSW. Tolmie and Silburn (2003) estimated that deep drainage greater than 10–20 mm/yr was necessary to maintain this low soil chloride level, based on steady state Cl⁻ mass balance. Modelled deep drainage for pasture in the Maranoa averaged 23 mm/yr for a Kandosol with 95 mm PAWC, consistent with the estimate from soil profile chloride analysis.

(b) Deep drainage and change with pasture for a range of soils

Estimates for the northern Darling Downs showed that deep drainage increases as PAWC decreases for both cropping and pasture (Figure 1). This is despite the poorer soils giving higher runoff (data not shown). Similar results were found for the Maranoa, though deep drainage rates were slightly lower due to the lower rainfall (data not shown). The deep drainage is driven by the ‘lumpiness’ of rainfall (large falls or wet weeks, months or years), but this is exacerbated by water storage during falls in the cropped system (Freebairn et al. 2006).

Soils with lower PAWC have less capacity to store these large rainfall events. For the poorer soils, considerable deep drainage occurred under pasture. However, from the point of view of reducing deep drainage, the largest change in drainage was obtained on soils with lower PAWC (Figure 2). The reduction in drainage was 4–5 times greater for the soils with low PAWC than for the soils with high PAWC. Thus, the area of soils with high PAWC converted to pasture would need to be 4–5 times greater than for low PAWC soils, to achieve the same reduction in the volume of recharge.

The Red Kandosol soils and other shallower soils in our landscapes often occur upslope from the heavier soils (A. Biggs, NRW, personal communication). Thus, reducing deep drainage where these soils are cropped in the upper slope may reduce salinity and water-logging in the better cropping soils lower in the landscape. In contrast, reducing deep drainage in the lower landscape may not be as effective.
not prevent salinity occurring there, if groundwater discharge arrives from the upper slopes. Clearly a landscape approach is required to analyse each situation, determining sources of increased groundwater recharge and likely outflow points.

(c) Managing/preventing salinity and maintaining profitability

Wylie (1997) found that cropping was less profitable on lighter or shallower soils (i.e., lower PAWC) than on deep clays (high PAWC) in both wetter (650 mm/yr rainfall) and dryer cropping areas (570 mm/yr rainfall). Fewer crop rotations were profitable on the poorer soils, particularly for wetter locations. A wheat-lucerne ley rotation was the most profitable rotation on the poorer soil in dryer locations, so long as the infrastructure for livestock was in place. Thomas et al. (1995) found that soils with a PAWC of less than 120 mm, in the Maranoa, would be either marginal or unsuitable for cropping, independent of soil type. Converting cropland to pasture on poorer, shallower soils (PAWC <120 mm) should give the greatest reduction in deep drainage, may be better in terms of the spatial arrangement in the landscape, and should maintain profitability. However, results will vary with changes in the relative prices for crop produce and livestock, the age of cultivation, which will determine the nitrogen status of the soil (Thomas et al. 1995), and the scale and capital costs involved in the enterprise.

Effects of cropping and pasture on soil properties

Decline in soil organic matter, carbon and nitrogen

Soil organic matter (measured as organic carbon, OC) is widely observed to decline under cropping (Dalal and Mayer 1986a; Dalal et al. 1991; Bridge and Bell 1994; Connolly 2000). This decline occurs when the rate of loss from the soil through decomposition exceeds the rate of organic additions (e.g. crop residues, manure). Decomposition is enhanced through changes in temperature, aeration and soil water, and aggregate disruption associated with tillage, which exposes new soil surfaces, improving microbial access to organic matter. Soil OC is associated with reduced supply of plant-available nitrogen, reduced aggregate stability and poorer infiltration. The decline in OC is illustrated for Sodosols in Figure 3a.

Restoring soil fertility

Dalal et al. (1991) gives the main management options for restoring or maintaining soil fertility, particularly N status, in cropping systems as:

• fertiliser application
• minimum or zero tillage
• rotations containing grain legumes
• rotations containing legume and grass-based legume pastures
• combinations of the above.
Minimum or zero tillage with fertiliser application may maintain, but not greatly increase, soil carbon (Dalal 1989; Bell et al. 1997). The legume options can contribute significantly to nitrogen supply to crops and offer economic diversity (Holford 1980; Littler 1984; Thomas et al. 1996; Wylie 1996; Doughton and Holford 1997; Robinson 1998). The adaptation of legumes in the region is discussed by Lloyd et al. (1991; 2007) and Doughton and Holford (1997). Legume pastures and crops can sometimes also be used to reduce problems with weeds, pests and diseases. Here we discuss the role of the main legume option, lucerne, in restoring soil conditions (e.g. organic carbon and soil physical properties) — nutrient fertility is covered in the references cited above.

(a) What about lucerne?

It is important that a farming system is profitable if it is to be adopted, and only then will environmental benefits be achieved. Lucerne pastures in the studies cited above, on average, produced substantial improvements in yields, protein levels, prices and gross margins of subsequent wheat crops. Lucerne can be profitable where livestock are already part of the enterprise, though greater management skills are required (Lloyd et al. 1991; Wylie 1996; Robinson 1998). There is also considerable evidence from these experiments that fertility benefits from lucerne persist for several years. This strategy has been made more achievable with the development of more winter-active lucerne varieties that have higher yields and improve soil nitrogen more than older varieties, as nitrogen benefits are directly related to dry matter production.

With the soil-fertility benefit of lucerne to subsequent crops, lucerne will be used mainly as a ley pasture in rotation with crops. However, rotation of lucerne with grass pastures should be considered on soils that are marginal for cropping, given the large response of rundown pastures to nitrogen (Graham et al. 1981).

Soil organic carbon increased under grazed lucerne (10–20% in 4 years) in some trials, but declined in others owing to lower rainfall and low yields (Doughton and Holford 1997). Soil OC change is related to the amount of dry matter produced. Thus, lucerne alone will probably not improve surface soil OC on marginal or less productive soils or in dryer areas. Lucerne grows deep roots (>2 m) where soil properties are not limiting, and extracts water to greater depths and to dryer water contents than annual crops, giving a higher PAWC (Neal Dalgliesh, CSIRO, personal communication). Thus lucerne-wheat and lucerne-grass pasture rotations may provide profitable options for the deep drainage reduction and salinity management discussed above. The effects of lucerne on soil hydraulic properties are discussed later.

(b) Recovery of soil OC under grass pastures

Grass-based pastures (grazed in situ) should provide the greatest rates of recovery of soil carbon, as they provide the greatest carbon input. Even so, rates of recovery of soil OC under pasture are relatively slow. Bell et al. (1997) found little increase in OC with 2–4 years of pasture on Krasnozem (Red Ferrsol) soils, although considerable undecomposed dry matter was present, some of which would presumably cycle into soil OC. Bell et al. (2003) have since reported that
‘a 4-year grass ley (kikuyu, *Pennisetum clandestinum*), followed by a return to cropping using direct drill practices, increased soil OC in the 0–0.05 m layer by 40% and labile C by 60%. Conventional tillage after the ley resulted in a loss of approximately 66% of both total and labile organic C from the top 0.1 m of soil within 5 crop years.’ Mathers *et al.* (2006) found that the total soil OC mass in 1.1 m depth of soil after 22 years of pasture was twice that under adjoining cultivated cropland on a Red Ferrosol near Kingaroy.

R.D. Connolly (personal communication) found that soil OC increased under pasture (on previously cropped soils), by 0.02% OC/yr in the 0–0.1 m layer on most soils (Ferrosols, hard-setting red soils, light Vertosols and Sodosols (Figure 3b), and by 0.1% OC/yr on heavy Vertosols. At a rate of increase of 0.02% OC/yr, 50 years of pasture is needed for soil OC to increase by 1% OC, a substantial increase. The rate of increase in OC was about half these rates in the 0.1–0.3 m soil layer. However, results can be quite variable, ranging from 0.01–0.1% OC/yr on both light and heavy Vertosols. Further study is required before rates of carbon sequestration can be confidently assumed with re-establishment of pasture on previously cropped lands.

(a) What happens under cropping?

Tillage and wheel traffic can increase the prevalence or severity of infiltration-limiting layers by:
- exposing surface soil to raindrop impacts (causing surface sealing) by removing cover and contributing to structural degradation, causing seals to become more restrictive over time (Connolly *et al.* 1997; 2001).

*Figure 4. Inherent (*) and induced soil layers that restrict water entry. Source: Tolmie and Silburn (2001).*
creating smeared and compacted layers under the tilled layer (Silburn and Connolly 1995) or on the surface (Li et al. 2001; Tullberg et al. 2001). Tullberg et al. (2003) present clear photographs of various types of compacted layers.

- contributing to a run-down in soil organic matter and changing the composition of the organic matter, mechanical shattering of aggregates; soils may become more hard-setting and ‘cloddy’.
- increasing hard-setting if deep tillage or erosion brings sodic subsoil to the surface.
- reducing soil faunal activity by tillage and monoculture cropping (Simpson et al. 1995).

Connolly et al. (1997) measured changes in infiltration characteristics associated with years of cultivation for 5 soil groups in south-eastern Queensland, grouped according to soil type and texture into Sodosols, light Vertosols, heavy Vertosols, Red Ferrosols and Red Chromosols/Kandosols. They measured soil infiltration characteristics and water-holding properties of the cultivated layer (0–0.1 m deep) (i.e., surface seal) and the layer immediately below the plough layer (0.1–0.2 m deep). Hydraulic conductivity of surface seals decreased exponentially with period of cultivation in all soil groups; half of the decline occurred within 2–6 years of first cultivation (Figure 5a). Hydraulic conductivity, macroporosity, and the moisture characteristic of the 0.1–0.2 m layer were similarly affected by longer periods of cultivation in all but hard-setting red soils (Figure 5b). Steady infiltration rates measured in the field, which respond to both the surface seal and plough pan layers, also declined with period of cultivation for all soil groups.

The changes in soil hydraulic properties caused runoff and soil evaporation to increase, and infiltration and soil water storage to decrease, with age of cultivation (Connolly et al. 2001). Crop yields were estimated to decline by 13–40% after 50 years of cropping with intense tillage, depending on soil type, due to the combined effects of decline in soil carbon, nitrogen and infiltration properties. Gross margin declined at a faster rate, falling by 20–50% after 50 years, because more fertiliser was required to compensate for declining soil fertility. Under some circumstance this may mean cropping becomes unprofitable.

(b) Changes under pasture on previously cropped soils

Pastures typically have only a small effect on soil water storage capacity in previously cropped soils (Bridge and Bell 1994; Bell et al. 1997), but have large effects on water transmission properties. Bell et al. (1997) found grass pasture leys significantly improved the physical fertility of continuously cropped Red Ferrosols within 2–4 years. ‘The most significant effects were on the creation of improved surface and subsurface macroporosity, and in a reduction in surface crust formation under high-energy rain due to improved aggregate stability. Final steady state infiltration rates under well-managed leys increased 4-fold compared with those in continuously cropped soil’ (Bell et al. 1997). Kikuyu was more effective than rhodes grass (Chloris gayana) in improving aggregate stability under rain, and soil hydraulic properties, and was also more resistant to the compacting influence of cattle trampling during wet weather.

Connolly et al. (1998) measured soil hydraulic properties of the 5 soil texture groups studied by Connolly et al. (1997), on cropped areas replanted to grass pasture and grazed. Hydraulic conductivity of the surface seal and the plough pan improved
over time, for 4 soil groups. Where grazing was excluded for 1 soil group (Sodosols), the rate of improvement in the surface seal was greater but there was no difference for plough pan characteristics (Figure 6). No improvement occurred in either soil layer on hard-setting Red Chromosol/Kandosols. Surface sealing can be a consequence of a soil’s inherent hard-setting nature and poor aggregation, and in some cases cannot be modified by management. For the 0.1–0.2 m soil, no plough pan had developed on these soils, so no improvement was likely (Connolly et al. 1997).

The rate of improvement in hydraulic conductivity with period of pasture varied with soil layer, and was affected by grazing and soil type. The rate of improvement, however, was slower than the rate of decline in hydraulic conductivity when the soils were first cropped. The period of pasture required to return hydraulic conductivity to pre-cultivated levels ranged from 5 to 40 years, which was in general about 2–3 times the period of cultivation that caused the degradation.

(c) Changes when cropping is recommenced

Beneficial effects of 2.5–4.5 year ungrazed grass pasture leys on both soil layers declined rapidly under conventional cultivation on the Sodosol, with large declines in the first 0.5–1 yr of re-cropping (Connolly et al. 1998). Some benefit persisted in the surface soil for up to 5 years after cultivation was recommenced. Improvements in the 0.1–0.2 m layer were lost within 1 year. These rates of decline in hydraulic conductivity were faster than observed on previously uncultivated soils. Both Connolly et al. (1998) and Bell et al. (1997) emphasise the need to manage cultivation and traffic, to preserve the benefits, when ley areas are brought back to cropping.

(d) Cropping and pasture ley systems

The integration of the results of Connolly et al. (1997) on rundown under cropping and of Connolly et al. (1998) on repair under pasture is shown in Figure 7, for subsoil hydraulic conductivity. Somewhat similar responses would occur for soil OC and surface structural stability and sealing. Pasture leys gradually improve the soil, but soils respond at different rates. Improved soil physical conditions developed under pasture are fragile and easily destroyed by tillage and trafficking. Careful management during crop phases between leys is needed to maintain the improvements, preferably no tillage, controlled traffic and adequate fertiliser to maximise the carbon input from crops. If tillage and trafficking are minimised during the cropped phase, benefits may accumulate from cycle to cycle.

(e) What about lucerne leys?

Foley et al. (2006) studied the effects of 0, 1, 2 and 5 years of lucerne ley after annual cropping on the plough pan (0.15–0.25m), on 2 heavy Vertosols (Bongeen and Waco) on the Darling Downs. All treatments were wet to drained upper limit prior to the measurements to exclude the effects of soil moisture content. Unsaturated hydraulic conductivity was similar (n.s.d.) for the 4 agronomic treatments at each site, indicating that there was no residual effect of the lucerne leys and that few macropores remained in the soil. This may be because there were too few wetting and drying cycles at that depth (Pillai-McGarry and McGarry 1999; see Section (g) below).
(f) What about grazing?

It is likely that heavy grazing will reduce the rate of amelioration of surface soil (0–10 cm) and crusting during a pasture phase. Mead and Chan (1992) found grazing reduced hydraulic conductivities in surface layers of hard-setting red soils by 50%. Both Connolly et al. (1998) and Bell et al. (1997) found that grazing reduced the effectiveness of pasture in ameliorating surface sealing on box-brigalow-belah soils (Sodosols) and Red Ferrosols, respectively. On the Ferrosol, kikuyu prevented re-compaction by trampling, due to a spongy mat of rhizomatous material, whereas rhodes grass did not. Livestock impacted only the top 0.05–0.1 m, but this often necessitated a shallow tillage at the end of a pasture phase (Mike Bell, personal communication). In the case of the Sodosols, about 30 years of grazed grass pasture would be needed to return surface seal hydraulic conductivity to pre-cropping levels, compared with 7 years for ungrazed grass pasture. Grazing had no substantial impact on amelioration of soil deeper than about 0.1 m, consistent with Bell’s observations.

The most damaging effects of grazing on soil are removal of soil cover, formation of foot pads which channel water and cause erosion, and trampling causing remoulding and compaction of surface soil in wet weather. Damage can be minimised by adjusting stocking rates to maintain a reasonable body of grass and restricting stock movement in wet weather. For example, Proffitt et al. (1995) found that controlled grazing, where grazing with sheep was avoided when the topsoil was close to its plastic limit, maintained the top-soil structure (defined by bulk density, infiltration rate, strength and macroporosity) similar to that of no grazing, whereas continuous grazing degraded the structure. Controlled grazing to avoid wet conditions prevented soil remoulding rather than simply preventing compaction.

(g) Amelioration of compacted layers

Soil swelling and cracking offer advantages in amelioration of compacted layers. Pillai-McGarry and McGarry (1999) found that 3–9 wetting and drying cycles repaired the structure (defined as soil pore and ped properties) of a compacted Vertosol. Mung bean and lablab gave improvements to greater depths and with fewer wet/dry cycles than wheat and sorghum. The effects of these changes in structure on soil hydraulic properties and infiltration were not determined. Similar structural improvement would presumably occur with grass or lucerne pastures, on shrink-swell soils. However, Radford et al. (in press, a) found that compaction from wheel traffic persisted for up to 5 years under dryland no-till, no-traffic cropping on a Vertosol in the field. They considered that: (1) there were insufficient wet/dry cycles to swell and shrink the entire compacted
layer, (2) soil loosening by tillage was absent and (3) there were fewer earthworms in the compacted soil.

Some evidence indicates that amelioration of compaction in non-swelling or less-swelling soils is more difficult and much slower. Bell et al. (1997) found that 2–4 years of pasture was not effective in ameliorating compacted zones below approx. 0.15m in old cultivation on a Red Ferrosol. U. Pillai, (UQ, personal communication) found little change in bulk density in a Red Ferrosol after 25 years of pasture. Without the benefits of macropores created by swelling and cracking, plant roots and soil fauna must ‘drill’ holes in the soil; once soil is compacted, this is more difficult. Soil fauna (especially earthworms) made a significant contribution in shallower layers (0–0.3 m) in the Red Ferrosol of Bell et al. (1997) and were more prevalent with less tillage and retention of crop residues on the surface (Simpson et al. 1995).

(h) What about ripping?

‘Deep’ ripping has been used to break up compacted layers. This practice fails to increase soil organic matter levels, so is not a substitute for pastures or other crop rotations that improve soil organic matter. Compaction will result if ripping is undertaken when the soil is too wet (i.e., above the plastic limit). The soil must be quite dry to achieve the required shattering. On clays, this means the soil must be near wilting point and draft on the tractor is very high. Done correctly, ripping is expensive, and the benefits can be easily lost with subsequent poor management. Economic returns to ripping are uncommon. Ripping ‘should only be used as a last resort’ (McGarry 2001). Biological options are preferred, although they may take longer.

In trials of ripping and reseeding of pastures on rundown sites with the Taroom Shire Landcare Group (2001), we found that ripping to >0.5 m caused an increase in total infiltration, but this lasted for only about 2 years. Final infiltration rates under rainfall, at any time after ripping, were not affected by ripping. The type of ripping used caused a significant initial reduction in pasture cover, exposing the soil to surface sealing, erosion and slumping, removing the effects of ripping. The influence of cover in maintaining infiltration overrode the effects of ripping and age of ripping. Thus, grazing following ripping must be managed to allow recovery of pasture cover. Ripping increased pasture biomass production for several years, probably due to the combination of improved infiltration and release of nitrogen locked up in organic matter (Graham et al. 1981). However, cattle grazing days per ha per mm of rainfall were variable between ripping and control treatments. Ripping and reseeding did rehabilitate a degraded site on sodic duplex soils, where light grazing and extended rest periods had not. However, several questions remain concerning these trials:

- Was deep ripping (>0.5m) needed? The restriction to infiltration appeared to be in the hard-setting surface/A-horizon. Shallow tillage at lower cost may have achieved similar results.
- Was ripping an economic practice? Would landholders have used ripping if it was not subsidised, as a trial, by Landcare funding?
- Was deep ripping on the poorest soils a better option than growing improved pastures or forage crops on better soils?

Bridge and Bell (1994) noted that effects of deep ripping may not last long on Red Ferrosols in the south Burnett region. Bell et al. (1997) found ripping during the pasture phase was effective in disrupting the compacted layer to at least 0.3m, although this was short-lived if potential compaction events (e.g. vehicle traffic and livestock trampling) were not managed correctly. The deep-ripped pasture re-compacted rapidly upon return to cropping if vehicle traffic was not constrained. Pasture biomass production was halved in the year of ripping. It is therefore better to rip prior to establishing the pasture. ‘If the pan is disrupted first, the pasture (and lack of traffic during establishment) has a good chance of stabilising and retaining the loosened condition, and also becoming productive more quickly. The grower may have other reasons to deep rip (like incorporating lime to address subsoil acidity), so a major disturbance like deep ripping may be able to achieve many things at once.’ (Mike Bell, personal communication).

Recommendations for further study

There have been too few directly comparable studies on the impact of different cropping and grazing practices on hydrology, erosion and, in particular, water quality, to make confident statements about the relative environmental performances of
these systems. While estimates of likely performance can be derived from models, confidence in our predictions would be improved with more purposeful studies directed at this issue. Given the growing concern of poor water quality reaching the Great Barrier Reef and inland streams, further comparative studies will be required at a range of scales.

While loss of soil physical and chemical fertility associated with cropping is well documented, there is little understanding of the impact of an ‘exploitation phase’ of cropping or overgrazing on subsequent pasture systems. Rundown in soil physical and chemical fertility also occurs under well established pastures, with a slow but distinctive decline in productivity, even under conservative management (Graham et al. 1981; Radford et al. in press, b).

Amelioration of soil degradation, whether due to cropping or grazing, remains problematic. If soil and nutrients have been removed, it is likely that the basic soil resource is sufficiently changed that the new climax vegetation potential will be very different, and most likely less productive, with amelioration beyond reasonable economic achievement. Recognition of these situations, and avoidance of further degradation, will remain an important area for investment.

The concept of ‘marginal land’ requires further exploration, given that there are instances of both successful and unsuccessful enterprises being carried out on what are classed as unsuitable or ‘marginal’ soils. Clearly, management is the distinguishing feature that separates marginality from success. This is an economic, social and biophysical system that requires further analysis to guide us to what are sustainable systems in cropping, grazing and mixed enterprises.

Grazing and cropping enterprises have been responsible for significant releases of greenhouse gases. Should carbon trading become viable, agriculture will need to have much more definitive estimates of sources and sinks of greenhouse gases (e.g. carbon sequestration) associated with different land uses and management practices. Further study is required before rates of carbon sequestration can be assured with re-establishment of pasture on previously cropped lands.

Successful establishment of pastures remains a technical and economic challenge in northern Australia’s highly variable climate. While basic principles are well known, unreliability in establishment remains a significant factor and a major uncertainty in any economic analysis. The best methods of establishment are reasonably clear in principle but are not necessarily used. Therefore, attitudinal studies that aim to discover the limitations of the principles are needed.

Many soils in subtropical and semi-arid Australia are inherently fragile. However, where the cut-off lies between sustainable and unsustainable, profitable and unprofitable, depends on the run of seasons and relative prices of grain, sheep and cattle. Seasonal and decadal weather forecasts offer some promise in managing risks associated with the variable climate, though there is clearly some way to go before many of these forecasts can be used routinely with any degree of confidence.

**Conclusion**

There are many instances where physical and chemical degradation of soil resources associated with cropping have resulted in reduced economic and environmental performance. These reductions occur slowly and can be hidden by climatic and economic variability and trends. Converting such croplands to permanent pasture, or pasture-ley systems, can reduce or arrest such decline. However, any such change in management needs to target the particular soil properties, which are limiting (surface or subsurface structure, nitrogen availability, organic carbon accumulation or soil cover improvement). Benefits to soil physical condition developed under pasture are fragile and easily destroyed by renewed tillage and trafficking. If tillage and trafficking are minimised during the cropping phase, benefits may accumulate from cycle to cycle.

Rainfall use efficiency is low under cropping, particularly on soils with low plant available water capacity and high runoff potential. Well managed pastures are inherently more efficient in using rainfall and result in less runoff and deep drainage than cropping. However, the economic efficiency of such systems depends on climatic conditions and on relative prices of commodities. In areas of high salinity risk, pastures have an important role in reducing deep drainage, although the placement of different land uses in the landscape needs to be considered on a case-by-case basis.
Acknowledgements

The authors acknowledge Grains Research and Development Corporation (GRDC), Land and Water Australia and the National Action Plan for Salinity and Water Quality (NAP), who have supported research that directly and indirectly contributed to this paper. The deep drainage modelling was funded by GRDC project DNR00006 and NAPSWQ project SIP-AG07. We also thank an unknown referee for useful comments and corrections.

References


and Research Development Corporation project DAQ124A. Queensland Department of Natural Resources, Brisbane.


