1. INTRODUCTION

1 Market gardening and the environment - Research Objectives

1.1 Market gardening as a pollution source

Market gardening (commercial vegetable production) is commonly characterised by a number intensive and potentially environmentally harmful land management practices. Generally, fertiliser is applied frequently and in large quantities, often in combination with intensive tillage and irrigation practices. Often the protective cover of vegetation is less than in undisturbed conditions so there is potential for accelerated soil erosion during storms. It is the intensity and frequency of land management practices in market gardening which set it apart from any other land use. There is a high risk of soil and nutrient loss, potentially resulting in both degradation of agricultural land and receiving waterways. Beaulac and Reckhow (1982, 1986) describe row cropping, a practice that is commonly used in market gardening, as a land use that "... promotes rapid water runoff (through channelisation) and high soil erosion, which cause large quantities of nutrients to be exported with sediments and particulate matter."

For a number of years it has been recognised that conventional land management practices in market gardening are potentially harmful to waterways (Cornish, Kuczera & Murison, 1992; Martin & Steain, 1995). NSW Agriculture has recently produced 'Best practice guidelines for growing vegetables' (NSW Agriculture, 1997). A plot trial has been underway since 1991 at the Somersby Agricultural Research Station near Gosford, where 5 vegetable growing systems (each on 0.1 ha plots) are being compared in terms of their productivity and their water pollution potential (Cornish et al., 1992). The findings thus far have indicated that traditional practices are the most productive, but also produce the most nutrient and soil loss in runoff (Nells, 1996).
While market gardening is readily perceived to adversely impact receiving waterways, there is remarkably little Australian field or farm-scale data to quantify these impacts. Current knowledge of the contribution of market gardens to pollution of the Hawkesbury-Nepean River is based on a few interstate and overseas studies (Marston, 1994).

Credible data are needed to justify a change in land management, in other words, to determine the impacts of a typical commercial market garden and how specific land management practices affect soil and nutrient loss in runoff. There is a need to relate land management practices to runoff quantity and quality.

1.1.2 The Hawkesbury-Nepean - a river under pressure

Degradation of rivers and streams is now and will continue to be, one of the major environmental issues in Australia. Freshwater is a resource on which society depends and the consequences of poor river health are far reaching. Stormwater transporting nutrients (nitrogen (N) and phosphorus (P)), chemicals, bacteria and litter from a variety of land uses is a major issue in the Hawkesbury-Nepean River (Kerr, 1993).

Nutrient enrichment has been the major issue in the Hawkesbury-Nepean River. Water quality monitoring in the Hawkesbury-Nepean has shown that concentrations of N and P exceed criteria for aquatic ecosystems and recreation (Kerr, 1993). Blooms of toxic blue-green algae (resulting from favourable conditions including high nutrient concentrations) have occurred on several occasions making water unsafe for human and stock consumption (Gallagher & Egerrup, 1995) and the risk of future blooms remains high (Kerr, 1993). Algal blooms are not only dangerous to humans and stock. As bacteria feed on the decaying algal matter, oxygen is drawn from the water column, decreasing its capacity to support aquatic life.

Point sources of pollution, such as sewage plants are relatively easy to quantify, while non-point or diffuse sources such as stormwater from agriculture and urban areas are far more difficult to quantify. Sewage treatment plants are recognised as a major point source of N and P in the Hawkesbury-Nepean River (Kerr, 1993), but the contribution from agriculture is less certain. Stormwater runoff from market gardens has been partially blamed for water quality...
decline in the Hawkesbury-Nepean River. While this blame has not been without some basis (Davis et al., 1991) the impacts of market gardening remain poorly understood.

1.1.3 Socio-economic circumstances

Changing land management requires social change. In the context of the market gardening industry in the Sydney region, this is the major challenge facing agricultural extension officers. Market gardening is not a highly profitable industry but it is extremely labour intensive. Kelleher, Chant and Johnson (1997) suggested annual gross income from market gardening is approximately $11,600 per ha. Low profit margins leave little room for risking changes to traditional practices that are currently providing a living.

A regulatory approach to achieving better land management will be difficult to enforce and is likely to threaten the industry if it is not accompanied by voluntary change. Undeniably, agricultural extension holds the key to bringing about widespread improvement to land management in the market gardening industry in the Hawkesbury-Nepean catchment.

With approximately 4 million people living in the Sydney metropolitan area, there is a large market for fresh vegetables. Traditionally, the major vegetable growing areas for the Sydney market have been the Class 1 land on the floodplain of the Hawkesbury-Nepean River (Kelleher et al., 1997). However, with the growth of the turf industry and the risk of flooding that comes with farming the floodplain, market gardens are now located predominantly on Class 2 and 3 land (Kelleher et al., 1997). Urban encroachment on market gardens in areas such as Box Hill and Rouse Hill has also contributed to this relocation. In their study of community attitudes to agriculture in Hawkesbury and Wollondilly Local Government Areas (LGA’s), Kelleher et al. (1997) reported that market gardening in the newer areas has been typified by considerable investment in soil fertility improvement and storages for irrigation water supply.

A major part of the problem of land and water degradation is that growers seeking to maximise profits need not be overly concerned about reducing nutrient and soil losses in stormwater
runoff because there is a cheap and readily available supply of poultry manure on the outskirts of Sydney.

Development of market gardens on Class 2 and 3 land has come at the social cost of conflict with peri-urban and rural residential neighbours. Kelleher et al. (1997) cite conflict in Wollondilly and Hawkesbury LGA’s arising over community concerns regarding clearing of land/destruction of wildlife habitat, chemical over/misuse, runoff pollution of waterways, in particular causing blue green algae, siltation, and raising the water table leading to salinity. Clearly, many in the community perceive land management in market gardening as a cause of land and water degradation.

Improvement of land management in the market gardening industry has not been assisted by friction between some market gardeners and some neighbouring land users. Social conflict in the Freeman’s Reach area actually led to the formation of the Currency Creek Landcare Group, which has a strong membership of market gardeners. The tension between market gardeners and some neighbouring residents was obvious at the first meetings (which I attended) which preceded formation of the Landcare Group.

With the goal of achieving ecologically and economically sustainable market gardening, the objectives of this research were to:
1) quantify sediment, N and P loss and assess the implications for waterways;
2) relate sediment, N and P losses to specific land management practices and assess their impacts on profitability; and
3) reflect on this research in terms of extension and adoption of better land management.

These objectives were met by a case study of a commercial market garden in which stormwater losses of sediment, N and P were measured at farm-scale and related to land management. A rainfall simulator was used to compare runoff volume from green manure and bare fallow beds.

---

Chapter 1: Introduction
1.2 CONTEXT OF THIS RESEARCH

1.2.1 A multi-disciplinary team project: ‘Land and water degradation due to agriculture in the Hawkesbury-Nepean’

The Catchment Management Support System (CMSS) is a simple model developed as a tool to allow catchment managers, land use planners and catchment groups to estimate nutrient loads (kg) from various land uses, including various forms of agriculture. Nutrient exports (kg ha\(^{-1}\) yr\(^{-1}\)) from agriculture in the Hawkesbury-Nepean catchment were largely unknown and when CMSS was developed for the Hawkesbury-Nepean, nutrient runoff estimates were based largely on interstate and overseas studies (Marston, 1994).

A study commenced in 1995 to address the lack of local data. Funded by the National Landcare Program, the study titled *Land and water degradation due to agriculture in the Hawkesbury-Nepean* (referred to herein as the NLP project) monitored stormwater runoff from agricultural land in 3 subcatchments located at Currency Creek, near Camden and Mangrove Mountain. The main project objective was to measure nutrient export rates from a number of agricultural land uses, including dairy, beef grazing, market garden and peri-urban land uses.

The monitoring entailed measuring and sampling flow at 11 stations predominantly in ‘nested’ arrangements (monitoring stations are located on land use boundaries to determine the inputs from each land use) in 3 small rural subcatchments ranging from 140 ha to 225 ha, located within the Hawkesbury-Nepean catchment. In each of the subcatchments, automatic monitoring stations were set up to monitor and sample stormwater runoff. Runoff samples were analysed for suspended solids, soluble and particulate forms of N and P, conductivity and pH.

1.2.2 My role in relation to the NLP project

The case study farm was one of the market gardens that participated in the NLP project. I worked intensively on the case study farm, performing 4 roles:
1) Design: sampling strategies and flow control structures, methodology for land management data collection;

2) Field operations: equipment maintenance, sample collection, land management data collection;

3) Analysis and interpretation of land management and water quality data; and

4) Education of farmers regarding soil and nutrient management.

In many ways this research and the NLP project complemented each other. Regular maintenance of the monitoring equipment gave the author the opportunity to closely observe land management on the case study farm and develop a working relationship with the farmer. A good rapport was developed with the farmers and as a result they were willing to spend time discussing their land management. The detailed land management description generated from this research assisted interpretation of the runoff data. Interpretation was aided both for the case study farm which was not within the nested arrangement of monitoring stations in the Currency Creek subcatchment, and also for the other market garden within the nested subcatchment.

A field day for the NLP project led to the rainfall simulator being used in an experiment on the case study farm for this research. The rainfall simulation experiment conducted on the case study farm assisted the NLP project to:

1) develop the rainfall simulator as an extension tool for use in market gardening, and

2) increase awareness of the collaborating farmers of the role of land management in minimising runoff losses of soil and nutrients.

1.3 Thesis Structure

Chapter 2 discusses in detail some of the factors that have influenced location, size and management of market gardens in the Hawkesbury-Nepean catchment. Common management features are discussed. References are made to the case study farm so the reader can compare it with industry practices. Improvement of land management practices on market gardens will only be achieved if extension issues are carefully considered and these are discussed.
Chapter 3 reviews the literature on sediment and nutrient loss in agricultural runoff. Techniques for measuring sediment and nutrient loss are briefly discussed in 3.1. Current estimates of nutrient exports from market gardens are reviewed in 3.2. Soil erosion is discussed in 3.3. N and P sources and transformations in soils are reviewed in 3.4 and 3.5 respectively. The transport of N and P in runoff is discussed in 3.6. The potential for contour banks, dams, wetlands and filter strips on market gardens is outlined in 3.7.

Chapter 4 brings together physical and socioeconomic issues from Chapters 2 and 3 and presents the key research questions.

In Chapter 5, the materials and methods for the case study are described, including a general description of the farm, soil measurements, collection of land management data, stormwater monitoring and rainfall measurements.

Chapter 6 provides information to enable analysis of the farm-scale monitoring. It describes results from the soil and sediment sampling. Aspects of farm management on the case study farm that are likely to influence runoff quality and quantity including fertiliser inputs, and soil (plant) cover and surface soil condition are described in detail.

Results from the farm-scale monitoring of runoff are presented in Chapter 7. The quantities and forms of N and P in runoff are described and the factors that influenced sediment, N and P losses are discussed in light of Chapter 6.

Chapter 8 examine links between management and runoff at plot scale. An *in situ* rainfall simulator experiment was carried out to compare runoff volume from a green manure crop with the more extensive practice of bare fallow. Observations of interaction with the farmer that provide an insight into the farmer’s perception of land management are included.

In Chapter 9 the aims of the research are addressed. Recommendations are made for extension and adoption of improved land management practices.
2. PHYSICAL AND SOCIAL FEATURES OF THE MARKET GARDENING INDUSTRY IN THE HAWKESBURY-NEPEAN CATCHMENT

2.1 LOCATION OF MARKET GARDENS IN THE HAWKESBURY-NEPEAN CATCHMENT

Figure 2-1 indicates the main areas where market gardening is concentrated within the Hawkesbury-Nepean catchment.

Figure 2-1. Areas where market gardening is concentrated in the Hawkesbury-Nepean catchment
(Source: Jinadasa et al., 1997, showing sampling sites including site 29 - the case study farm)
2.1.1 Geology and Soils

Market gardening is carried out on 3 major soil types in the Sydney Basin (Murison, 1995):

1) Deep river loams of the flood plains which are well drained, highly fertile and a pH range of 6.0 - 6.5.

2) Loams derived from Wianamatta Shale formations which are less fertile than 1), are poor draining, and have pH values ranging from 5.0 - 5.3. Successful production has only been possible with the use of raised beds and application of organic and inorganic fertilisers. The case study farm falls into this category.

3) Hawkesbury sandstone derived soils found in the north-east (Gosford, Wiseman’s Ferry) area. These are well drained but infertile, with a pH range of 4.8 - 5.2 and respond well to fertilising.

Jinadasa et al. (1997) surveyed soils on 29 market gardens in the Hawkesbury-Nepean catchment and described the soils according to geological history and topography. Five categories were described:

1) strongly weathered clay loams, light clays and loams on Triassic shale, gently undulating topography;

2) mildly weathered clay loams, loams and loamy sands on a Recent river terrace;

3) moderately weathered sandy loams and sand on a late Pleistocene river terrace;

4) strongly weathered loamy sands on Triassic interbedded shales and sandstone, hilly topography;

5) strongly weathered sands and loamy sands on Triassic sandstone and sandy early Pleistocene alluvium.

2.1.2 Climate

Bureau of Meteorology records show that irrigation is essential for vegetable production in the Hawkesbury-Nepean catchment. Sydney and Katoomba receive an average annual rainfall of 1212 mm and 1412 mm respectively. However, much of the Hawkesbury-Nepean catchment, particularly where market gardening is most concentrated (and this is also where the case study farm is located) receives considerably less rainfall.
The average annual rainfall at Richmond is 822 mm and the average annual class A pan evaporation is 1402 mm. In all months but May and June, pan evaporation exceeds precipitation (Table 2-1). Generally, summer months receive more rainfall than winter months. Average monthly rainfall peaks in February with 103 mm, falling to 47 mm for July. On average there are 111 rain days per year.

Table 2-1. Average monthly precipitation, evaporation, maximum and minimum temperatures at Richmond (Source: Bureau of Meteorology)

<table>
<thead>
<tr>
<th>MONTHLY AVERAGE</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>Precipitation</td>
<td>98</td>
<td>91</td>
<td>90</td>
<td>69</td>
<td>60</td>
<td>62</td>
<td>47</td>
<td>44</td>
<td>43</td>
<td>57</td>
<td>72</td>
<td>76</td>
</tr>
<tr>
<td>Evaporation</td>
<td>177</td>
<td>143</td>
<td>132</td>
<td>96</td>
<td>60</td>
<td>50</td>
<td>59</td>
<td>88</td>
<td>109</td>
<td>150</td>
<td>160</td>
<td>181</td>
</tr>
<tr>
<td>Max temp</td>
<td>29.5</td>
<td>28.7</td>
<td>27.1</td>
<td>24.0</td>
<td>20.3</td>
<td>17.7</td>
<td>17.2</td>
<td>18.8</td>
<td>21.5</td>
<td>24.3</td>
<td>26.9</td>
<td>29.0</td>
</tr>
<tr>
<td>Min temp</td>
<td>17.5</td>
<td>17.5</td>
<td>15.7</td>
<td>12.1</td>
<td>8.0</td>
<td>5.3</td>
<td>3.6</td>
<td>5.1</td>
<td>7.6</td>
<td>11.1</td>
<td>13.9</td>
<td>16.1</td>
</tr>
</tbody>
</table>

Temperature variations are significant both seasonally and diurnally. Maximum daily temperatures average 29.5° C for January and 17.2° C for July. The average minimum daily temperatures for the same months are 17.5° C and 3.6° C respectively.

2.2 MARKET GARDENERS AND THEIR FARMS

2.2.1 Cultural backgrounds

The market garden industry in the Sydney region is strongly influenced by a number of cultures (Senn, 1996). Many of the farmers are of ethnic origin. Each of the ethnic groups generally farm different crops and use different farming practices. Maltese growers occupy larger farms and grow cabbage, lettuce, spinach, potato and summer crops. Italian growers generally occupy smaller farms and predominantly grow tomatoes. Anglo-Australians occupy larger farms on the flood plains in the Windsor area and grow potatoes, cauliflower and sweet corn. Chinese and Vietnamese growers occupy smaller farms (2 ha) and grow Asian vegetables and herbs. Lebanese grow cucumbers in greenhouses and commonly use hydroponic systems (Senn, 1996).
A detailed survey of Sydney region market gardeners was conducted by Murison (1995). He reported that Italian, Maltese and Anglo-Australian are the largest groups representing 30, 28 and 21% respectively. Senn (1996) however, states that just under 50% of the growers in the Sydney region are Maltese and that 70% of produce volume and cultivation area is farmed by Maltese. Neither Murison's or Senn's work specify whether these figures exclude second generation Australians. Italian market gardeners have moved into flower production and this may be the reason for the discrepancies in these statistics (i.e. if Murison's figures include Italian flower producers who occasionally grow vegetable crops).

Maltese growers have received their fair share of attention from NSW Agriculture. Murison described Maltese growers as "extremely interested in their farms" and their farms as nearly always neat, with high levels of cropping and fertiliser use. From 1993 to 1996, NSW Agriculture conducted a project with 10 market gardeners, 9 of whom were Maltese. The aim was to encourage the adoption of 'permanent' or 'semi-permanent' beds (Senn, 1996).

Market gardeners are regarded as hard working, commonly working from dawn to dusk, 6 days a week. Most farms are run by a family on a fulltime basis and little or no outside labour is employed (Senn, 1996).

2.2.2 Farm size and economies of scale

In Windsor and Camden areas there are over 1000 market gardens with a total area of 4478 ha (Murison, 1995) with an average size of 4.3 ha. Farms in the Sydney region range in size from 0.4 ha to 80 ha, but most are around 2 to 10 ha in size. According to Senn (1996) market gardens are larger now than several decades ago, and at least 8 ha are needed to support a family, unless hydroponics and greenhouses are used.

2.2.3 Agricultural practices

Fertiliser use

A key feature of market gardens is high fertiliser use. As mentioned in Chapter 1, poultry manure is used extensively on many market gardens in the Hawkesbury-Nepean catchment.
Jinadasa et al. (1997) reported that on 29 market gardens in the Hawkesbury-Nepean catchment, P inputs were typically 450 kg ha\(^{-1}\) yr\(^{-1}\) and generally 65 % was added in poultry manure (P≈290 kg ha\(^{-1}\) yr\(^{-1}\)). Assuming a N:P ratio of 3:2 (Tunney, 1980; Derrick, 1996) N input from poultry manure is equivalent to 435 kg ha\(^{-1}\) yr\(^{-1}\). Application rates of poultry manure range from 7.5 m\(^3\) ha\(^{-1}\) to 37.5 m\(^3\) ha\(^{-1}\) (A. Senn, pers. comm., 1995) and assuming a bulk density of 500 kg m\(^3\) and N and P concentration of 3.2 % and 2.15 % respectively (Embrey & Allan, 1984), this is equivalent to 120-600 kg ha\(^{-1}\) of N and 81-403 kg ha\(^{-1}\) of P. Generally poultry manure is not applied to crops in summer due to heat stress on crops.

A multitude of chemical fertilisers are used in conjunction with poultry manure, including urea, ammonium nitrate, general NPK fertilisers, superphosphate and others (A. Senn, pers. comm., 1996). Soil amendments such as lime and gypsum are used less frequently.

According to NSW Agriculture (1997) many market gardens with a history of intensive cropping in the Sydney region have soil P concentrations at least twice that required for optimal vegetable growth. NSW Agriculture recommends that soil with Bray or Colwell P >150 mg kg\(^{-1}\) should not be fertilised with additional P for at least 12 months (NSW Agriculture, 1997). Jinadasa et al. (1997) reported a median increase of 874 mg kg\(^{-1}\) total P (using HNO\(_3\)/HCL as the extractant) above background levels.

**Tillage**

Tillage practices vary according to soil type. On the well drained floodplains and on sandy soils raised beds are not needed, while on the poorly drained soils, raised beds are needed.

According to Senn (1996) in recent years, industry practices have changed from destroying beds completely after each crop, to the use of semi-permanent or permanent beds. Permanent or semi-permanent beds are raised beds that are used for a number of crops, but are usually cultivated on the top of the beds between crops. Previously, implements such as ploughs (mouldboard, chisel and disc) disc cultivators and large rotary hoes completely destroyed beds and often furrows (tractor laneways) too. The availability of small rotary hoes of about
100 cm width, in the mid 1980’s led to the widespread shift in tillage practices (ie. a reduction in tillage), particularly amongst Maltese growers (Senn, 1996).

The case study farm used semi-permanent beds.

**Irrigation practices**

Seventy percent of the farmers surveyed by Murison (1995) use spray irrigation and 25 % use drip irrigation. NSW Agriculture encourages market gardeners to use irrigation which leads to lower water use (NSW Agriculture, 1997). Surface and subsurface drip irrigation used 43-74 % of water used by furrow irrigation in California (Hanson et al., 1997). Yields of lettuce were more consistent from the drip than the furrow irrigation. Sammis (1980) found that on a clay loam and a sandy loam, the highest yield and water use efficiency was achieved with subsurface and surface drip irrigation, respectively.

The success of drip irrigation may vary with different crops and soil types, even from paddock to paddock. According to Sammis (1980) high yields and water use efficiency can be achieved on lettuce, using any irrigation method that can supply light, frequent irrigations. Lettuce was the major crop grown on the case study farm and overhead sprinklers were used.

### 2.2.4 Market Gardeners' perspective of the impacts of their enterprises in the Hawkesbury-Nepean catchment

A major issue confronting market gardeners is maintaining profitability in a competitive, changing industry. The combination of new varieties which have extended growing seasons and greater productivity from an increasing number of farmers, creates oversupply and consequently lower prices. Profitability is also threatened by pests and diseases, which are developing resistance to chemical sprays. The farmers generally have a poor understanding of pesticides and complain that chemicals "... are not as strong as they used to be" (Murison, 1995).

The expanding Sydney metropolitan area is encroaching on agricultural land in Western Sydney (Kelleher et al., 1997). Conflict between market gardeners and neighbouring land
users has arisen over issues such as chemical spray drift, noise pollution, and tree clearing (Senn, 1996). Environmental legislation and bureaucracy are also seen by the farmers as a threat (Senn, 1996). Not surprisingly then, the offsite impacts of their farming practices are not a high priority to market gardeners, unless they are related to productivity and profitability of their enterprise (eg. reducing tillage because of operating cost savings rather than minimising offsite impacts).

The fact is that economic viability of market gardens is not heavily reliant on minimising the loss of soil and nutrients. In a land use study of Hawkesbury Shire, Kelleher et al. (1997) found that market gardens are being established increasingly on relatively poor agricultural land, but are able to improve soil fertility with inputs of waste from the poultry industry. This probably contributes to a perception amongst the growers that soil is a renewable resource because poor soil can be improved with poultry manure. In his survey Murison (1995) found that 49% of market gardeners felt that there was little to no erosion on their farm, and nature was the cause of any erosion. Nearly 50% of the farmers had never had a complete soil test done despite the fact that only 32% said the cost was prohibitive. Over 92% said they would use soil and leaf tests if the results were received "quickly". Murison (1995) found that nearly 60% of the farmers surveyed had goals to either make more money or provide their family with a good life. Less than 11% had the goal to be a better farmer.

Obviously then, to achieve a better environmental outcome, soil, fertiliser and water saving practices must be cost effective. (This seems to be a major reason why Senn (1996) reported widespread adoption of reduced tillage practices. He reported that the use of semi-permanent beds (reduced cultivation) provided substantial savings in operating costs (fuel, tractor maintenance and labour) with no loss in production.)

Unfortunately Senn's 'permanent bed' project did not monitor erosion and/or nutrient runoff. The question remains, did the change in land management achieve a better environmental outcome and if so, by how much? While it most probably did, studies discussed in 3.6.1 show that reducing cultivation can actually lead to an increase in the concentration of soluble nutrients in runoff despite a substantial reduction in the total nutrient concentration, which consists predominantly of sorbed/particulate forms. Although the soluble and particulate
fractions may interact (e.g. adsorption/desorption) and non-bioavailable forms can become bioavailable (e.g. desorption of particulate P in anoxic conditions), the immediate result may well be an increase in algal growth in waterways.

The small size of most farms and low disposable income means that in effect, economies of scale present a financial impediment to ameliorating soil erosion and nutrient loss. That is, due to the small size of the enterprises, profitability is too marginal to warrant expenditure of time and money on ameliorative land management. However, farmers concerned only with maintaining or increasing productivity may have financial incentives to minimise losses of water, soil and fertiliser nutrients.

2.3 Extension Issues

2.3.1 Limited baseline data and stakeholder recognition of the extent of the problem

Landholders who underestimate losses of their soil and nutrients are less likely to improve their land management practices to conserve soil and water resources (Chamala et al., 1984). The role of land management in preventing soil and nutrient loss is subsequently underestimated. Given the lack of locally derived erosion and nutrient runoff data it is likely that market gardeners in the Sydney region would not be aware of the extent of soil and nutrient loss in runoff.

2.3.2 Limited resources for farmers' preferred information source: personal contact

Extension resources for the market gardening industry in the Hawkesbury-Nepean are stretched. NSW Agriculture employs 2 fulltime district horticulturalists who service over 1100 market gardens in the entire Sydney Basin and the Central Coast (Senn, 1996). In agriculture generally, one-to-one personal contact is the highest rated source of information and it also allows 2-way communication which can more effectively impact on the decision making process than less costly broadcast methods (Woods et al., 1993). In the Hawkesbury-Nepean catchment, other farmers are the single greatest source of advice sought (28 %) followed by NSW Agriculture (15 %). Other significant sources of advice are chemical suppliers (13 %), produce agents (11 %) and chemical and seed company representatives (10 %) (Murison,
1995). According to Murison, many market gardeners are willing to participate in farm experiments and this was the case in the work of Senn (1996) and the NLP project (including this work).

Current extension needs are compromised by limited financial resources. Providing personal contact is time consuming and therefore expensive. Due to dwindling financial resources the Australian trend in extension services is reduced interpersonal contact (Woods et al., 1993). Broadcast methods alone are an ineffective form of technology transfer (Woods et al., 1993).

2.3.3 The extension theory debate

The traditional extension approach, often called transfer of technology (TOT) and positivist thinking that underpins TOT has been criticised by workers who have used more progressive extension approaches based on constructivism (Hamilton, 1995; Senn, 1996). TOT assumes a one-way flow of information, for example, from researcher → extension officer → farmer → diffusion to other farmers. Participatory Learning and Action Research (PLAR or action research) acknowledges the reflective nature of adult learning and incorporates farmer participation in the research process (Hamilton, 1995; Senn, 1996). In complex environmental issues it is argued that both a high level of technical skills (hard systems) and people skills (soft systems) are required.

The 'permanent bed' project run by NSW Agriculture aimed to encourage reduced cultivation amongst a target group of market gardeners. The project was initiated partly out of concerns about the impacts of market gardening on the Hawkesbury-Nepean and was based on the premise that reduced cultivation would reduce deleterious environmental impacts. Senn (1996) reported adoption of semi-permanent beds by 7 out of 9 of the participating farmers and a greater interest in science, and attributed much of this success to the development of rapport with individual farmers. PLAR principles showed that reduced tillage is a viable land management practice. However, it can be criticised if only a small number of farmers can be reached. To reach a larger number of farmers, NSW Agriculture has produced leaflets, displays, and 'Best Practice Guidelines for Growing Vegetables' (NSW Agriculture, 1997) which publicise the benefits of reduced tillage.
A project based on PLAR principles which aimed to improve fallow management practices in the broadacre cropping areas of south-east Queensland, was run by the Viable Farming Systems Group (VFSG) of the Queensland Department of Natural Resources. It was based on research which suggested that soil erosion would be dramatically reduced by maintaining vegetation cover above 30% and using conservation tillage techniques such as direct drilling (Hamilton, 1995). Earlier TOT extension had had limited success (30% of farmers were using stubble retention fallow management). The VFSG extension program was largely based on the use of research tools such as a rainfall simulator and a soil corer, in an interactive process. For example, in their rainfall simulator demonstrations, the farmers nominated the treatments to be compared and to a large extent made their own interpretation of the results. Results of a farmer survey carried out after the demonstration showed that 90% of the farmers felt that the rainfall simulator would influence their future decision making on soil surface management (Cawley et al., 1992). Regional and local broadcast methods were used to attract farmers. About 3000 farmers participated in field days held on farms. A decision support tool called 'How Wet' was also incorporated into the extension program (Hamilton, 1995).
3. **LITERATURE REVIEW: SEDIMENT AND NUTRIENT LOSS IN AGRICULTURAL RUNOFF**

3.1 **TECHNIQUES FOR MEASURING SEDIMENT AND NUTRIENT LOSS IN RUNOFF**

A number of approaches can be taken to studying soil and nutrient movement in runoff from landscapes. The most appropriate approach depends on the purpose of the study and of course on budgetary limitations. Hudson (1993) reviewed a range of techniques for measuring soil erosion and runoff for plot and field studies.

Some inexpensive methods of measuring soil erosion include using point measurements of the soil surface to calculate soil loss or deposition, and making volumetric measurements from gullies, streambanks, and catchpits. Neil and Galloway (1989) measured sediment deposited in farm dams to estimate rates of soil erosion from a rural catchment.

If more information is required than soil erosion or deposition, it may be necessary to measure runoff, either as total volume or discharge (volume per unit of time) and also measure sediment/nutrient concentrations to determine loads. Runoff can be monitored at a range of scales, anything from small plots (e.g. 1 m²) to large catchments (>20 km²). Scale of monitoring is an important consideration for the type of equipment used. In plot studies the total volume of runoff can be collected, but this approach is obviously limited by the capacity for storing runoff. Alternatively, runoff can be channelled through a control structure (such as a flume or weir) or tipping bucket to instantaneously measure discharge. Tipping buckets are only practical for plots, for example a 20 L tipping bucket is suitable for a 600 m² plot (C. Ciesiolka, pers. comm., 1998). Automatic sampling can be performed by tipping buckets, rising stage samplers or by programmable automatic samplers which are now produced by several manufacturers.
Nutrients can occur in soluble and particulate forms and there can be a significant degree of transformation between forms. Cullen and O'Loughlin (1982) state that in any nutrient monitoring work it is therefore necessary to measure total concentrations and as many separate forms as possible. While runoff quantity can be monitored continuously with various water level or flow sensors, or tipping buckets, as yet there are no sensors capable of doing a full range of chemical analysis in situ. Therefore to accurately calculate total nutrient loads, a sampling strategy must be used to obtain a representative sample of runoff. Sampling can be incremented on time or flow. Time-based sampling is less accurate than flow-based sampling because most nutrient and sediment movement occurs in high flow events that occur infrequently (Geary, 1981; Olive & Walker, 1982; Cullen, 1991; Hairsine et al., 1993). For example, Cullen (1991) reported that 61% of P was transported in one percent of the time from the Monkey Creek Catchment near Lake Burragorang.

There is an increasing range of programmable equipment that can be used for flow-based sampling. Alternatively, rising stage samplers are cheap, but are limited to sampling on the rising limb of the hydrograph.

Loads can be calculated by multiplying the flow-weighted mean concentration by runoff, or by multiplying sample concentrations by runoff from corresponding sections of the hydrograph (Dann, 1986).

Field and farm-scale monitoring are more expensive than plot work, but some very useful data has been gathered, particularly in regard to the effects of land management. One ha contour bays on the Darling Downs were monitored by Queensland Department of Primary Industries (now Natural Resources) (Freebairn & Wockner, 1986; Freebairn, Wockner & Silburn, 1986). Runoff was channelled through H-flumes and V-notch weirs and samples were collected in rising stage samplers and by automatic samplers. The contour bays represented the broadacre farms more realistically than small plots, hence the results had greater application to farm management. For example, Freebairn and Wockner (1986) were able to assess the effectiveness of contour banks as well as different levels of soil cover.
Runoff plots have been used by many researchers, relying on natural or simulated rain to generate runoff. Rainfall simulators are reviewed in Chapter 8. Runoff plots have the advantage that treatments can be highly controlled and therefore, relationships between land management variables and runoff quantity/quality can be closely studied. Hudson (1993) argues that runoff plots are useful for:

- demonstration (extension) purposes, where more than one land management practice is compared;
- comparative studies; or
- to construct or validate models.

Plot data has been incorporated into catchment scale models. The Universal Soil Loss Equation (USLE) has been largely based on and validated for particular geographical areas using plot studies, hence it only applies to field scale (Rosewell, 1993). The Agricultural Non-Point Source pollution model (AGNPS) developed in the US incorporates the USLE with nutrient generation and assimilation data to estimate nutrient loads at catchment scale (Young et al., 1994)

Hudson is very sceptical about the value of plot studies, mainly because of the limitations of scale. Sumner et al. (1996) argue that a minimum plot size of 50 m² is needed to gain any understanding of sediment and nutrient movement. The critical factor is length rather than area because it is slope length that determines what processes are dominant.

There is an important difference between monitoring a 'real world' situation and monitoring an experimental plot or field. In an experimental situation land management practices are more easily controlled and therefore relationships between specific land management practices and runoff can be studied in isolation. The validity of a plot study depends on whether it represents the 'real world'. The larger the catchment is, the less chance there is that land management will be uniform. Therefore many variables come into play and it could be expected that relationships between specific land management practices and runoff are more difficult to identify. Cullen and O'Loughlin (1982) point out that in small catchments (<1 km²) some empirical measurements such as soil cover can be made. They also warn of problems
associated with trying to extrapolate from small catchments with uniform land management to large catchments with heterogenous land management.

Catchment monitoring is appropriate for determining nutrient loads or exports (load = kg, export = kg ha\(^{-1}\) yr\(^{-1}\)) from the catchment, but not necessarily for determining exports from the land uses which occur within the catchment. Determination of exports from land uses requires monitoring at an appropriate scale, so that inputs can be separated from surrounding land uses. If the average size of market gardens in the Hawkesbury-Nepean catchment is approximately 8 to 10 ha and they are generally not aggregated, then it would be logical to monitor catchments of this size. Monitoring larger areas would cause under-estimation of exports because of assimilation between the market garden and the monitoring point.

Cullen (1991) reported on a monitoring project undertaken in Monkey Creek catchment (part of the Lake Burragorang catchment near Sydney) comprised of forest, dairying, hobby farms, horticulture, aquaculture and small townships. The first objective of the project was to "...develop an approach to identifying the different contributions to pollutant movement made by different land use areas, and possibly different land management approaches." The project was initiated out of concern for water quality in the water storage and this influenced the scale of the catchments monitored. Six subcatchments were monitored (ranging in size from 10170 ha to 180 ha) in a nested arrangement where 4 stations were located on Monkey Creek itself and the other 2 were situated on tributaries. By nesting the sites within a catchment, contributions from subcatchments were assessed. However, streams may store and/or release nutrients thereby complicating interpretation. It is not sufficient to calculate exports from a land use within a catchment by dividing catchment exports by the portion of the catchment occupied by the land use (Marston, 1994).

Some soil types, such as soils derived from basalt, store large quantities of P. Erosion of these soils is potentially a mechanism for P movement. Caitcheon et al. (1995) used magnetic and geochemical tracing techniques on sediment in the Chaffey Reservoir and found that the major source of P in the Reservoir was eroded basalt-derived soil from a small area of the upper catchment.
Soil factors that determine P sorption include Ca, Fe, and Al content, to which P is strongly sorbed; amount and type of clay, to which P is sorbed; pH; and organic matter concentration (Chapman, Bliss & Smails, 1982; Sims & Wolf, 1994). Sandy soils tend to have a low capacity for P sorption. In these soils P moves more freely in soluble form.

3.2 CURRENT ESTIMATES OF NUTRIENT EXPORTS FROM MARKET GARDENING

Every review of the literature on exports of sediment and nutrients from agricultural land uses in Australia refers to the paucity of the data. Marston (1994) comprehensively reviewed Australian export studies and stated that less than 20 studies had been done, and of those, the thoroughness of their methodology and data accuracy varied widely.

Studies of exports from market gardening in south-eastern Australia have been limited to Mt. Lofty Ranges in South Australia. Some monitoring of market gardening did occur in the Hawkesbury-Nepean catchment, but the data were not suitable for determining exports. In the Monkey Creek project one of the subcatchments included market gardening. The 3060 ha subcatchment is comprised of grazing, market gardening, poultry farms and many large farm dams. Over 11.3 months, 766 kg of P were exported, equivalent to 0.25 kg ha\(^{-1}\) (Cullen, 1991). Clearly, the scale of monitoring was not appropriate for determining exports from market gardening.

The export values from CMSS are shown in Table 3-1. The values for market gardening are based on the work done in South Australia (Marston, 1994). The export rates are several times higher than for urban land.

In the Onkaparinga catchment in South Australia, Buckney (1979) reported exports of 14.4 kg ha\(^{-1}\) yr\(^{-1}\) for total P and 22.5 kg ha\(^{-1}\) yr\(^{-1}\) for nitrate-N and 54.4 kg ha\(^{-1}\) yr\(^{-1}\) for total N for one catchment that included market gardens and orchards, though sufficient explanation of land use is not given to determine the export from each. Note that these values are well in excess of the CMSS estimates. In Mt Bold Reservoir in South Australia, Holmes (1978)
reported N exports of 25.3 kg ha\(^{-1}\) yr\(^{-1}\) from market gardens with fertiliser inputs of 130 kg ha\(^{-1}\) yr\(^{-1}\). Seventy percent of the exported N was soluble nitrate (NO\(_3\)).

Table 3-1. CMSS estimates for land uses in South East Australia (Source: Marston, 1994)

<table>
<thead>
<tr>
<th>Land Use</th>
<th>P export [kg ha(^{-1}) yr(^{-1})]</th>
<th>N export [kg ha(^{-1}) yr(^{-1})]</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Range</td>
<td>Best estimate</td>
</tr>
<tr>
<td>Urban</td>
<td>0.4 - 3.6</td>
<td>1.0</td>
</tr>
<tr>
<td>Improved pasture</td>
<td>0.07 - 0.3</td>
<td>0.2</td>
</tr>
<tr>
<td>Unimproved pasture</td>
<td>0.05 - 0.07</td>
<td>0.06</td>
</tr>
<tr>
<td>Cropping</td>
<td>0.2 - 0.3</td>
<td>0.25</td>
</tr>
<tr>
<td>Forests</td>
<td>0.04 - 0.1</td>
<td>0.07</td>
</tr>
<tr>
<td>Market gardening-Horticulture</td>
<td>2.7 - 14.3</td>
<td>7.3</td>
</tr>
</tbody>
</table>

It is important to distinguish between soluble and particulate forms of N and P in runoff because the relative proportions of these greatly influence their fate and impact on ecosystems. Assimilation of particulate forms occurs more readily in the landscape than does assimilation of soluble forms (Finlayson & Silburn, 1996). Soluble forms are bioavailable and are more easily transported. The CMSS model does not differentiate between soluble and particulate forms of nutrients.

The values shown in Table 3-1 are given as export (kg per unit area per year). While export data from various sized catchments can be compared, the effect of scale should first be considered. Studies done on different scales are not easily comparable because different transport and deposition processes prevail. Overland flow processes differ from in-stream processes.

Prairie and Kalff (1986) reviewed the effect of catchment size on P export, within land uses. The land uses they compared were forest and agriculture. Agriculture was separated into mixed, non-row crops, row crops, and pasture. They found that in agricultural catchments total P export varied as the 0.77 power of catchment area, compared to 0.99 for forested
catchments. That is to say, in agricultural catchments as scale increases, P export per unit area decreases. Furthermore, they found the scale effect varied between agricultural land uses and was most pronounced in row crop catchments (power of 0.589). Comparisons can be drawn between erodibility of row cropping and market gardening. It is reasonable to expect that P is exported predominantly in particulate form from such land uses. Most P is exported in particulate form, adsorbed onto clay particles and organic matter, particularly when land management combines cultivation and fertiliser application (Beaulac & Reckhow, 1983). Therefore it seems logical to presume that scale has a strong affect on exports from market gardening because it is conducive to high sediment losses and sediment storage increases with increasing catchment size. Therefore export of sediment and sediment bound P decreases with increasing scale.

Export of nutrients is generally high where land use factors combine high rates of soil erosion and high rates of fertiliser application. Many plot and catchment studies have shown that fertiliser application results in a major increase in the export of nutrients (Cullen & O'Loughlin, 1982). In the Peel-Harvey estuary in Western Australia, Birch (1982) found that export of P from grazing land was positively correlated with rates of superphosphate application. A characteristic of market gardens in the Hawkesbury-Nepean catchment is their high fertiliser use. Jinadasa et al. (1997) found that P application rates are typically very high on the 29 market gardens that were surveyed in the Hawkesbury-Nepean catchment, with a mean input of approximately 450 kg ha⁻¹ yr⁻¹.

Geology and soil type are an important factors influencing P export (Marston, 1994; Caitcheon et al., 1995). Exports of P are likely to vary considerably between each of the soil types on which market gardening is carried out, because they have different background nutrient concentrations, different nutrient adsorption capacities and different drainage properties. On 29 market gardens in the Hawkesbury-Nepean catchment Jinadasa et al. (1997) found that the background P content in soils was low (Bray P generally < 20 mg kg⁻¹). Total P was greatly increased from fertiliser application (median increase of 874 mg kg⁻¹ of acid extractable P). Therefore it is reasonable to presume that P exports from market gardening in the Hawkesbury-Nepean catchment would be considerably higher than surrounding unfertilised land uses.
3.3 SOIL EROSION

3.3.1 Definitions and processes

Soil erosion by water was defined by Ellison (1947) as the process of detachment and transport of soil particles by an erosive agent. Meyer and Wischmeier (1969) expanded this definition to include 4 discrete processes: detachment by raindrops, transport raindrops, detachment by overland flow, and transport by overland flow. This definition ignores the interaction between these processes. When raindrops strike the soil surface, soil particles may be dislodged or displaced by smaller droplets that radiate out from the point of impact. This is referred to as rainsplash. Sheet erosion, or erosion by overland flow, is the term used to describe removal of a relatively uniform depth of soil by overland flow (Rose, 1993). Morgan (1986) states that the overland flow rarely occurs as a uniform sheet of water, but rather a mass of small braided water courses with no defined channel. Overland flow transports soil particles detached by rainsplash. The combination of these 2 processes is termed interrill erosion or rain-flow. Rill erosion occurs when flow is concentrated into small ephemeral channels up to 300 mm deep (McDonald et al, 1984).

Physiochemical attraction of fine particles to each other (determines aggregate formation and strength) creates resistance to erosion. Aggregate breakdown has been closely linked to raindrop impact (Wace & Hignett, 1991; Loch & Foley, 1994). Aggregate breakdown is an important process because fine particles which make up aggregates are easily transported once entrained (Olive & Walker, 1982).

Sutherland et al. (1996) studied interrill erosion under simulated rainfall on microplots 0.18 m² and separated rainsplash from sheet flow. They found that rainsplash detached coarse sand sized particles (500-1000 μm), significantly coarser than aggregates transported by overland flow. This indicates the difference in energy generated by rainsplash and sheet flow at this scale. They found, however, that the net movement of particles by rainsplash was highly dependent on slope. On a 20° slope, transport by sheet flow occurred in waves. Palis et al. (1990b) suggested that synergism occurs between the action of raindrops and runoff. They found that soil loss from the combination of raindrop action and overland flow produces soil loss greater than the sum of soil loss when the 2 processes act in isolation.
Singer and Walker (1983) reported that interrill erosion accounted for a minimum of 64 % of soil loss (fine sandy loam) from a 3 x 0.55 m flume, on a 9 % slope, under a range of surface cover conditions. On a sandy clay loam, Palis et al. (1990a) found that rainsplash was the only process that occurred on a 0.1 % slope, in a 5.8 x 1 m flume, with rainfall of 100 mm h⁻¹. On a 3 % slope, interrill erosion occurred. Loch and Donnollan (1983a) studied the prevalence of interrill and rill erosion on a non-swelling clay loam and a cracking black (Irvin) clay, in plots on a 4 % slope that were from 1.5 to 22.5 m long and 4 m wide. Tillage was orientated down and across slope and on the 22.5 m downslope plots, one furrow instead of 10 was created so that the volume of runoff in the furrow was effectively equivalent to a 225 m slope length. They observed that rill erosion did not occur on the shorter plots and deposition occurred in the furrows. On all but the 225 m plots rilling was not visually obvious, but sediment concentrations indicated that there was a sharp transition from interrill to rill erosion (i.e. a noticeably higher sediment concentration where rill erosion occurred).

Erosion and depositional processes are sediment size selective. Morgan (1986) argues that rainsplash is a highly selective process transporting particles between 63 and 250 µm, but this is contradictory to the findings of Sutherland et al. (1996). Loch and Donnollan (1983b) found that when the affect of aggregate breakdown was taken into consideration there was little or no evidence of size selective transport in rill and interrill erosion. Aggregate breakdown was more pronounced in interrill than rill erosion. There were more particles <2 µm in sediment transported by interrill erosion. Palis et al. (1990a) found that under rainfall detachment alone, eroded sediment was initially much finer than the original soil, and became coarser with time. After 40 minutes though, it was still considerably finer than the original soil. This difference was also attributed to aggregate breakdown. When runoff was combined with the rain drop action the sediment size distribution was similar to the original soil.

Gully erosion generally occurs as rills enlarge. A gully is defined as a steep sided, incised channel more than 300 mm deep and is more permanent than a rill (McDonald et al., 1984; Morgan, 1986; Rose, 1993). The definition of the 300 mm depth is derived from a more general definition that a gully cannot be traversed by tractor.
Gully erosion is a less selective process than rill and interrill erosion. Erosion is concentrated at the head of the depression, particularly at the base of the scarp, which results in deepening of the channel and undermining of the headwall. Erosion subsequently occurs downslope as streambank erosion, which is a combination of scouring and slumping, a form of mass movement (Morgan, 1986). Gully erosion is not of great importance on market gardens because farms are generally small and divided into drainage networks, thus, slope lengths are minimal. While gully erosion may be tolerated by market gardeners if it occurs on the fringe of their vegetable growing area, it is not likely to be a common occurrence on most market gardens in the Hawkesbury-Nepean catchment.

In flow, smaller particles are carried in suspension (suspended load), larger particles bounce, slide or roll along the soil surface (bed load) (Rose, 1993). A commonly cited relationship between velocity of flow and erosion-transport-deposition of particle size is shown in Figure 3-1. Generally, as flow increases larger particles can be transported. As flow decreases, increasingly smaller particles fall out of suspension.

The particle size of the suspended load varies according to the energy of flow, and generally ranges from colloid to fine sand (<2-1000 μm) (Olive & Walker, 1982). The bed load varies from gravel to silt depending on the energy of flow (Olive & Walker, 1982). As flow velocity increases, larger soil particles can be brought into suspension. In a study of plots up to 22.5 x 4 m, Loch and Donnollan (1983b) defined bedload and suspended load as particles larger and smaller than 20 μm, respectively. They found that the concentration of bedload varied considerably with time, due to intermittent supply and movement. In contrast, the concentration of suspended solids was relatively stable due to a continuous supply of particles <20 μm from aggregate breakdown by rain drop impact.

Deposition is important because of differences between deposited sediment and the parent soil. Rose (1993) argues that sediment deposited differs from the original soil; the deposited sediment is less cohesive than the parent soil particles. This is a useful distinction to make because deposited sediment will therefore be more erodible, than the more cohesive parent soil. Particles that are dislodged by rainsplash can then be transported by sheet or rill flow.
Palis et al. (1990a) observed a build up of a deposited layer of sediment similar to the original soil, after 35 minutes of simulated rain at 100 mm h\(^{-1}\) combined with run-on, on a 3 % slope.

![Graph showing relationship between particle size and velocity of flow](image)

**Figure 3-1. Relationship between particle size and velocity of flow**
(Source: Hjulstrom, 1935)

**Effect of scale on sediment export**

Olive and Walker (1982) presented Australian sediment export data from various catchment studies. The catchments ranged in size from 1.2 km\(^2\) to 139,000 km\(^2\) and exports ranged from 6.8 t ha\(^{-1}\) yr\(^{-1}\) from a 422 km\(^2\) catchment to 0.03 t ha\(^{-1}\) yr\(^{-1}\) from a 139,000 km\(^2\) catchment. When all the data are shown on a scatter plot there is no obvious relationship between sediment export and catchment size. Obviously there are many other variables such as climate, land use, and geology which also need to be considered. Olive and Walker (1982) also presented soil loss data (kg ha\(^{-1}\)) from Australian plot studies. Clearly where land management is comparable, the sediment exports are much higher in plot studies than in catchment studies. The results included some very high values including 353, 382, and 100 kg ha\(^{-1}\), measured over one year from small field sized plots in Goondi, Queensland. Very high values were also given from bare one ha plots in Gunnedah, NSW. As much as 180 kg ha\(^{-1}\) were lost from a single storm.
Drainage density and slope in relation to catchment size are illustrated in Figure 3-2. Figure 3-2 shows an idealised fluvial system, with distinct zones of sediment production, transport and deposition.

![Diagram of sediment production, transfer, and deposition zones along a stream profile over distance.]

**Figure 3-2. Idealised fluvial system** (Source: Schumm, 1977)

Sediment delivery ratio is the proportion of soil eroded within a catchment that is transported out of the catchment. Generally the sediment delivery ratio decreases with increasing catchment size. Prairie and Kalff (1986) cited 2 possible reasons for this:

- Drainage density (stream length divided by catchment area) is generally lower in larger catchments. Therefore in larger catchments eroded particles have further to travel and are thus more likely to be deposited before entering the stream.
- Catchment size and slope are generally inversely correlated and therefore larger catchments are more conducive to deposition.
3.3.2 Nutrient enrichment of eroded sediment

The energy of the depositional environment determines the size of particles deposited (Figure 3-1). As the energy of flow decreases, finer particles settle out of suspension. Therefore deposited sediment can be well sorted (consist of a small range of particle sizes).

Because finer particles have a larger surface area: volume ratio, they have a larger capacity to adsorb nutrients and agricultural chemicals. The nutrient concentration of eroded sediment can vary from that of the soil from which it was eroded. This is because organic matter and silt and clay particles which generally have higher concentrations of sorbed nutrients, are preferentially transported once entrained. The enrichment ratio ($E_R$) is the concentration of the nutrient in eroded sediment over that of the original soil. The $E_R$ reflects the particle size selectivity of erosion processes and is therefore influenced by prevailing erosion and deposition processes (Palis et al., 1990a&b; Finlayson & Silburn, 1996).

Theoretically, particles eroded from a market garden will become progressively finer as they are transported through the catchment. As the particles are transported along the catchment and the slope decreases and velocity of the water column slows, the particle size distribution will become progressively finer. Therefore, $E_R$ will increase.

Palis et al. (1990a&b) studied $E_R$ for N in sediment eroded from a sandy clay loam under rainsplash, overland flow and a combination of the 2 (interrill). $E_R$ for rainfall detachment alone decreased from 1.95 after 0.6 minutes to 1.17 after 35 minutes. On interrill erosion $E_R$ decreased from 1.42 after 0.6 min to 0.95 after 35 min. The decrease in $E_R$ was explained by a gradual build up of coarser sediment on the soil surface. Presumably the finer sediment was transported from the plot and $E_R$ was greater than unity. The lower values for interrill erosion indicate that it is slightly less size selective than rainfall detachment alone.

Values of $E_R$ range from less than one to as high as 10. Highly erosive conditions when all particle sizes are eroded equally on well aggregated soil produce values close to one. Conditions that limit erosion, such as high cover percentage, on light textured or poorly aggregated soils produce high $E_R$ values because, in these circumstances fine particles are easily detached/entrained and transported and coarse 'nutrient-poor' particles remain.
According to Finlayson and Silburn (1996) the values for interrill erosion are 2 to 3 times higher than rill erosion.

### 3.3.3 Factors that Influence Erosion

The "Universal Soil Loss Equation" (USLE) is based on empirically derived data. The inputs are factors that influence sheet and rill erosion: rainfall erosivity (R), soil erodibility (K), slope length and gradient (L and S), crop practice (C), and conservation practice (P). The equation is:

\[ \text{Soil loss} = R \times K \times L \times S \times C \times P \]  

[1]

The USLE is not really universal. It is only applicable to sheet and rill erosion, in particular locations where the empirical studies from which it is derived, initially in the US. It has been adapted to inland NSW (Rosewell, 1993). Despite its empirical basis there is wide acceptance that the factors identified do influence soil erosion and the USLE has been used as an educational tool. The following discussion is based on the USLE.

### Rainfall

Total rainfall is obviously a significant influence on runoff generation and soil erosion. Seasonal rainfall distribution and annual variations in rainfall have been correlated with erosion (Weaver, Pen & Reed, 1996). What annual and monthly rainfall totals do not necessarily indicate are the characteristics of rainfall that determine its erosivity, *viz.* intensity and kinetic energy (Kinnell, 1983; Goff, Bent & Hart, 1994). Intensity (*I*) is usually expressed as mm h\(^{-1}\). The kinetic energy of rainfall (*E*) is expressed either as energy per unit area per unit time (J m\(^{-2}\) s\(^{-1}\)) (*E_k*) or as energy per unit area per depth of rain (J m\(^{-2}\) mm\(^{-1}\)) (*E_\theta*) (Kinnell, 1987).

The terminal velocity of water drops is a function of their size, and therefore the energy of natural rain can be calculated from drop size data, using equation [2] (Kinnell, 1987).

\[ E_k = \frac{\rho \pi}{12 At} \sum_{j=1}^{n} N_j D_j^3 V_j^2 \]  

[2]
where \( N \) is the number of drops, \( D \) is the equivalent spherical diameter of the drop, \( V \) is the velocity, \( t \) is the time period over which the sample is taken, \( A \) is the sample area, and \( \rho \) is the density of water.

A rainfall distrometer is a device that is used to continually measure the impact of drops hitting a transducer, so that the size of the drop can be determined from the strength of the pulse. Studies of rainfall kinetic energy have been done in Australia using distrometers, including 2 in eastern Australia (Rosewell, 1986; Kinnell, 1987) that collected data at Brisbane, Canberra and Gunnedah. Kinnell (1987) compared \( I-E_B \) relationships between these 3 locations and Miami and Zimbabwe. According to Kinnell (1987) there is no substantial difference between the \( I-E_B \) relationship at Canberra and Gunnedah. The only coastal data are for Brisbane (Rosewell, 1986) but there is doubt about the accuracy of the data because of technical reasons related to the particular distrometer used (Kinnell, 1987). Figure 3-3 shows the \( I-E_B \) relationship for Canberra. The data exhibit variability which is due to different types of rainfall (convectional, orographic, frontal uplift) and wind on drop size. The distribution of data demonstrates that high intensity rainfall is relatively infrequent. Data from overseas studies suggest \( E_B \) actually decreases slightly at intensities exceeding 80-100 mm hr\(^{-1}\) (Hudson, 1993). No \( I-E_B \) studies have been done in the Sydney region. Examination of Figure 3-3 shows that \( E_B \) rapidly increases from 0 to 20-25 mm h\(^{-1}\) and then tapers off.

![Figure 3-3. Intensity-energy data and relationship for Canberra](Source: Kinnell, 1987)
Rainfall intensity and kinetic energy are positively correlated with erosion (Kinnell, 1983). The $E$ for $I > 25$ mm h$^{-1}$ was proposed by Hudson (1965) as alternative index for $EI_{50}$ (the maximum amount of rain falling in any 30 minute period) used in the USLE. Hudson proposed $I > 25$ mm h$^{-1}$ as an alternative $E$ index on the basis that there is a need to differentiate rainfall that exceeds the rainfall acceptance ($I_s$) of the soil. Kinnell (1983) suggested that the rate for the $I$ index should actually be tailored to suit particular soil types (ie. high $I$ value for high $I_s$). $I > 25$ also ignores the fact that infiltration rate is influenced by antecedent soil water.

Kinnell (1983) measured runoff from 0.1 ha plots and rainfall intensity at one minute intervals ($I_t$) to determine periods during a storm when $I$ exceeded the acceptance rate ($I_s$) of the soil (the average rate at which rainfall can infiltrate into the soil). $I$ was compared with runoff from a plot. $E$ was calculated from drop sizes, measured with a distrometer. The soil was a yellow podzolic and was in bare fallow. The slope was 4.1-4.3 %. Kinnell reported that $I_s$ frequently lay between 2-6 mm h$^{-1}$ and attributed this low rate to surface sealing (see following discussion).

**Soil properties**

Important soil properties that influence erosion are particle size distribution and structural stability (Hignett, 1991; Wace & Hignett, 1991; Rosewell, 1993; Guerra, 1994). These properties also influence the infiltration rate which affects the volume of runoff, and the availability of particles to detachment and entrainment. Antecedent soil moisture conditions also affect infiltration by determining pore space and microlief storage (McIsaac & Mitchell, 1992; Rudolf, Helming & Diestel, 1997). Figure 3-4 is a conceptual framework of factors that affect soil erodibility.

**Soil texture**

The particle size distribution or texture is closely related to the mineral composition and texture of the parent rock. The texture can be modified by selective erosion and deposition of fine sediment during flooding, and mixing of differently textured soil horizons during cultivation (Murtha, 1988).
Figure 3-4. Conceptual framework of factors affecting soil erodibility

The nomograph can be used to determine the K factor for a given soil for the USLE, if certain soil qualities are known. The inputs required are the percentage of silt and very fine sand, the percentage of organic matter, soil structure classification and permeability classification. The SOILOSS program (Rosewell, 1993) suggests soil erodibility (K) values based on texture classes if data for the nomograph are unavailable. Some of these are shown in Table 3-2. The fine sandy loam and silt loam are the most erodible texture classes.

<table>
<thead>
<tr>
<th>Texture</th>
<th>sand</th>
<th>loamy sand</th>
<th>loam</th>
<th>fine sandy loam</th>
<th>silt loam</th>
<th>clay loam</th>
<th>silty clay loam</th>
<th>sandy clay</th>
<th>heavy clay</th>
</tr>
</thead>
<tbody>
<tr>
<td>Suggested K value</td>
<td>0.015</td>
<td>0.020</td>
<td>0.040</td>
<td>0.050</td>
<td>0.055</td>
<td>0.030</td>
<td>0.040</td>
<td>0.017</td>
<td>0.012</td>
</tr>
</tbody>
</table>

In a study of aggregated clay soils, Loch and Pocknee (1995) found that the K factor is more accurate if derived not from the nomograph, but from the proportion of particles <0.125 mm combined with the wet density of sediment. The particle size distribution can have a major influence on the infiltration rate and therefore, the generation of runoff. Generally, sandy soils
have high permeability, and soils with a high percentage of clay have low permeability, but this depends on the type of clay and soil structure (Murtha, 1988).

Particle and aggregate density influences erodibility. Soil particles are denser than aggregates of the same size and aggregates of the same size can vary in density between soils. Denser particles are less easily transported. Bedload transport for swelling clay Vertisols can be 80 % higher than in oxisols because of lower aggregate density. Loch and Donnollan (1983b) estimated the wet density of vertisol aggregates to be 1.42 Mg g⁻¹ and 1.76 Mg g⁻¹ for oxisol aggregates.

Clay dispersion, aggregate slaking, aggregate stability, and surface sealing
Soil structure refers to the distinctiveness, size and shape of aggregates (also called peds) (Murtha, 1988). Aggregate stability encompasses the resistance of aggregate breakdown to slaking and dispersion. Slaking refers to the disintegration of aggregates into their constituent particles due to physical mechanisms. Dispersion is the repulsion of clay crystals away from each other. This occurs when lowered concentration of the bulk-electrolyte solution causes exchangeable ions to diffuse away from a tightly packed configuration or due to the exchange of cations which have tightly packed hydration shells for cations with larger hydration shells which fit more loosely around clay particles (Hamblin, 1987). Typically, this occurs when Na⁺ replaces Ca²⁺. Water stable aggregates are stable in a solution of distilled water (Emerson, 1967) implies a resistance to dispersion.

The degree of breakdown depends on the aggregate strength to slaking and dispersion when impacted upon by raindrops (Loch & Donnollan, 1983b). During this process, fine particles are translocated to fill pores in the top few millimetres of the soil. A layer of reduced hydraulic conductivity (HC) called a surface seal forms (Morgan, 1986; Hignett, 1991). Surface sealing reduces the infiltration capacity of the soil, increasing runoff (Morin & Van Winkel, 1996).

Rainfall energy is important in causing surface sealing as identified in rainfall simulator studies (Loch, 1982; Wace & Hignett, 1991; Loch & Foley, 1994; Hignett et al., 1995). The work of Loch (1982) and Loch and Foley (1994) involved field rainfall simulators. Loch (1982) compared infiltration on an Irving clay under 80100 and 8070 nozzles, which produce different
drop sizes. The result was that surface sealing occurred only under the 80100 nozzle, even though a higher intensity was used for the 8070 nozzle (made possible by varying the dwell time). He implied that total energy (i.e. irrespective of intensity) is the main cause of surface sealing. This theory was supported by Wace and Hignett (1991) who used a laboratory simulator, with controllable drop size on soil samples representing a number of soil types. However, Hignett et al. (1995) suggested that kinetic energy per depth of rain also influences surface sealing.

The major affect of surface sealing is that infiltration is reduced and therefore infiltration rate is sometimes used to determine whether surface sealing is occurring. For example, infiltration rate was used to compare surface sealing in 19 soils, by Stern, Ben-Hur, and Shainburg (1991).

Surface sealing could be an important factor on some of the soil types that market gardening is carried out on in the Hawkesbury-Nepean catchment. On soils where surface sealing is a problem, the degree of sealing would be reduced by maximising vegetative cover, therefore increasing infiltration. This would be benefit farmers by improving the efficiency of irrigation. Runoff from storms would be reduced, thereby reducing the loss of soil and nutrients.

After finding that lowered HC was due to aggregate slaking rather than clay dispersion, Abusharar (1993) hypothesised that clay dispersion on dispersive soil takes place at the ultimate stage of aggregate slaking. Rudolf et al. (1997) compared 5 simulated rainstorms of 12 mm each applied at 30 mm h⁻¹ at one week intervals, with one continuous 60 mm storm at the same intensity. The results showed that aggregate stability affected microrelief and that microrelief was less stable in a wetting-drying-wetting cycle. They suggested that this was due to aggregate slaking caused by the escape of air during rapid wetting.

Traditionally, exchangeable Na % (ESP) >15 is considered to be the threshold above which dispersion occurs, but it has been shown that ESP has an almost linear relationship with aggregate stability (Crescimanno, Iovino & Provenzano, 1995). Levy and Torrento (1995) also found that ESP affected dispersion on a smectitic soil. They found that increasing ESP from 0.5 to 5.5 resulted in an increase in dispersed clay from 22.2 - 30.4 %. However, Stern et al. (1991) found no trend between ESP and dispersion (as measured by infiltration rate).
Dispersion varies with clay type; kaolinite is less dispersive than illite and smectite. Stern et al. (1991) found that kaolinitic and illitic soils which contain small amounts of smectite, are dispersive and soils containing only kaolinite are stable. Seta and Karathanasis (1996) found that soils dominated by kaolinitic minerals are less dispersive than those dominated by minerals with greater charge. Barzegar et al. (1997) found that Na adsorption ratio (SAR) and organic carbon (OC) influenced dispersibility and aggregate formation in smectite clay, more so than in illitic loams.

Chorom, Rengasamy and Murray (1994) reported a positive relationship between pH and the percentage of dispersible clay in Alfisols, Oxisols, Aridisols, and Vertisols, and attributed this to changes in the net charge of clay particles. Stern et al. (1991) found that pH of dispersive soils is slightly, but significantly higher than stable soils.

Structural characteristics of soil and the stability of aggregates influence the infiltration rate of soil. The spaces between aggregates provide pathways for water entry, so structure is usually associated with permeability. However, swelling clays with a strong structure may have high initial water acceptance when dry, but subsequent swelling leads to closure of pores, hence low saturated HC (Murtha, 1988). Soils with high levels of exchangeable Na often have strong columnar, blocky or prismatic structure. However, such soils often have low permeability (Murtha, 1988).

**Organic matter**

It is a widely held view that OC content strongly influences soil structure. Guerra (1994) and Watts and Dexter (1997) reported that OC correlates with aggregate stability. According to McTainsh (1993) humus which is derived from organic matter, combines with clays to form a clay-humus complex.

The role of OC however is complex and some studies have shown that it does not necessarily contribute to aggregate stability. Gu and Doner (1993) found that organic polyanions, especially humic acid, can act as dispersive agents in Na-clays and soils. In their study, treatment of Na-soils with an anionic polysaccharide increased aggregate stability as measured
by wet sieving, but redispersion occurred after leaching with distilled water. Packer, Hamilton and Koen (1992) found poor correlation between water stability of aggregates and OC on red and yellow duplex soils ($r^2$ values of 0.15 and 0.44 respectively ($p<0.01$)).

**Slope length and gradient, and topography**

Slope gradient and length influence the velocity and volume of runoff and therefore the erosive power and transport capacity. Morgan (1986) describes the relationship between soil erosion and slope as:

$$Q_s \propto \tan^m \theta \; L^n$$  \hspace{1cm} [3]

where $Q_s$ is sediment discharge per unit area, $\theta$ is the gradient angle and $L$ is the slope length. Morgan (1986) quotes values of 1.4 and 0.6 for $m$ and $n$ respectively, but reports that other studies indicate that $m$ and $n$ are sensitive to other factors in the erosion-slope relationship. According to Freebairn, Loch and Silburn (1996) the exponent for $L$ varies between 0 - 0.9.

Morgan (1986) cites work done in tropical environments that suggests a value for $m$ of approximately 2 and other work on different shaped slopes on 3 m plots, and suggests there is an increase in $m$ in the order of concave $<$ straight $<$ convex. This order may also hold for larger scale fields and catchments, where sediment export is lower from concave landform terrains than convex landform terrains.

Loch (1982) suggested that the exponential relationship between gradient and erosion on 3 x 0.6 m plots is the transition from rill to interrill erosion at approximately 10 % gradient.

Slope gradient can to some extent be controlled by the angle of cultivation of row crops in relation to contours of the landscape. Slope length is reduced by contour banks. The presence or absence of contour banks and the orientation of tillage to slope is incorporated into the P factor in the USLE. By cultivating across slope, runoff is directed on a more gentle slope. Loch and Donnollan (1983a) found that tillage across slope delayed and reduced runoff and reduced soil loss on plots up to 22.5 m on a 4 % slope. The major disadvantage of cultivating across slope is increased risk of failure in intense and prolonged storms.
From a soil conservation point of view it might be desirable for market gardeners to cultivate parallel to the contour. On less well drained soils in the Hawkesbury-Nepean catchment market gardeners would be reluctant to cultivate on the contour, not only because of the risk of failure in heavy storms, but also because the reduced drainage would lengthen the time after heavy rain that the beds remain unworkable, ie. they would remain 'muddy' for longer.

Vegetation and soil cover

Several studies have found that surface cover, whether it be in the form of vegetation (Greene, Kinnell & Wood, 1994), crop stubble (Bradford & Huang, 1994; Malinda, 1995), or rocks (Benkobi, Trlica & Smith, 1993), or a combination of these (Costin, 1980), has a profound influence on soil erosion. There are 2 mechanisms by which this occurs (Moss, 1989):

- interception of rainfall, thereby reducing rainsplash and surface sealing; and
- providing resistance to overland flow thus slowing surface water velocity and generating turbulence which hinders concentration of flow, thus minimising rill development.

Raindrops which are intercepted by vegetation and do not travel to the ground surface via stemflow, or return to the atmosphere via evapotranspiration, may reach the ground either as large gravity drops that are released at zero velocity from low points on the plant surface, or as smaller impact droplets ejected at relatively high velocities from the point of interception. Gravity drops have a higher terminal velocity and can therefore be highly erosive if the fall height is sufficient. Moss (1989) illustrates that a 5 mm drop has a terminal velocity in excess of 10 m s\(^{-1}\) and will reach approximately 5 m s\(^{-1}\) after falling only 1 m.

Because of their size, impact droplets decelerate rapidly to their terminal velocity and are virtually non-erosive (Moss, 1989). Moss and Green (1983) reported that drops 0.81 mm in diameter transported 1/7 that of 1.27 mm drops.

In the absence of a ground cover or litter layer, vegetation structure determines the proportion and energy of potentially erosive gravity drops and less erodible impact droplets that reach the soil surface. Short, fine grasses are highly effective drop energy dissipaters, but tall plants release trapped intercepted water as erosive gravity drops (Moss, 1989).
Soil loss data from field scale plots on the Eastern Darling Downs in Queensland, under various soil cover conditions is shown in Figure 3-5, which indicates that erosion is negatively and exponentially correlated with the percentage of cover. Reduced sediment concentration in runoff under increasing cover is due to reduced raindrop impact and reduced erosive power of overland flow.

![Graph showing soil loss ratio vs. cover percentage](image)

**Figure 3-5.** Soil loss ratio as influenced by cover on Vertisols of the Darling Downs compared to the relationship derived from the USLE (Source: Freebairn *et al.*, 1996)

Figure 3-5 shows similar cover-soil loss relationships between the work in Queensland and the USLE and indicates that Vertisols are more responsive to soil cover than most soils (Freebairn *et al.*, 1996).

Freebairn and Wockner (1986) found linear negative correlation between cover (%) and runoff (as percentage of rain) at given levels of antecedent soil moisture. The effect of cover on runoff coefficients (proportion of rainfall that became runoff) was negligible when antecedent moisture was very high, hence when the soil profile is saturated, surface cover has little hydrological affect. Cover has a more pronounced influence on infiltration under high energy/intensity rain. Freebairn *et al.* (1996) found that on 2 cultivated treatments under simulated rainfall at 100 mm h⁻¹, infiltration was in excess of 75 % of applied rain on 100 % cover and about 20 % of applied rain on 0 % cover.
Freebairn and Wockner (1986) recommended cover, provided by either a standing crop or crop residue, maintained above 20 to 30% would achieve optimal reduction in soil loss and runoff. As yet, there are no guidelines for maintenance of minimum cover levels in the market gardening industry.

NSW Agriculture is comparing 5 alternative market gardening farming systems in a plot study at the Somersby Research Station. One of the practices being trialed is the use of permanent cover crops, into which vegetables are grown. According to T. Wells (pers. comm., 1997) this practice produces much less soil loss, although the productivity of this system is less than that of 'district' (traditional) practice.

**Tillage**

The effect of tillage on erosion has also been studied quite intensely and it is clear that tillage increases erosion. This occurs by physically aiding particle detachment and aggregate breakdown (Bradford & Huang, 1994; Gaynor & Findlay, 1995) and causing oxidation of organic matter which results in reduced aggregate stability (Guerra, 1994; Malinda, 1995). Malinda (1995) related the amount of stubble retention to OC in the top 4 cm of soil and soil loss, using simulated rain on 1 m² split plots. In the same study, soil loss was related to soil OC content in the top 3 cm of the profile, by equation [4] (r² = 0.87).

\[
\text{Soil loss (t ha}^{-1}\text{)} = 31 + 16 \times \text{OC} - 45 \times \sqrt{\text{OC}}
\]  

[4]

In long term tillage trials at Cowra on a hard setting sandy loam and a hard setting loam, Packer et al. (1992) found that runoff was significantly reduced in minimum tillage treatments, due to the development and maintenance of macropores. In some situations however, tillage can increase infiltration by breaking crusts and creating a rough, porous surface. The degree of tillage determines its affect on runoff, as more intensive tillage makes the clods progressively finer, thereby reducing depression storage (Freebairn et al., 1996).
Research into the influence of cover and tillage on erosion has been reflected by a significant extension push towards reduced or conservation tillage and stubble retention in broad acre agriculture (Cawley et al, 1992).

3.4 NITROGEN SOURCES AND TRANSFORMATIONS IN SOILS ON MARKET GARDENS

Chemical fertilisers contain nitrogen in various forms: ammonium-N (NH₄⁺), nitrate-N (NO₃⁻), and urea (CO(NH₂)₂). Organic fertilisers also contain various forms of N. Figure 3-6 shows the forms and fate of N in poultry manure, a source of N commonly used on market gardens.

![Diagram of nitrogen transformations](image)

**Figure 3-6. Forms and fate of nitrogen in poultry manure** (Source: Sims & Wolf, 1994)

Complex forms of organic N include feathers and undigested food. Labile organic N is uric acid, which is rapidly hydrolysed by the enzyme uricase to urea, which is then transformed to gaseous ammonia (NH₃) (Zublena, Barker & Carter, 1993; Sims & Wolf, 1994). Nitrate is generally absent in fresh manure, but forms in the nitrification process (Sims & Wolf, 1994).

3.4.1 Volatilisation of ammonia NH₃

Volatilisation can result in significant losses of N in poultry manure when it is applied to the soil surface. The literature shows that losses of ammonia can be reduced significantly by incorporating manure into the soil, thereby decreasing exposure to the atmosphere. When
NH₃ is mixed into the soil it changes to NH₄⁺ which can be temporarily adsorbed onto clay particles and organic matter (Zublena et al., 1993). According to Sims and Wolf (1994) 50% of total N in poultry manure can be lost via volatilisation when it is surface applied.

Tunney (1980) reported that 90% of ammonia in surface applied manure, volatilised within 4 days. The losses were drastically reduced when the manure was incorporated into the soil. Schilkegartley and Sims (1993) reported between 4 and 31% of total N in surface-applied poultry manure was volatilised, but that losses of NH₃ were significantly reduced when the poultry manure was incorporated, particularly when it was incorporated immediately after application.

Conditions that are conducive to volatilisation include high temperature, sufficient moisture, high pH, and strong wind (Sims & Wolf, 1994). Robinson and Sharpley (1995) attributed changes in runoff losses of NH₄ from poultry litter to an increase in volatilisation at 35°C compared to 4°C.

Market gardeners typically store poultry manure in an uncovered stockpile, before application. Usually when it is applied, it is rotary hoed into the soil. Presumably, the loss of N from stockpiled poultry manure due to volatilisation occurs mainly from the surface of the stockpile and therefore should be negatively exponentially related to the storage period. The period between spreading and incorporation will be an important factor determining loss of ammonium, because at this stage the poultry manure is highly exposed to the atmosphere.

### 3.4.2 Mineralisation/nitrification

Mineralisation is the microbial conversion of organic N to inorganic N (ammonium, NH₄⁺) and nitrification is the sequential oxidation of NH₄⁺ to nitrite (NO₂⁻) to nitrate (NO₃⁻) (Sims & Wolf, 1994). Of N remaining in the soil after volatilisation, NH₃ tends to be the end product, which is then available for biological uptake, denitrification or leaching (Baird, 1996).

Hadass et al. (1983) applied pelleted poultry manure to clay and sand soils at 25°C and found that 42 to 54% of total N mineralised within 13 weeks. Nitrification is restricted by low
oxygen levels, low temperature, inadequate moisture, pH extremes (<5 or >8) and toxic levels of NH₃ (Alexander, 1977).

### 3.4.3 Denitrification

Denitrification is the conversion of NO₃ or NO₂ to gaseous N₂ or N₂O by microbial activity (Sims & Wolf, 1994). Denitrification losses are greatest in poorly drained, inundated soils with high levels of organic matter. Jarvis et al. (1991) found that denitrification occurred in short bursts after rain or irrigation. Cooper et al. (1984) applied poultry manure to a clay loam over a 5 year period and after 7 years found that 51 to 58 % of applied N was lost. They postulated that denitrification was the major mechanism for loss because the soil was seasonally inundated. Denitrification might therefore occur on market gardens after prolonged wet periods or excessive irrigation. Presumably, this would cause a fall in the concentration of NH₄ in runoff.

### 3.5 PHOSPHORUS SOURCES AND TRANSFORMATIONS IN SOILS ON MARKET GARDENS

#### 3.5.1 Bioavailability of P in runoff

The tendency of P to be adsorbed to sediment is fortunate from a water quality perspective, because this minimises the amount of P available to uptake by algae. However, under certain conditions desorption can occur so it is important to consider total P as well as the soluble and particulate forms that comprise it. Although particulate P is considered to be not immediately available to plants for biological uptake, there are mechanisms by which particulate P can be desorbed. Hence in the long term, erosion of particulate P is equally important as losses of soluble P, even though in the short term, it is the soluble, bioavailable fraction which is important in regards to algal blooms in water ways.

There is disagreement about what constitutes bioavailable P in runoff. Sharpley (1993) tested and proposed the use of ironoxide impregnated paper strips as a means of measuring bioavailable P in both soil and runoff. The strips are mixed with a >50 mL unfiltered runoff
sample for 16 hours. The P is removed with 0.1 M H₂SO₄. This implies that P sorbed to colloids and organic P in runoff water is to some extent bioavailable.

3.5.2 Forms and sources

Phosphorus generally is not readily available for biological uptake. Depending on the P sorption capacity of the soil, most P is adsorbed onto clay particles or incorporated into organic matter, as opposed to being in a soluble form. Black (1968) divides P in soil into 3 categories, all of which originate from orthophosphate (PO₄³⁻):

1) inorganic compounds in which P is combined largely with Ca, Mg, Fe, Al, and with clay minerals;
2) organic compounds that are present in plant and animal residues and the products of microbial synthesis; and
3) organic and inorganic compounds that are contained within the cells of living organisms, which are regarded as the link between 1) and 2).

It is important to discriminate between forms which are soluble and insoluble. Soluble P can be divided into molybdate reactive and non-reactive. Soluble reactive P (SRP) is sometimes referred to in the literature as orthophosphate (ortho-P) PO₄³⁻. Soluble non-reactive P includes organic P which is too fine to filter. Particulate P is comprised of a small labile portion (readily mineralised by microbial activity) that is exchangeable with inorganic P in solution. The major portion of particulate P is relatively inert, comprising crystalline, occluded, and organic P (Johnson et al., 1976; Sims & Wolf, 1994).

Organic P constitutes 30 to 50 % of total P in most soils (Paul & Clark, 1989). Sharpley and Smith (1995) studied the effect of adding beef, poultry, and pig manure on 20 soils over 35 years. The application rates were equivalent to 100-1000 kg ha⁻¹ yr⁻¹ of N and 37-270 kg ha⁻¹ yr⁻¹ of P, well below the P input of 450 kg ha⁻¹ yr⁻¹ as found by Jinadasa et al. (1997) on market gardens in the Sydney region. The manure applications increased the amount of organic N in the soil, but had little influence on the distribution or availability of inorganic forms of N. In contrast, there were large soil stores of bicarbonate extractable inorganic P which is generally considered to be bioavailable P. There was also an average increase in labile (readily mineralisable) organic P of 162 %. Overall, in untreated soils,
organic P was 64% of the total P, and in the treated soils, inorganic P was 60% of total P. Sharpley, Smith and Bain (1993) also studied N and P in soils amended with poultry litter for 12 to 35 years. They compared P and N levels in treated and untreated soils. The P application rate was equivalent to 90 kg ha⁻¹ yr⁻¹. In the top 5 cm of the soil, Bray P (an acid extractant (HCl) used to estimate plant available P, consisting of soluble inorganic P) averaged 188 and 9 mg kg⁻¹ respectively for the treated and untreated soils. The difference diminished rapidly below 5 cm depth.

This suggests that soils on market gardens in the Sydney region which are heavily treated with poultry manure, an inorganic source of P, are likely to contain higher levels of inorganic P (including plant available P) than organic P.

3.5.3 Adsorption and desorption of P in soil and sediment

Adsorption is the bonding of ions to particles. Desorption is the release of ions from particles. Phosphorus adsorption and desorption processes are complex and not well understood.

Soils have a finite P adsorption capacity that varies with soil texture, mineral composition and organic matter content. Sandy soils have a lower P sorption capacity than clayey soils, hence in sandy soils, inorganic soluble P is likely to remain in this form longer than in clayey soils. This explains why sandy soils in the parts of the Hawkesbury-Nepean catchment are responsive to fertilisers (applied P remains bioavailable for longer). On the heavier soils with a higher P sorption capacity, plant available P is more quickly adsorbed and therefore unavailable to plant uptake. However, as P levels are built up in the soil, added soluble P will remain bioavailable for longer because the P sorption capacity will be reduced (Sharpley, Robinson & Smith, 1995).

P desorption occurs in reducing conditions and/or acidic conditions, for example if the soil is inundated for extended time periods. Desorption is an important process in substrate sediment, thereby releasing P for biological uptake. Mann (1996) suggested that P was desorbed in deeper sections of constructed wetlands, where conditions were anoxic.
Even if point and non-point/diffuse sources of soluble P are reduced in the short term, if there is a significant store of particulate P in the waterway, it is possible that desorption of P in substrate sediment would provide a source of soluble, bioavailable P for some time. The extent to which this would occur is probably dependent on local conditions, particularly the prevalence of anoxic conditions and pH in the substratum.

### 3.6 Transport of Nitrogen and Phosphorus in Runoff

In agricultural catchments, P is contributed to waterways principally by processes of erosion of particulate P, and runoff and drainage of soluble P. N behaves quite differently to P in that it is more dynamic in form and not strongly associated with soil particles. Pionke et al. (1996) measured N and P exports in a 7.4 km² rural catchment where 70% of P export occurred in 10% of the time, in storm flow. In contrast 60% of NO₃ export occurred in non-storm periods, and concentrations were highest in elevated baseflows. Unlike P which was transported as a result of soil erosion, much of the N was transported in soluble form, independent of storm runoff. Ammonium (NH₄⁺) is the exception because it is positively charged, it can be adsorbed onto negatively charged colloids. Hence, ammonium can be transported on clay particles as a result of erosion.

#### 3.6.1 The effect of fertiliser application and tillage

Beaulac and Reckhow (1982) reviewed a number of nutrient export studies and identified factors of fertiliser usage that affect nutrient export. According to Birch (1982) the form and amount of fertiliser applied has some bearing on nutrient exports, but application close to runoff events results in greater fluxes. They also identified incorporation of fertiliser into the soil as a means of reducing nutrient export. The main aspects of manure application that influence N and P losses in runoff include the rate, method, and timing (Sharpley, 1997).

As stated in the previous section, soils have a finite P sorption capacity. Sims and Wolf (1994) refer to the ‘critical degree of P saturation (%)’ above which leaching of soluble P becomes a concern. Although P leaching is generally thought to be negligible, on market gardens with a
long history of high rates of P application and on market gardens on the sandy soils, P leaching and runoff of soluble P is an issue that requires investigation.

Application method and tillage

Reduced tillage is advocated as a method for minimising soil loss. The one positive aspect of conventional tillage is that it allows fertiliser to be thoroughly mixed with the soil. This has 2 effects. Firstly, it reduces the amount of fertiliser directly exposed to runoff. Secondly, it provides maximum contact with soil particles and plant roots, thereby maximising adsorption and plant uptake. Of course the major problem with conventional tillage is that it leads to high rates of soil erosion. In reduced tillage systems, the problem is that fertiliser is not as well mixed. Barisas et al. (1978) compared losses of N and P in runoff, from different tillage systems used for corn in the US, using a rainfall simulator. For each treatment fertiliser was applied prior to tillage. The results indicated that on reduced tillage there was less erosion of sediment bound N and P, but loss of soluble N and P was greater (Figure 3-7). The loss of total P was reduced by reducing tillage because soluble P was a small portion of total P loss.

![Graph](image)

**Figure 3-7. General trend of the effect of tillage on losses of N and P** (Source: Adapted from Barisas *et al*., 1978)

McIsaac *et al* (1995) observed the same trend in soluble P, but found that sub-surface application of the fertiliser reduced soluble P concentrations in runoff considerably. In a 4 year study Sharpley *et al.* (1995) sampled runoff from various tillage/crop treatments (sorghum, wheat, and peanuts). They measured 'bioavailable' P by extracting P from iron oxide
Impregnated paper strips that were mixed with runoff samples. They found that the loss of soluble P increased in the order of reduced tillage (98 g ha⁻¹ yr⁻¹) < native grass (160 g ha⁻¹ yr⁻¹) < no tillage (382 g ha⁻¹ yr⁻¹) < conventional tillage (678 g ha⁻¹ yr⁻¹). They also reported that total P loss was 93% lower from no till than conventional till of which the bioavailable fraction was 73% and 28% respectively, which is consistent with Figure 3-7.

In a study of 7 field sized catchments (2 to 5 ha in size) Jones et al. (1995) compared nutrient and pesticide losses from stubble incorporation and no tillage, without fertiliser application. While no tillage resulted in 54% less soil loss, both tillage systems produced relatively small losses of soluble and particulate P (<3 kg ha⁻¹ yr⁻¹ total N and <1 kg ha⁻¹ yr⁻¹ total P) compared to fertilised catchments.

Timing of application

Nitrogen tends not to accumulate in the soil. N which is not taken up by plants is likely to escape as gaseous NH₃, or leach into groundwater or be transported in runoff as NO₃ (Baird, 1996). Poultry manure is high in NH₄-N and uric acid. The result is that a large proportion of N in poultry manure can be converted to NO₃ - N in a matter of weeks. If this does not coincide with plant growth, leaching or surface water transport may result (Sims & Wolf, 1994). In a rainfall simulation study, Moe, Mannering and Johnson (1967) found that up to 15% of applied ammonium nitrate N was removed by runoff. P however, tends to be sorbed to soil particles. Hence P transport occurs predominantly via erosion, particularly when fertiliser is incorporated into the soil (Barisas et al., 1978; Beaulac & Reckhow, 1982).

Sharpley (1997) found that NH₄-N concentrations in runoff decreased from 2.34 to 7.54 mg L⁻¹ to 0.11 to 5.53 mg L⁻¹ when simulated rain was applied at 1 and 35 days respectively after application. This decrease over time was attributed to volatilisation. Similarly, bioavailable P decreased from 0.99 to 0.65 mg L⁻¹, which was attributed to cumulative P sorption and biological uptake over the 35 days. Robinson and Sharpley (1995) measured losses of N and P that occurred in 5 simulated rainfall storms after a single application of poultry litter. They reported that of total N and P lost over the 5 events, 60% of N and 40% of P was lost in the first event. Edwards et al. (1994) applied simulated rainfall
at 50-100 mm h\(^{-1}\) onto various rates of poultry litter applied to fescue pasture, after 1, 4, 7, and 14 days. They found that particulate matter eroded from applied poultry litter was not affected by the interval between application and first rainfall, but was by the number of rainfall events. The loss of particulates returned to background levels by the third rainfall.

### 3.6.2 The effect of scale and drainage density

Comparing surface water runoff studies at different scales is problematic (Hudson, 1993; Sumner et al, 1996). Different erosion and depositional processes prevail from plot, to farm, to catchment scale. Generally speaking, as scale increases sediment storage increases, or in other words, the sediment delivery ratio decreases with increasing catchment size (Bolton & Ward, 1993; Finlayson & Silburn, 1996). Prairie and Kalff (1987) reviewed studies of nutrient exports from various land uses and found that the scale effect on P export was most pronounced on row cropped land. Marston (1994) suggested that this is because of the dominance of sediment bound nutrients from row cropped land. Bolton and Ward (1993) found that sediment and total P concentrations were highest in overland flow and progressively decreased from agricultural drains to ephemeral streams to rivers.

N is more readily transported in soluble forms in groundwater and runoff, and assimilation is due more to biological and chemical processes rather than physical processes. Therefore the effect of scale on N is less pronounced.

The importance of drainage density in delivery of nutrients (and sediment) to waterways has been alluded to (Sonzogni et al., 1980; Prairie & Kalff, 1986; Marston, 1994). It is generally thought that in catchments with high drainage densities, nutrients and sediments have shorter distances to travel to enter waterways (Figure 3-2), thus there is less opportunity for assimilation within the landscape.
3.6.3 Temporal variation of sediment and nutrient concentrations in runoff

Sediment and nutrient concentrations vary over time, within and between events. Sharpin (1992) described 7 relationships between pollutant concentrations and discharge but also reported a high degree of apparently random variation.

The relationships are described as:

I: leading (first flush)
II: coincident
III: lagging
IV: decay
V: growth
VI: flat
VII: decay/growth

These are illustrated in Figure 3-8.

![Image of Figure 3-8](source: Sharpin, 1992)

Concentrations vary if the source is limited, so that supply of the pollutant is depleted part way through an event. The supply of sediment to can be lower on the receding limb of a hydrograph than on the rising limb. This is partly because more easily removed particles can be quickly stripped, as found by Olive and Walker (1982), and Cooper and Riley (1996).
3.7 Land Management Practices that Reduce Offsite Impacts of Soil and Nutrient Loss

3.7.1 Contour banks

Contour or graded banks are used to divide and therefore shorten the slope length and are built to provide a non-erosive, depositional channel. To some extent they are a means of trapping eroded sediment. Freebairn and Wockner (1986) reported that 80 to 90% of eroded sediment from rill and interrill erosion was deposited in single and double spaced contour bank channels. Sediment concentration in runoff water was reduced from 6.9% above the banks, to less than 0.9% below. However, they also reported widespread failure of contour banks in severe storms, which resulted in flow concentration and gully formation. Failure was due to sediment deposition in the channel which reduced its drainage capacity. The effectiveness of contour banks in severe storm events is enhanced if adequate surface cover is maintained in the catchment (Freebairn & Wockner, 1986).

Contour banks per se are not a common feature of market gardens in the Hawkesbury-Nepean, possibly due to limitations they impose on tractor work. However, drains are sometimes used to dissect a slope, thereby performing similar function as contour or graded banks.

3.7.2 Farm dams

Another aspect of farm management which should be mentioned here is the role of farm dams as sediment retention basins. They can be incorporated into farm design to facilitate depositional processes as well as storing water for irrigation. According to Fennessey and Jarrett (1997) basin geometry can influence their effectiveness at trapping and retaining sediment. The most important features include:

1) length : width ratio should be greater than 2:1
2) maximum surface area.
3) design to avoid 'short-circuiting' so that particle travel distance is maximised.
Fennessey and Jarrett (1997) compared sediment retention in 2 basins, 0.46 m and 0.15 m depth, for 12 storm events. They found that the basins retained as much as 97 % of eroded soil from a single storm event. The amount of sediment re-suspended was greater in the shallower pool.

Most water storages on market gardens are not designed specifically for trapping sediment though. On many market gardens space is at a premium and the primary purpose is generally to store as much stormwater runoff as possible. Hence, the sediment trapping performance of many farm dams on market gardens is probably less than results reported for purpose-built structures. On some farms, the farm dam may not be situated to receive all or any runoff from the vegetable growing area, as on the case study farm.

3.7.3 Constructed Wetlands

Constructed wetlands have been used in the removal of soluble contaminants from sewage treatment plants, industrial effluent, and urban and agricultural runoff (Reddy et al., 1982; Mann, 1996). They should be considered as an additional tool for removing sediment from agricultural runoff.

Nutrient removal by artificial wetlands occurs by biological uptake (Reddy et al., 1982) and by adsorption of nutrients onto soil particles. Various high nutrient sorption quality, industrial by-products have been used as substrate material (Mann, 1996). Redding, Todd and Midlen (1997) found that aquatic macrophytes significantly reduced concentrations of NH₄, NO₃ and P. They attributed uptake to physical, biological and chemical mechanisms. They found evidence of microbial assimilation and sedimentation particularly in submersed macrophytes. According to Richardson (1985) the initially high P removal rates found in newly constructed wetlands, is eventually followed by large P exports. That is, they eventually become saturated. This has been found for both adsorption and biological uptake (Mann, 1996; Redding et al., 1997). Redding et al. (1997) recommend periodic cropping to maintain biological uptake over the long term. Mann (1996) found that between 60 and 80 % of P removed by newly constructed wetlands, was due to P sorption to substratum sediment. The capacity of soil and
sediment to adsorb P is limited, and under certain conditions desorption occurs, releasing P into solution, thereby becoming bioavailable.

Mann (1996) used various industrial by-products high in P-sorbing elements, in substrata of constructed wetlands and found that P sorption was correlated to Ca ($r = 0.96$), and Mg ($r = 0.93$). Sorptivity of Ca was determined by pH. Mann (1996) found that Ca was effective in high pH conditions. Organic matter does not have a major capacity to sorb P directly, but it is thought to form metal-organic matter complexes, eg. with Al, Ca, and Fe. Organic matter results in the formation of humic acids which are believed to compete with P ions for adsorption sites (Sims & Wolf, 1994).

While it is unlikely to expect that market gardeners have the time, money and expertise to construct and maintain purpose-built constructed wetlands, it is not uncommon to see farm dams and drains inhabited by various macrophytes. These opportunistic plants are likely to enhance deposition by reducing turbulence and flow velocity and biological uptake of nutrients, and should be considered as a useful management tool.

### 3.7.4 Grassed filter strips

Grassed or vegetative filter strips have potential to induce deposition and remove nutrients in surface water runoff from cultivated areas (Maggette et al., 1989; Chaubey et al., 1995; Hairsine & Prosser, 1997).

Caubhey et al. (1995) reported good reductions in total (Kjeldahl) N, NH$_3$-N, total P and PO$_4$-P from fescue strips that were 21.4 m long and treated with poultry manure on the upper 3.1 m. Maggette et al. (1989) found that fescue filters 4.6 m and 9.2 m long, were far more effective at removing sediment than N and P from runoff from bare plots 22 x 5.5m treated with surface-applied urea-ammonium-nitrate and broiler manure. They found that nutrient removal was highly variable, generally decreasing with more runoff events, which was also the case for sediment removal. They concluded that to be effective, grassed filter strips need to be of sufficient size to maintain an adequate vegetated : unvegetated area ratio.
The spatial requirement for effective grassed filter strips is the major limitation for their use on market gardens. Water quality in runoff might be improved significantly if adequate space is allocated.

3.8 SUMMARY

Methods for measuring nutrient and sediment losses in stormwater are expensive and labour intensive. Current CMSS estimates of N and P exports from market gardens are higher than for any other land use. On market gardens, soil erosion is likely to be an important mechanism for nutrient transport and is likely to be most severe in high intensity storms when soil cover is minimal and soil disturbance is maximal. Although an organic fertiliser, poultry manure can result in high concentrations of N and P in runoff. Careful consideration should be given to farm design to maximise sediment and nutrient attenuation on farm. Slope lengths should be minimised by channels. Significant improvements in water quality can certainly be achieved if it is directed through appropriately designed dams, constructed wetlands and grassed filter strips. To some extent these measures may counteract poor agronomic practices. However, to effectively do this, a large proportion of the farm must be allocated to non-productive uses. More appropriately, these structural measures should complement good agronomic practices.
4. **SYNTHESIS OF RESEARCH QUESTIONS**

4.1 **SEDIMENT AND NUTRIENT LOSS - IMPLICATIONS FOR WATERWAYS**

There is a lack of data on the offsite impacts of market gardening in Australia and in the Hawkesbury-Nepean catchment. Presumably, market gardening is detrimental to waterways because CMSS estimates which are based on interstate and overseas data, indicate that losses of N and P may be as much as 35 and 15 kg ha\(^{-1}\) yr\(^{-1}\) respectively, well above other land uses (Marston, 1994). There is very little data on soil losses from market gardening, but soil erosion from comparable land uses is highly variable. Influencing factors include soil type, topography, rainfall, horticultural practices and erosion control measures.

*Research question:* What were the sediment and nutrient losses from the case study farm? Are CMSS estimates of N and P export reliable for the case study farm? What are the implications for receiving waterways?

4.2 **LINKS BETWEEN MANAGEMENT AND STORMWATER LOSSES - ECONOMIC IMPLICATIONS**

Theoretically the loss of sediment, N and P in stormwater is a cost to the farmer and decrease profitability. High rates of soil loss would reduce profitability in the long term and purchase of fertiliser is an ongoing production cost. Maximum crop uptake of fertiliser nutrients should be an economic imperative for the farmer.

4.2.1 **Linking farm management to impacts - the scale issue**

Making links between land management and runoff, soil and nutrient loss depends heavily on scale and complexity of land management practices. Soil erosion and nutrient loss can be monitored at various scales from plot, field, farm, and catchment. Each approach has its merits. Monitoring real world situations of farms and catchments may give an accurate
indication of total loads, but adequately describing land management practices and relating them to runoff becomes increasingly difficult.

Much data on other agricultural land uses has been collected in plot studies (Barisas et al., 1978; Bradford & Huang, 1994; Edwards et al., 1994; Gaynor & Findlay, 1995) and to a lesser extent field studies (Costin, 1980; Freebairn & Wockner, 1986) where land management is homogenous. Land management on market gardens is heterogenous, therefore blurring links between specific land management practices and runoff. The advantage of the farm-scale approach is that the impact of the whole farming system including on and off-field practices can be determined.

In general, rates of sediment and nutrient export are high when land uses include intensive cultivation, minimal soil cover and over use of fertiliser (Beaulac & Reckhow, 1982). P losses from such land uses are characterised by a predominance of particulate P rather than soluble P. There is less certainty about the forms of N in runoff, because it more readily changes between solid, liquid and gaseous forms, and is more prone to leaching. This is discussed in more detail in Chapter 3.

*Research question:* What value were the farm and plot-scale studies?

### 4.2.2 Fertiliser use

Market gardens currently apply high rates of poultry manure as well as inorganic fertilisers (Jinadasa et al., 1997). Although organic fertiliser is perceived as an ‘environmentally friendly’, slow release fertiliser, the literature suggests that high levels of inorganic P can be expected in soils (Sharpley & Smith, 1995) and runoff (Sharpley et al., 1995) when applied at high rates. There is also potential for losses of N from poultry manure if it is not stored appropriately and incorporated immediately into the soil when it is applied (Sims & Wolf, 1994).

The timing and method of fertiliser application have been shown to influence nutrient losses (Barisas et al., 1978; Robinson & Sharpley, 1995). Runoff events that occur a short time after
fertiliser application are likely to have the highest nutrient concentrations. If fertiliser is incorporated into the soil, losses of soluble nutrient forms can be reduced. However, unless fertigation is used, this necessitates tillage, which then increases the erodibility of the soil.

Ideally, addition of fertiliser P should equal crop uptake plus runoff losses. Crop uptake of fertiliser N of 50% should be achievable (Baird, 1996), in which case stormwater loss of N should be well below 50% (allowing for other losses, eg. gaseous and leaching). In South Australia, Holmes (1979) reported that N lost in runoff was equivalent to 19% of fertiliser inputs.

Research question: How efficient were fertiliser practices on the case study farm? What savings could be made by reducing fertiliser inputs without decreasing yield?

4.2.3 Cultivation and soil cover

Soil erosion is an important mechanism for the transport of pollutants. Cultivation and crop cover are management variables that strongly influence soil loss (Hamblin, 1987; Moss, 1989, Freebairn & Wockner, 1986).

Research question: What management factors most strongly influenced sediment loss? What are the implications for management?

4.3 Extension and Adoption of Improved Land Management

Land management in the market gardening industry is influenced by a range of economic, social, and physical factors. Market gardeners are faced by many issues such as low profit margins, pests, poor literacy and conflict with neighbouring land users (Murison, 1995; Senn, 1996; Kelleher et al., 1997). There is a general perception that market gardeners are not well aware of land and water degradation.

Education of farmers is the key to achieving widespread improvement in land management as demonstrated by Senn (1996). Research should be shaped by the needs of farmers.
Conducting more research on farms should lead to greater understanding and stronger communication links between scientists, farmers and extension officers.

Research questions: This research provided the opportunity to maintain regular contact with a market gardener for 2 years. What insights can be gained from this experience regarding extension and adoption of sustainable land management?
5. Materials and Methods

For the Case Study

5.1 The Case Study Farm

5.1.1 Location and size

Located near Freemans Reach, between North Richmond and Windsor, in the Hawkesbury-Nepean catchment, the case study farm is situated on the bank of Currency Creek several km above its confluence with the Hawkesbury River (Figure 5-1). The farm is 8.8 ha in total and has a cultivation area of 6.6 ha (Figure 5-2 and Plate 1). Two or 3 beef cattle are run on the remainder of the farm.

The farm is owned and operated by 2 Maltese vegetable growers who are in their late 30’s. They wish to remain anonymous so I will refer to them as Jim and Sue Saliba (pseudonyms) or simply the case study farmer(s). Mr and Mrs Saliba have been on the farm for 12 years. They have 4 children, aged between 6 and 14. Mr and Mrs Saliba do not employ any outside help. The children help out after school and in school holidays. Mr and Mrs Saliba work long hours and are hard working. They are also very passionate about their work. Mr Saliba goes to Flemmington Markets 3 days a week to sell their produce.

Generally, the farm is not dissimilar to the ‘typical’ Maltese market garden described by Murison (1995). The Salibas keep their farm neat and tidy, a common characteristic of many Maltese market gardens.
Figure 5-1. The Currency Creek subcatchment, showing the case study farm (stn 6) and Currency Creek
Plate 1. View of the case study farm, looking upslope at the drainage channel in dry and wet conditions.
Figure 5-2. Schematic of layout of the case study farm
5.1.2 Choice of farm for the case study

Two market gardens were involved in the NLP project in the Currency Creek subcatchment (Figure 5-1). These are owned by the Salibas (station 6) and the Borgs (station 2) (Borg is also a pseudonym).

A note about pseudonyms used

Both farms mentioned here were involved in an action learning project involving permanent beds (Senn, 1996). Senn describes the individual farmers using pseudonyms. To allow cross referencing between Senn's work and this work the same pseudonyms have been used.

After careful consideration of the research topic, it was decided that the best approach was to use Saliba's market garden as the case study farm. The Saliba farm was monitored for the longest period and its management was observed closely. A brief comparison of the sites follows, supporting the use of station 6 for the case study.

The Saliba farm was the first market garden monitored in the NLP project. Installed in May, 1995, station 6 was the first of 5 stations installed on market gardens in the NLP project. Monitoring of the other market gardens commenced about 6 months later.

The second advantage of station 6 was related to assessing land management practices. The catchment of station 6 included more than 95% of the cultivation area on Saliba's farm. This meant that fertiliser inputs over the whole farm were virtually the same as for the catchment area. Therefore, the affect of land management practices such as fertiliser use, could be related to runoff losses from the catchment. Of the other market gardens, a smaller proportion of their crop growing was monitored. Station 2b included 9.1 of 15.6 ha. This created uncertainties in assessing the importance of fertilisation and other land management practices. For example, a known quantity of fertiliser may have been applied to the whole Borg farm, but the percentage applied within the catchment areas of stations 2a and 2b could not be determined with confidence.

Chapter 5: Materials and Methods for the Case Study
The Borg cropping area (station 2) was larger, more complex, and due to the broader range of crops grown, tillage was less seasonally dependent. The Saliba cultivation area however, was smaller and simpler in paddock layout. Cropping was more seasonal; the Saliba’s did not grow any lettuce over summer and as lettuce was their main crop, most of the cropping area was left in fallow over summer. The extent of land in summer fallow on the case study farm is probably greater than most market gardens, but was regarded as a positive. It was presumed that this would lead to seasonal variations in rates of soil and nutrient loss, therefore providing a stronger link between land management and nutrient and soil losses in runoff.

The third reason for choosing Saliba’s farm for the case study related to management of the market gardens. Two people only worked on Saliba’s farm (Mr and Mrs Saliba). Mr Saliba was making most of the management decisions and was also heavily involved in the day-to-day work. This made obtaining land management data for Saliba’s a relatively straightforward (but not necessarily easy) task. However, 4 people worked on Mr Borg’s market garden, with less defined roles, which made finding out management details a more complicated process. Mr Borg was the owner and manager, but his son was carrying out much of the on-farm work and it seemed that Mr Borg’s son was working under very broad instructions. Mr Borg’s son definitely seemed less interested in soil management than both Mr Borg and Mr Saliba. Mr Saliba was particularly interested in soil management and demonstrated this by carrying out independent trials (one of which is the subject of Chapter 8).

Fourthly, station 6 had a more clearly defined catchment boundary compared to stations 2 and 8/9. In comparison to the other 2 market gardens, Saliba’s farm received runoff from only a small area of adjoining grassed land (Figure 5-1). Station 2 received runoff from approximately 70 ha, which included irrigated dairy pasture as well as the 20 ha market garden at station 2 (Mr Borg’s farm). The Mangrove Mountain market garden (station 8/9) received runoff from approximately 90 ha, which included the market garden plus poultry, another market garden and intensive horticulture (orchards). At both stations 2 and 8/9, most of the runoff water entering the farm was monitored, nevertheless it was difficult to separate the total nutrient/sediment load from upstream land uses with confidence. The results from station 6 can be attributed to solely market gardening land management practices.
It was envisaged that station 6 (Saliba's farm) provided the best opportunity to draw conclusions about relationships between land management and soil and nutrient losses. Consequently, station 6 on Saliba’s farm was the only market garden for which a relatively large amount of runoff and land management data had been gathered.

5.1.3 Soil

The soil on the case study farm is a Brown Podsolic (Db2.41, Db1.41), hardsetting and prone to surface sealing (Cornish et al., 1997). The soil on the vegetable growing beds is moderately to highly erodible, due to high silt and fine sand content (74%), poor structure, and low organic matter. The A₁ horizon is moderately dispersive, while the A₂ horizon is moderately to highly dispersive (Cornish et al., 1997) a factor which contributes to surface sealing (Abusharar, 1993).

5.1.4 Topography

Assessment of topography of the catchment was determined using a 1:4000 contour map. Slope of the beds and drainage channel was measured with a clinometer.

The slope of the furrows ranged from level to gently inclined, with an average gradient of 3%. The furrows sloped into a level to very gently inclined drainage channel (average gradient of about 1%) with a slightly concave profile. Due its low gradient, the drain acts as a sediment trap, which the farmer excavates every 2 or 3 years, and then spreads the excavated sediment back onto the paddock (Plate 1).

This occurred at the end of 1995, after which the channel quickly filled with sediment eroded from the beds. By the end of 1996 there was evidence that erosion and depositional processes in the drain were reaching equilibrium. Alluvial fans and rills were observed in the drainage channel. Some rilling was observed in the furrows, particularly where they sloped into the drainage channel, ie. at the bottom end of the furrows.
5.1.5 Crops
During the study lettuce was the major crop grown, along with spinach, capsicum, cabbage and cucumber. As lettuce is susceptible to disease in summer, much of the farm was left in fallow over summer. Each summer Mr Saliba planted ‘green manure’ crops on small areas. In the 95/96 summer about 0.4 ha of oats were planted at the eastern extremity of the cultivation area. In the 96/97 summer, 2 plots (total area of ≈1 ha) were planted with sorghum as the green manure.

5.1.6 Irrigation
Water was supplied to crops via overhead sprinklers. Water was pumped from a 31 ML dam located on the south-east corner of the property, which was fed by an ephemeral stream during larger rainfall events. If the dam spilled, outflowing water rejoined the original stream on the adjacent property and did not mix with the monitored runoff. Failing sufficient rain, water was pumped into the dam from the Hawkesbury River.

The Salibas varied the frequency of irrigation according to their judgement of preceding and forecast weather conditions, crop type and age, and soil moisture. During warm, dry conditions crops were irrigated once or twice daily. In still conditions in cooler months when there was heavy dew, the frequency was reduced to once every several days. Runoff from irrigation often results in small flows entering Currency Creek (via the drainage channel).

5.1.7 Cultivation practices
Although the Salibas relied heavily on machinery (rotary hoes, bed formers etc.), to prepare the vegetable beds, they liked to work in bare feet, to “get a feel for the soil”. Mr Saliba was very particular about his tillage practices.

Like other market gardeners on the poorly drained loams in the Hawkesbury-Nepean catchment, the Salibas grew their vegetable crops on raised beds (Plate 2). As previously mentioned, the Salibas participated in the action learning project involving trialling ‘permanent beds’ (Senn, 1996). According to Senn (1996) permanent or semi-permanent beds have
become the industry norm in the Sydney region since the inception of this research. The Saliba's cultivation practices are described by Senn as semi-permanent beds, where the furrows remain largely intact in the process of cultivation, and the beds are not deeply rotary hoed after every crop. The furrows measured approximately 1.5 m apart. The height from the top of the beds to the bottom of the furrow varied between 0.15 to 0.3 m. Senn (1996) outlines the Saliba's cultivation practices before and during the permanent bed project.

Senn (1996) noted that during the permanent bed project, Mr Saliba reduced his cultivation markedly and consequently, his cultivation time dropped from 28 hr ha$^{-1}$ to 8 hr ha$^{-1}$. After the project finished his cultivation increased, but not to anywhere near its previous level. At the time of the permanent bed project, Mr Saliba's cultivation techniques included:

1. 100 cm rotary hoe @ 5-10 cm
   - x 1 pass, or herbicide without any soil disturbance

2. ripper/hiller @ 5-10 cm below level of gutters
   - x 1

3. 100 cm rotary hoe @ 5-10 cm
   - x 1 pass

After the permanent bed trial Mr Saliba started using the 200 cm rotary hoe after every third or fourth crop (presumably several passes) though still maintaining furrows as tractor laneways.

In late 1995, Mr Saliba described his cultivation techniques to the author, as a 3-crop rotation:

*After crop 1:*

1) 200 cm rotary hoe @ 10-15 cm
   - x 3 passes(at least)

2) ripper (depending on weather and soil moisture)
   - x 1 pass

3) hiller
   - x 1 pass

4) 100 cm rotary hoe @ 7-10 cm
   - x 1 pass

*After crop 2:*

1) 100 cm rotary hoe @ 7-10 cm
   - x 1 pass

*After crop 3:*

1) bedlifter @ 30-38 cm
   - x 1 pass

2) 100 cm rotary hoe @ 7-10 cm
   - x 1 pass
Sometimes cultivation for crop 2 was repeated after crop 3 if time was limited. Poultry manure was applied after crop 1, between steps 3 and 4. Herbicides were often used in conjunction with tillage to control weeds, being careful to avoid disturbing the soil after the herbicide was applied, so that germination of more weeds was minimised.

On some visits, it was observed that some of the beds and furrows were rotary hoed to a fine tilth (Plate 3) while at the same time, others were rotary hoed to less than 10 cm depth and to a coarser tilth (Plate 4). Obviously, there was some perceived benefit to periodically resorting to traditional tillage practices.

Weeds such as fireweed and milkweed were sometimes prevalent on the vegetable beds, generally after a crop had been harvested and prior to preparation of the beds for the following crop. Weeds were controlled by a combination of mechanical and chemical means. Mr Saliba often sprayed weeds that germinated after cultivating the beds in preparation for the following crop.

When beds were prepared, the furrows were loose and crumbly. Heavy rain in this state resulted in movement of soil from the furrow walls and floor to the floor of the furrows downslope or the drainage channel which drained most of the cultivation area. The furrows were compacted with each pass of the tractor after the beds were prepared and became hard-set after wetting-drying cycles.

In the case of the cover/green manure sorghum crop grown in the 1996/97 summer, the beds were covered quite densely by weeds which were sprayed after a very shallow ($\approx 5$ cm) pass of the 100 cm rotary hoe. The sorghum was sown in the cover of dead weeds (Plate 6).
Plate 2. Raised beds with recently planted lettuce seedlings and sprinkler irrigation system (Note the neat, weed-free appearance that is typical of the case study farm)

Plate 3. Finely tilled beds with maximum relief
Plate 4. Coarsely tilled beds with lower relief (Note large clods and weed residue)

Plate 5. Lightly tilled beds with partially intact cover of dead weeds, into which a sorghum green manure crop was sown
5.1.8 Fertiliser types and application

Poultry manure was the major form of fertiliser used on the farm and was supplemented with ammonium nitrate. Lime was applied in January, 1996 at 1.5 t ha\(^{-1}\). Generally, the poultry manure was applied between late summer and early autumn. Ammonium nitrate was applied throughout the year as base and side dressings. All fertiliser was incorporated into the soil.

Fertiliser inputs were quantified in consultation with Mr Saliba and are given in Chapter 6.

5.2 SOIL MEASUREMENTS

Soil samples were collected on 2 occasions using an auger. In March, 1995, samples were collected from the A\(_1\), A\(_2\), and B horizons. On the case study farm, samples were taken at depths: 0-20 cm, 20-40 cm, and 40-60 cm. Samples were also taken on Borg’s farm at 0-5 cm, 5-30 cm, and 30-60 cm. In January, 1997, samples were collected only from the A\(_1\) horizon on Saliba’s farm. All soil samples were analysed by staff at the Department of Land and Water Conservation (DLWC) Scone Research Service Centre, NSW.

For the A\(_1\) horizon samples, more than 6 sub-samples were collected from within a 10 m radius, and thoroughly mixed. Subsurface samples were collected and analysed singularly. Samples collected on Saliba’s farm were taken from the cultivation area, adjacent to station 6. In 1995, samples were also taken from the adjacent unimproved pasture to provide an indication of soil qualities prior to market gardening.

Analyses included:

- particle size analysis (PSA)(DLWC method P7B/1: determination by seiving and hydrometer) - clay (<2\(\mu\)m), silt (2-20\(\mu\)m) fine sand (20-200\(\mu\)m), coarse sand (200-2000\(\mu\)m) and gravel (>2mm);
- dispersion % (DLWC method P8A/2: ratio of particles <0.005 mm that remain in suspension in distilled water after 2 hours settling time compared to total amount of materials <0.005 mm, determination by hydrometer);
• Emmerson aggregate test (EAT) (DLWC method P9B/2: an 8 class classification of soil aggregate coherence, related to Australian Standard AS1289);
• pH (DLWC method CA2/2: 1:5 soil/water);
• electrical conductivity (EC) (DLWC method C1A/3: 1:5 soil:water suspension);
• organic carbon (OC) % (DLWC method C6A/2: Walkley Black method, organic matter is oxidised in dichromate and the amount of dichromate reduced is determined by titration with ferrous sulphate using Ferroin indicator.);
• total N % (DLWC method C7A/2: macro-Kjeldahl total N, conversion of nitrogen to ammonium by digestion with sulphuric acid and subsequent determination of NH$_3$ liberated from digest by steam distillation with NaOH. NH$_3$ is determined by titration with sulphuric acid.);
• plant available P (DLWC method C8A/2: Bray method, extraction of absorbed P with HCl and NH$_4$F. Concentration is determined by spectrophotometer);
• P sorption (DLWC method C8B/1: a standard P solution is added to a soil. After equilibration, the P remaining in solution is measured colourimetrically and the P adsorbed by the soil determined by the difference); and
• CEC, Na, K, Ca, Mg, and Al cation exchange (DLWC method C10A/2: atomic absorption spectroscopy on a 1:5 soil:water extract using the <2 mm fraction shaken for 1 hour).

Drainage channel sediment

In January, 1997, 3 core samples were taken from the channel which drained the vegetable beds, adjacent to the monitoring station and at 100 m intervals upslope. This was undertaken to assess the storage of plant available P in the channel and to assess the particle size distribution of the deposited sediment. At the time of sampling, there was no surface water in the channel.

Core samples were collected in 100 mm diameter PVC tubing, hit into the sediment, capped, removed and then sealed on the bottom end. The sealed tubes were sent to the DLWC Scone Research Service Centre (within 2 weeks) where they were cut lengthwise to extract the sample. The deposited sediment was easily distinguished from the underlying soil, being much
darker in colour. Analysis was only performed on the deposited sediment. Analyses performed were:

- particle size analysis (PSA)(DLWC method P7B/1: as for soil samples) - clay (<2 μm), silt (2-20 μm) very fine sand (20-100 μm), coarse fine sand (100-200 μm), coarse sand (200-2000 μm), and gravel (>2mm);
- pH (DLWC method CA2/2: 1:5 soil/water);
- electrical conductivity (EC)(DLWC method C1A/3: as for soil samples);
- organic carbon (OC) % (DLWC method C6A/2: as for soil samples);
- total N % (DLWC method C7A/2: as for soil samples);
- P sorption (DLWC method C8B/1: as for soil samples);
- plant available P (DLWC method C8A/2: as for soil samples); and
- total P (ACIRL method MEM-010: mixed acid digestion, run on ICPAES)

5.3 COLLECTION OF LAND MANAGEMENT DATA

At the time of the study, the Saliba's kept no records of their land management practices. Instead, general information on aspects of land management such as fertiliser use, tillage, irrigation and weed management, were collected through informal discussions held on an ad hoc basis, usually on visits to tend to the monitoring equipment.

Every effort was made to avoid unnecessary disruptions to the Saliba's work. Generally if asked, they were willing to recall information such as fertiliser volumes, provided they could do so while continuing their work (and provided it was recent).

5.3.1 Quantification of fertiliser inputs

Two methods were used for calculating fertiliser inputs:

Mr Saliba's estimates of fertiliser rates (e.g. m³ ha⁻¹) were based on the number of beds or sprinkler lines (generally, 9 beds = 1 line) covered by one load of the manure spreader. For example, poultry manure rates were calculated by estimating the number of beds covered by one load of the manure spreader.
For example:

1 load of manure spreader covered 7 beds [width] by 15 pipes [length]
= 4m$^3$ covered 0.14 ha (10.5m x 135m)

*Application rate is 28 m$^3$ ha$^{-1}$.*

The problem with the estimates of application rate was that: 1) the beds were not all the same length because the cultivation area was not rectangular, so one of 3 variables could have easily been incorrect; and 2) the actual number of beds covered by one load might not have been a whole number (e.g. the number might have been 6.6 not 7) as stated by Mr Saliba.

The alternative was to determine total inputs, ie. purchases. For example:

4 truckloads of poultry manure (@32m$^3$) covered 6.6 ha
128m$^3$ (4 x 32m$^3$) covered 6.6 ha

*Application rate is 19.4 m$^3$ ha$^{-1}$.*

Poultry manure has a bulk density of approximately 500 kg m$^{-3}$ (Embry & Allan, 1984). A study of N and P content in poultry manure in Western Sydney gave average values of 3.2 % and 2.15 % respectively (Embry & Allan, 1984). These values are consistent with values found in studies in other areas and overseas (Tunney, 1980; Derrick, 1996) though some variation occurs due to differences in diet and addition of other materials (e.g. straw instead of wood shavings). These values were used to calculate the N and P inputs from volumetric estimates of poultry manure inputs.

5.3.2 Assessment of soil cover and soil condition

In January, 1996, a methodology was developed to survey land management conditions that influence soil and nutrient movement in runoff. It involved mapping of soil cover and soil disturbance at the time of each runoff event. *For comparison, the Borg farm was surveyed in the same manner.*
The following definitions were used:

**Soil cover %**: the percentage of the site, including furrows, occupied by the vertical projection of foliage and other material (vegetable plant, weeds, plastic mulch, stuble and residue). This was estimated to the nearest 10 %.

**Soil condition (SC)**: a qualitative assessment of the degree of disturbance to the soil surface. Three categories were used, on a scale of 1 - 3, with 1 being low and 3 being high.

- **SC1**: No obvious surface disturbance. Beds/furrows usually lower relief than SC3. Surface seal/crust present.
- **SC2**: Evidence of soil disturbance on part of the bed due to mechanical weed removal, planting or harvesting; or partial settling of soil surface after heavier ploughing.
- **SC3**: Soil surface loose over whole bed due to recent use of rotary hoe, ripper or hiller.

The cultivated area was subdivided into many smaller separately managed units (particularly on the Borg farm) on which there were different crops at different stages of maturity, with different levels of cover and soil disturbance. To get around this problem, a map of each farm was traced from an aerial photograph. Details of the farm layout were traced, including adjacent vegetation and buildings, tillage orientation, dams, and buildings to provide reference points. After each major runoff event, the farms were visited and each crop area inspected. The parameters were assessed visually, and recorded on the farm map.

The areas of 10 cover classes were grouped into 3 larger classes:

- 0 - 20 %
- 21 - 50 %
- > 50 %

A digitiser was used to measure the area on the farm maps of each plot, which was then recorded under the respective class of soil cover and soil disturbance. After doing this for all of the plots which made up the cultivation area, the areas under each category of soil cover and soil condition were summed and calculated as percentages of the total cultivation area.
5.3.3 Irrigation and soil water

Observations of irrigation practices were recorded in the field notebook. To determine the rate of application from irrigation, 20 cylinders were placed randomly on the vegetable beds during a one hour of irrigation.

No measurements of soil water content were made because of time limitations and the heterogenous soil water management on the farm at any point in time. Instead, a crude water balance was used for the farm over a 3 month period when 2 contrasting runoff events occurred, to calculate spatially averaged available soil water ($ASW$). Maximum $ASW$ was assumed to be 70 mm, based on the calculated maximum infiltration from the farm-scale runoff monitoring (Chapter 7). Spatially averaged $ASW$ was calculated from areas of the cultivation area that fell into the following categories:

1) irrigated crops
2) non-irrigated crops and weeds
3) bare fallow

The following assumptions were adapted from Cornish (1983).

Assumptions

Irrigated crops: Comparison of rainfall and pan evaporation ($E_{pan}$) data and observations of irrigation frequency suggested that any area carrying crops was irrigated and therefore would rarely have had a soil water deficit ($SWD$) of >10 mm. This conclusion assumes the soil remained sufficiently wet to allow Stage 1 drying (Cornish, 1983) at a rate equivalent to pan evaporation.

Weeds: Stage 1 drying was assumed to occur for a $SWD$ of up to 20 mm because the B horizon was relatively impermeable and likely to create a perched water table after rainfall. In Stage 2 drying, evapotranspiration ($ET$) was assumed to be 0.5 of $E_{pan}$. Weeds generally had soil cover of 50% or less on the case study farm.

Bare fallow: Stage 1 drying assumed for $SWD$ up to 20 mm. In Stage 2 drying, soil evaporation ($E_a$) was assumed to be 0.2 of $E_{pan}$.

Drainage: No drainage or deep percolation because of the impermeable B horizon.
5.4 Surface Water Runoff

5.4.1 Continuous discharge monitoring and automatic sampling system

The monitoring system performed 2 main functions; firstly to record flow (with emphasis on high flow) in the drainage channel, and secondly, to take water samples during runoff events, on the rising and falling limbs of the hydrograph (discharge (m³ min⁻¹) plotted over time). The sampling strategy was based on discharge increments and was therefore dependent on recording flow. Provided the discharge increments are appropriate for the magnitude of the runoff event, sampling based on discharge provides better representation of the entire hydrograph than do strategies based on time or depth sampling (Cornish et al., 1996).

The monitoring station is illustrated in Figure 5-3. A data logger (Datataker DT50) controlled the system. It received water level depth readings in the channel spillway from a water level (Greenspan) sensor. The sensor was placed inside a vertical 10 cm diameter PVC tube, joined to a horizontal 2.5 cm diameter PVC tube that allowed water to enter when the water level rose above the invert of the weir spillway (Figure 5-3). The data logger also received signals from an electronic rain gauge (Dataflow ‘dripping’ type).

The data logger calculated discharge by inputting the water level depth into the programmed rating curve for the spillway of the drainage channel. The rating curve is the relationship between water depth and discharge. To determine the rating curve, the spillway was identified and surveyed. To improve accuracy of the rating curve a rectangular weir (0.8 m wide x 0.6 m deep) with a straight 3 m channel and a free overfall was installed, early in 1996 by the author. The equation for the rating curve of the weir was

\[ Q = 1.346 \, WL^{1.5} \]

where \( Q \) = discharge (m sec⁻¹) and \( WL \) = height of water above weir invert (m)

The data logger was programmed to log at fixed 3 hour intervals during dry weather. This ‘sleep’ mode preserved battery power. During rainfall, pulses from the rain gauge activated the monitoring system into ‘awake’ mode, with a 2 minute logging interval, thus providing a
virtually continuous record of rainfall intensity. The ‘awake’ mode was also activated by a change in discharge above a preset level.

Calibration of the rainguage was done in the laboratory by adding known depths of water to the guage and using the data logger to count the number of drops passing through the guage. The depth (mm) divided by the number of drops gave the depth per drop, which was entered into the program.

When the discharge rate increased or decreased sufficiently, the data logger triggered a Gamet autosampler, which was used in pulse mode, to take a sample. This occurred when the discharge rate rose above a specified level (the initial trigger) and thereafter, when the discharge rate rose or fell by the specified discharge increment.

The Gamet autosampler contained 24 bottles, so it was important to set an appropriate ‘initial trigger’ and discharge increment. To increase the discharge increment during larger events, an increment multiplier was used to double the increment when discharge rose above a preset discharge, for example, for a one year ARI (average recurrence interval) event (an event with a one year return period).

Figure 5-3. Schematic diagram of the monitoring equipment at station 6
Plate 6. Monitoring station 6, on the boundary of the case study farm

The sampling point was a 25 mm hose located in the channel one metre upstream from the spillway. The hose was fixed beneath a float that slid freely on a steel rod hit firmly into the channel. This ensured that the sampling inlet was located above the channel bottom, thus avoiding sampling the bedload.

The system was powered by a 12 volt battery which was replaced every 4 to 5 weeks. The data logger and autosampler were housed in an elevated shelter constructed of compressed fibro and 1.6 mm galvanised steel sheeting (Plate 6). The data logger was placed in a plastic container to provide additional protection, inside of which 10 to 20 g of desiccant was placed.

The autosampler was not refrigerated so samples had to be collected as soon as possible after being taken. As a result, sample collection was a priority for this work, particularly during prolonged runoff events. Samples were usually collected within 2 days of sampling. Sample handling and analysis is described in 5.4.4 and 5.4.5.

Chapter 5: Materials and Methods for the Case Study
5.4.2 Manual sampling

Occasionally, manual samples were collected from the sampling point for the automatic monitoring system and analysed as for the samples collected from the autosampler.

5.4.3 Calculation of runoff quantity

Table 5-1 shows an extract of raw data from the monitoring system during a rainfall/runoff event. To calculate the quantity of runoff, the discharge (m$^3$ sec$^{-1}$) was multiplied by 60 to give m$^3$ min$^{-1}$.

<table>
<thead>
<tr>
<th>Date</th>
<th>Time (hr:min:sec)</th>
<th>Rain (mm)</th>
<th>Water level (m)</th>
<th>Discharge (m$^3$ sec$^{-1}$)</th>
<th>Get sample</th>
<th>Acknowledge sample</th>
</tr>
</thead>
<tbody>
<tr>
<td>12/02/97</td>
<td>7:10:11</td>
<td>0.39</td>
<td>0.376</td>
<td>0.31074</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>12/02/97</td>
<td>7:12:11</td>
<td>0.26</td>
<td>0.380</td>
<td>0.31511</td>
<td>1*</td>
<td>0</td>
</tr>
<tr>
<td>12/02/97</td>
<td>7:14:11</td>
<td>0.26</td>
<td>0.380</td>
<td>0.31511</td>
<td>0</td>
<td>1</td>
</tr>
</tbody>
</table>

*Sample triggered after a rise in discharge (m$^3$ sec$^{-1}$)

To explain how the quantity of runoff was calculated I will refer to Table 5-1. The quantity of runoff for the 2nd record was calculated by multiplying the average discharge [m$^3$ min$^{-1}$] from the 1st and 2nd records by the number of minutes from the 1st to the 2nd record. The quantity of runoff that passed through the weir between the 1st and 2nd records was:

runoff quantity [m$^3$] = \((0.31074 \times 60) + (0.31511 \times 60)/2\) \times 2

= 37.551 m$^3$

The runoff quantity for the 3rd record was calculated by averaging discharge between the 2nd and 3rd records by the time between the 2nd and 3rd records. The sum of runoff quantity for each record in a runoff event gave the total quantity of runoff (m$^3$) for the event. These calculations were performed using a computer spreadsheet (Excel). To create a hydrograph, the 24 hour time recordings were converted to cumulative time. This was done by using the program function in Excel.
5.4.4 Sample collection and handling

Before deciding on the collection and handling strategy, detailed studies were made by the CSIRO Water Quality Group to determine the most suitable methods for analysis. These quality assurance and quality control measures are described in detail by Cornish et al. (1997). The CSIRO Water Quality Group chose, optimised and validated the procedures for the NLP study.

Ideally, the samples would be collected from the field immediately, but the number and geographical separation of the monitoring stations limited the response time for sample collection. The samples were collected as soon as possible after rainfall, on most occasions within 2 days. The effect of leaving samples in the autosampler at ambient temperature for periods of up to 6 days was tested. The CSIRO Water Quality Group found that ambient storage for up to 2 days does not lead to significant changes in N and P species (Cornish et al., 1997).

Once collected from the field, the samples were frozen and stored at UWS for up to 4 weeks, when they were delivered to the CSIRO laboratory at North Ryde. Upon delivery, the samples were stored at -20°C until processed. CSIRO tested the effect of this procedure and found only NH₄ (which increased) was significantly altered by freezing for up to 4 weeks. Ideally NH₄ would be analysed within a few hours of collection (Cornish et al., 1997) but this simply was not possible and in any case was a small fraction of total N.

All sample containers were washed using a phosphorus free detergent (Decon 90) and rinsed 3-4 times in distilled water, and soaked in distilled water for 1-2 days.
5.4.5 Laboratory Analysis

The laboratory analysis was done at CSIRO, North Ryde, by CSIRO staff. The methods are outlined below.

Glossary of Terms for Chemical Components

**Soluble**

The fraction which passes through a 0.7 μm pore size glass fibre filter.

**Particulate**

The residue left on a 0.7 μm filter.

**SS**

Suspended solids. The concentration of particulates in the sample and is collected on a 0.7 μm filter.

**SRP**

Soluble reactive phosphorus. The phosphorus in a filtered sample that reacts directly with acidic molybdate to form the molybdenum blue complex.

**TSP**

Total soluble phosphorus. The phosphorus measured colorimetrically in a filtered sample that has been digested using alkaline persulphate.

**DOP**

Dissolved organic phosphorus. Equals the difference between TSP and SRP. Although in some circumstances (eg laundry effluent) this could be inorganic, in agricultural runoff this fraction is most likely to be organic P.

**Particulate P**

The phosphorus measured in the particulate residue after digestion using a Kjeldahl digest.

**NO\(_3^-\)**

Oxidised nitrogen. Consists of nitrate plus nitrite, measured colorimetrically in the filtered sample using the sulphanilamide/NED diazotisation reaction.

**NH\(_3\) or NH\(_4^+\)**

Ammonia. Measured colorimetrically from the filtered sample using the Berthelot reaction using salicylate.

**TSN**

Total soluble nitrogen. Nitrogen measured in a filtered sample after conversion to nitrate using an alkaline persulphate digest.

**DON**

Dissolved organic nitrogen derived by subtracting the mineral fraction, NH\(_3\) plus NO\(_x\), from the TSN.

**Particulate N**

The N measured in the particulate residue, after a Kjeldahl digest of the particulate residue.

Differentiation between soluble and particulate fractions was based on the use of 0.7 μm porosity glass fibre filters. These were the smallest porosity filters that were compatible with Kjeldahl digests. In large catchments such as the Darling, a significant fraction of suspended matter can be between 0.2 and 0.003 μm (Hart et al., 1995). Particles in runoff from the farm-scale catchments in the NLP project would appear to be in relatively larger size ranges. A
comparison was made between 0.7, 0.45 and 0.2 μm filters by CSIRO (Cornish et al., 1997) on runoff samples from the NLP project and no significant differences were found.

The sample processing procedures used are summarised in Figure 5-4.

Figure 5-4. Flow diagram showing sample processing methods (Source: Adapted from Cornish et al., 1997)
To thaw the samples, they were placed in flowing warm water (approximately 30°C). Care was taken to avoid heating the samples to more than 25°C. After thawing the samples, they were filtered in duplicate, through 0.7μm glass fibre filter discs. One filter disc was dried and weighed for suspended solids, the other was Kjeldahl digested for particulate N and P. The filtrates were combined and digested with persulphate (peroxodisulfate) to determine TSN and TSP (Cornish et al., 1997).

**Digestion procedures**

The Kjeldahl digest was carried out as follows: wet filter plus solid material was placed into a digestion tube with CuSO₄/Na₂SO₄ catalyst. Concentrated H₂SO₄ was added and the mixture heated to 95°C, then to 140°C to drive off water. The tubes were then heated to 320°C for 60 minutes to complete digestion (Cornish et al., 1997). The persulphate digest was carried out as follows: 15mL aliquots were subsampled at the time of filtering, into polypropylene bottles. Potassium persulphate/ NaOH reagent was added and the bottles were autoclaved at 120°C for 30 minutes (Cornish et al., 1997). An Alpkem Flow Solution 3 continuous flow autoanalysers was used to measure all the species determined, i.e. NH₄⁺, NO₃⁻, and PO₄³⁻.

The limits of detection and precision for suspended solids (SS) and the measured forms of N and P, are shown in Table 5-2.

**Table 5-2 Limits of detection for suspended solids and nutrient forms measured in runoff samples**

(Source: Cornish et al., 1997)

<table>
<thead>
<tr>
<th>Species Measured</th>
<th>Detection Limit in [mg L⁻¹]</th>
<th>Average % difference between duplicates for runoff samples a</th>
</tr>
</thead>
<tbody>
<tr>
<td>Suspended Solids</td>
<td>5</td>
<td>8</td>
</tr>
<tr>
<td>NOx</td>
<td>0.01</td>
<td>4</td>
</tr>
<tr>
<td>Ammonia</td>
<td>0.015</td>
<td>4</td>
</tr>
<tr>
<td>Total soluble N</td>
<td>0.02</td>
<td>3</td>
</tr>
<tr>
<td>Particulate N</td>
<td>0.09</td>
<td>9</td>
</tr>
<tr>
<td>Soluble Reactive P</td>
<td>0.005</td>
<td>6</td>
</tr>
<tr>
<td>Total Soluble P</td>
<td>0.01</td>
<td>4</td>
</tr>
<tr>
<td>Particulate P</td>
<td>0.03</td>
<td>10</td>
</tr>
</tbody>
</table>

a based on 100mL subsample
5.4.6 Calculation of sediment, nitrogen and phosphorus losses

Loads were calculated using the period-weighted method described by Dann, Lynch and Corbett (1986). The average concentration from 2 samples was multiplied by the total discharge (m$^3$) for that period. For the first sample the concentration was multiplied by the quantity of runoff up to that point. For the last sample of an event, the concentration was simply multiplied by the quantity of runoff from the point when the sample was taken to end of the runoff event. According to Dann, Lynch and Corbett (1986) the period-weighted method is the most accurate method for calculating loads provided samples are taken often enough during runoff events.

5.5 RAINFALL MEASUREMENTS AND ANALYSIS

5.5.1 Average recurrence interval (ARI)

The average recurrence interval (ARI) was determined from the intensity-frequency-duration (IFD) table for Richmond obtained from the Department of Engineering at the University of Newcastle.

5.5.2 Sum of rainfall greater than 25 mm h$^{-1}$ ($\sum J > 25$ mm h$^{-1}$) and peak intensity ($I_{\text{max}}$)

The monitoring system used on the case study farm recorded rainfall at 2 minute intervals. Although no kinetic energy measurements were made, rainfall readings equivalent to greater than 25 mm h$^{-1}$ were identified and summed to give $\sum J > 25$ mm h$^{-1}$. Hudson (1965) claims that rainfall at $I > 25$ 25 mm h$^{-1}$ is almost fully responsible for soil erosion in overland flow. Examination of the relationship between rainfall intensity and rainfall energy at Canberra (Figure 3-2) shows that above the rainfall intensity of $I = 25$ mm h$^{-1}$ rainfall kinetic energy ($J$ m$^2$ mm$^{-1}$) ($E$) rapidly levels off and below $I = 25$ $E$ increases sharply with increasing $I$ (Kinnell, 1987). Therefore $\sum J > 25$ mm h$^{-1}$ was calculated for each major runoff event to quantify the total of high energy rain and gives a crude indication of $E$ for a storm.

Peak intensity ($I_{\text{max}}$) was determined by selecting the highest rainfall reading from each storm period and multiplying by 30, to give the intensity in mm h$^{-1}$.
6. RESULTS AND DISCUSSION 1: SOIL AND LAND MANAGEMENT ON THE CASE STUDY FARM

6.1 INTRODUCTION

Agricultural runoff quantity and quality are influenced by soil characteristics and land management. Some soil types are inherently more prone to erosion and therefore difficult to manage. The erodibility of soil is also determined by the way in which it is managed (viz cultivation, soil cover, irrigation). Whether or not the nutrient status of the soil is naturally or artificially high (due to fertiliser inputs), there is also potential for nutrient transport in runoff.

This descriptive Chapter describes specific aspects of land management on the case study farm as a background the Chapter 7, which concerns the farm-scale runoff monitoring.

6.2 SOIL ANALYSES

6.2.1 Effects of market gardening practices on soil qualities

The results discussed here are from bulked, composite samples analysed at the DLWC Scone Research Centre. The results should be viewed with some caution because the sampling was not exhaustive and the areas with <6 months and 20 year vegetable growing histories were located 700 m and 2 km away from the main sampling area, respectively. No statistical analysis was possible and as such, variability was not determined. Nevertheless, the results indicated that some soil qualities changed due to farming history.

Table 6-1 shows the effect of market gardening on surface soil characteristics in the short, medium and long term. Electrical conductivity was higher in cultivated beds than the uncultivated soil. This was probably due to inputs of salts from fertilisers and irrigation water
and possibly also due to an increase in the water table caused by irrigation, bringing low to moderately saline groundwater to the surface. Generally, organic carbon (OC) decreased with cultivation despite the use of organic fertiliser (poultry manure), although it was higher on the Borg farm which had been farmed for 20 years than on the case study farm which had been farmed for 12 years. pH and available P were increased with inputs of lime and fertilisers. The increase in Bray P and decrease in P sorption with the number of years of cropping was striking. Bray P was increased markedly after one heavy poultry manure application. The high concentration of Bray P after one application of poultry manure might be partly due to the short time period between application and sampling (8-10 weeks).

Table 6-1. Comparison of electrical conductivity (EC), pH, organic carbon, available P, and P sorption in soils from the case study and adjacent farms, with different land use history

<table>
<thead>
<tr>
<th>Sample location and history</th>
<th>EC [dS m⁻¹]</th>
<th>pH</th>
<th>Organic carbon [%]</th>
<th>Avail. P (Bray) [mg kg⁻¹]</th>
<th>P sorp [mg kg⁻¹]</th>
</tr>
</thead>
<tbody>
<tr>
<td>unimproved pastureᵃ</td>
<td>0.07</td>
<td>5.2</td>
<td>3.00</td>
<td>5</td>
<td>110</td>
</tr>
<tr>
<td>vegetable, &lt; 6 monthsᵇ</td>
<td>0.51</td>
<td>6.9</td>
<td>1.96</td>
<td>155</td>
<td>133</td>
</tr>
<tr>
<td>vegetable, 10 yearsᶜ</td>
<td>0.15</td>
<td>8.1</td>
<td>0.79</td>
<td>174</td>
<td>129</td>
</tr>
<tr>
<td>vegetable, 12 yearsᵈ</td>
<td>0.41</td>
<td>7.6</td>
<td>0.95</td>
<td>304</td>
<td>104</td>
</tr>
<tr>
<td>vegetable, 20 yearsᵉ</td>
<td>0.26</td>
<td>7.5</td>
<td>1.40</td>
<td>479</td>
<td>56</td>
</tr>
</tbody>
</table>

ᵃ Sample taken from adjacent property, within 50m of monitoring station, March 1995.
ᵇ Additional vegetable growing area, cultivated for the first time, January 1997. Received one application each of poultry manure and lime.
ᶜ Main vegetable growing area, taken within 100m of monitoring station, March 1995.
ᵈ Main vegetable growing area, taken within 100m of monitoring station, January 1997.
ᵉ Vegetable growing area on Borg’s farm, March 1995.

The poor structure of soil on the case study farm was exacerbated by cultivation which leads to oxidation of organic matter (Rose, 1993). Cultivation explains the lower levels of OC found in the cultivated soil compared to the unimproved pasture. In Table 6-1, OC appears to have increased at the sampling point on the case study farm over the 2 year period. A genuine widespread increase in OC in this relatively short period is unlikely considering that management was unchanged.
6.2.2 Role of the drainage channel in sediment and P transport and storage

Particle size distribution of sediment in the drainage channel 100 m and 200 m upslope of the monitoring station was similar to that of the parent soil (Figure 6-1). However, 5 m upslope of the channel, the sediment contained a larger proportion of finer material indicating particle sorting. It is important to note that the channel had been excavated almost 2 years earlier. The samples taken from the drainage channel consisted of sediment deposited within 2 years.

![Graph showing particle size distribution](image)

Figure 6-1. Particle size distribution along the drainage channel and the parent soil

As P is adsorbed predominantly onto finer mineral particles, the concentration of finer particles nearest to the monitoring station would lead to higher P storage than upslope. Table 6-2 shows that total P was indeed higher in the sediment closest to the monitoring station, so too was P sorption. Nutrient enrichment due to preferential sediment transport and deposition can lead to as much as a 10-fold increase in the concentration of nutrients in sediment compared to parent soil (Finlayson & Silburn, 1996).
The results of the core sediment sample analysis (Table 6-2) shows that Bray P appeared to be as much as 10 times lower in the drainage channel sediment than in the original parent soil bearing in mind the results are from single samples. In the absence of total P data for the original soil this is difficult to explain. Further soil and sediment sampling would be required.

There may have been a transformation of labile P to non-labile P, in which case the total P of the parent soil would be equal or less than total P in the drainage channel sediment. There may have been a release of soluble P from the drainage channel sediment due to anoxia during extended periods of inundation, as reported by Mann (1996), in which case total P in the parent soil would exceed total P in the drainage channel sediment. This would imply the potential release of soluble P into the water column and then to Currency Creek and the Hawkesbury River.

### Table 6-2. Plant available (Bray) P, P sorption and total P in drainage channel sediment compared to the vegetable beds

<table>
<thead>
<tr>
<th>Location (m from stn)</th>
<th>Plant available P (mg kg⁻¹)</th>
<th>P sorp (mg kg⁻¹)</th>
<th>Total P (mg kg⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>veg. beds (100)ᵃ</td>
<td>174</td>
<td>129</td>
<td>nt</td>
</tr>
<tr>
<td>drain. ch. (200)ᵇ</td>
<td>16</td>
<td>352</td>
<td>430</td>
</tr>
<tr>
<td>drain. ch. (100)ᶜ</td>
<td>28</td>
<td>332</td>
<td>620</td>
</tr>
<tr>
<td>drain. ch. (5)ᵈ</td>
<td>16</td>
<td>539</td>
<td>750</td>
</tr>
</tbody>
</table>

ᵃ Main vegetable growing area, 0-20 cm, March 1995.
b 0-15 cm depth, January 1997.
c 0-15 cm depth, January 1997.
d 0-15 cm depth, January 1997.
nt = not tested

### 6.3 Land Management Data

#### 6.3.1 Fertiliser inputs

The fertiliser inputs of N and P over the monitoring period are shown in Table 6-3. The values are based on calculations derived from bulk purchases and were slightly lower than inputs
calculated from the farmer’s estimated application rates, for which there is more room for error (method described in Chapter 5).

Table 6-3. Total fertiliser inputs of N and P for the case study farm from June 1995 to May 1997

<table>
<thead>
<tr>
<th>Fertiliser Type</th>
<th>Weight [kg]</th>
<th>Timing</th>
<th>N [kg]</th>
<th>P [kg]</th>
<th>Cost [$]</th>
</tr>
</thead>
<tbody>
<tr>
<td>ammonium nitrate</td>
<td>4000</td>
<td>base and side dressings</td>
<td>1880</td>
<td>0</td>
<td>1560</td>
</tr>
<tr>
<td>poultry manure</td>
<td>112 000</td>
<td>annual from summer-autumn</td>
<td>3584</td>
<td>2408</td>
<td>2800</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td></td>
<td></td>
<td><strong>5464</strong></td>
<td><strong>2408</strong></td>
<td><strong>4360</strong></td>
</tr>
<tr>
<td>kg ha(^{-1}) yr(^{-1})</td>
<td></td>
<td></td>
<td><strong>414</strong></td>
<td><strong>182</strong></td>
<td><strong>$330 ha yr(^{-1})</strong></td>
</tr>
</tbody>
</table>

The values are lower than Mr Saliba’s estimated average annual application rate of 25 m\(^3\) ha\(^{-1}\), which over 6.6 ha equates to 165 m\(^3\). From February to May, 1996, 4 truckloads (each 32 m\(^3\)) of poultry manure were purchased and applied to 6.3 ha (0.3 ha was not fertilised), giving a total of 128 m\(^3\) at a rate of 20.3 m\(^3\) ha\(^{-1}\). (Over the full 6.6 ha this rate equals 134 m\(^3\)). In October, 1996, 5 truckloads of poultry manure were purchased, of which approximately 3 truckloads were applied from February to May, 1997. Senn (pers. comm., 1995) noted a typical rate of 22.5 m\(^3\) ha\(^{-1}\) and a heavy application of 37.5 m\(^3\) ha\(^{-1}\), in the early 90’s. This is consistent with the claim by Mr Saliba that he had more than halved the application of poultry manure since 1993.

P inputs of 182 kg ha\(^{-1}\)yr\(^{-1}\) are considerably lower than the 450 kg ha\(^{-1}\)yr\(^{-1}\) given by Jinadasa et al. (1997). However, Jinadasa et al. (1997) gave a typical P input from poultry manure as 270 kg ha\(^{-1}\)yr\(^{-1}\), which is more consistent with Mr Saliba’s input considering that he halved the rate in 1993. In previous years Mr Saliba used chemical fertilisers containing P. Since completing the fieldwork for this thesis, Mr Saliba purchased a small number of bags of superphosphate so the P inputs may have increased after this study.

The poultry manure was spread on top of the beds and incorporated into the soil with the 100 cm rotary hoe. This was done once per year over most of the cultivation area.
6.3.2 Soil cover and conditions for runoff events since 1996

This section presents the results derived from the farm maps of soil cover and condition, carried out before or after major runoff events. The farm mapping exercise was carried out on the Borg farm as well as on the case study farm. The results for Borg’s farm are also presented to enable some comparison with the case study farm.

Seven runoff events occurred from January, 1996 to the end of the monitoring period. Six of these events occurred as ‘pairs’ separated by 4 weeks or less. During 1996, events occurred in January, February, May, August and September. During 1997, events occurred in January and February. Farm surveys of soil cover and soil condition carried out after an event, were also used for the following event, if it occurred within 4 weeks. As a result, 4 complete surveys were carried out for the 7 events. The surveys were carried out in February, May and September in 1996, and February, 1997. Soil cover was relatively unchanged over such a time frame. Soil condition, however, was affected by the cumulative effect of rainfall resulting in settling and surface seal/crust formation.

Figure 6-2 shows the proportion of each of the three soil cover classes (0-20 %, 21-50 % and >50 %) for the cultivation areas on the case study and Borg farms. Note that the tones depict vegetation cover (black = high, white = low).

Figure 6-2. Soil cover on the cultivation areas of a) the case study farm and b) the Borg farm, as calculated from 4 farm surveys
On the case study farm, apart from May, 1996, 0-20 % cover prevailed over approximately three quarters of the cropping area. In May, approximately 40 % of the cropping area had 0-20 % cover. The area of >50 % cover was considerably larger in February, 1997 than in January and February, 1996. This was due to a sorghum crop that was grown in the 96/97 summer for green manure. Oats was grown the previous year for the same reason but over a smaller area. The 05/96 survey showed considerably larger areas of both >50 % cover and 21-50 % cover. The 08/96 survey of cover showed similar soil cover to the 96/97 summer.

Figures 6-3 shows soil condition for the cultivation areas on the case study and Borg farms. Very little tillage was carried out between closely spaced events due to restricted tractor access in wet soil conditions. Due to the settling action of rainfall on tilled soil, it was difficult to estimate the soil disturbance at the beginning of the event from the surveys that were done after an event.

![Soil condition chart]

**Figure 6-3.** Soil condition* on the cultivation areas of a) the case study farm and b) the Borg farm, as calculated from 4 farm surveys

* SC3 = recently rotary hoed, SC1 = settled surface, SC2 = partially disturbed/settled

According to field observations made on the case study farm over the 2 year monitoring period, less crop cover and greater soil disturbance (tillage) occurred over summer. During this time crop cover consisted of small areas planted to crops such as cucumbers and spinach and some area of land planted with a green manure crop. The sorghum green manure crop
grown in the 96/97 summer occupied a larger area than oats green manure crop grown in the 95/96 summer. Therefore during the summer months a small proportion of the farm was occupied by a high percentage of soil cover, while the majority of the farm had little or no soil cover and large areas of recently cultivated soil.

A comparison of soil cover and soil condition from the case study farm shows an inverse relationship between soil cover and soil disturbance (Figure 6-4), consistent with field observations. Field observations suggested that soil cover and soil disturbance from tillage, weeding and harvesting, were roughly negatively correlated, ie. when a crop provided a high percentage of cover, the crop was at or near maturity and there had been some time since the last tillage. Conversely, when soil was completely bare it had usually been tilled recently in preparation for the next crop. Some paddocks were left in bare fallow for months, allowing the soil to settle, but in such cases weed growth usually provided some soil cover.

![Figure 6-4. Relationship between soil cover and soil disturbance (as measured by SC2 + SC3) on the case study farm](image)

Soil cover is a more reliable parameter than soil condition and not likely to change significantly between closely spaced rainfall events. Table 6-4 shows spatially averaged soil cover for the cultivation area of the case study and Borg farms. The values are derived from the areas of each soil cover class multiplied by the median cover for that class.
Table 6-4. Spatially averaged soil cover [%] on cultivation areas of the 2 market gardens during selected runoff events

<table>
<thead>
<tr>
<th>Event</th>
<th>Saliba cropping area [%]</th>
<th>Borg cropping area [%]</th>
</tr>
</thead>
<tbody>
<tr>
<td>Jan/Feb 96</td>
<td>15</td>
<td>25</td>
</tr>
<tr>
<td>May 96</td>
<td>31</td>
<td>22</td>
</tr>
<tr>
<td>Aug/Sep 96</td>
<td>19</td>
<td>24</td>
</tr>
<tr>
<td>Jan/Feb 96</td>
<td>19</td>
<td>25</td>
</tr>
<tr>
<td>Average</td>
<td>21</td>
<td>24</td>
</tr>
</tbody>
</table>

Considering the crucial role of vegetation cover in soil erodibility, Table 6-4 implies that erodibility varied between runoff events on the cultivation area of the case study farm. However, this variation was within a relatively small range. Because of the heterogeneous nature of the case study farm, it is difficult to compare the effect of cover with other studies where conditions were more homogenous. Nevertheless, on 3 out of 4 occasions total soil cover was less than 30%, the critical value suggested by Freebairn and Wockner (1986) to minimise soil erosion on Vertisols in south-east Queensland. On the Borg farm total soil cover was well below the 30% value on all 4 occasions.

During the cooler months of the year total soil cover was generally higher and consisted generally of larger areas of a larger proportion of the farm with some vegetation, young and mature crops and weeds and crop residue from harvested crops. There were also significant areas of land with no cover and fine tilth, as beds were prepared for the following crop. This indicates the complexity of land management on the case study farm, which is even more evident on the Borg farm, which is more than twice the size of the case study farm.

6.3.3 Irrigation and soil water

Table 6-5 shows the pattern of cropping for sections of the cultivation area on the case study farm over a 14 month period, determined from the farm maps and field notes. Table 6-5 divides the vegetable farming area into 5 sections. The last column gives the percentage of the farming area with crops that would require irrigation, i.e., any vegetable crop that had yet to be harvested.
Table 6-5. Cropping patterns and % of vegetable growing area under irrigation

<table>
<thead>
<tr>
<th></th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>% under irrigation</th>
</tr>
</thead>
<tbody>
<tr>
<td>January 96</td>
<td>L</td>
<td>L, S, C</td>
<td>L, bf</td>
<td>bf</td>
<td>st</td>
<td>30</td>
</tr>
<tr>
<td>February 96</td>
<td>bf</td>
<td>bf, S, C</td>
<td>S, C, L, bf</td>
<td>bf</td>
<td>st</td>
<td>18</td>
</tr>
<tr>
<td>March 96</td>
<td>L, bf</td>
<td>L, S, C</td>
<td>S, C, L, bf</td>
<td>bf</td>
<td>st</td>
<td>22</td>
</tr>
<tr>
<td>May 96</td>
<td>L</td>
<td>L, S, C</td>
<td>S, C, bf, L</td>
<td>L, bf</td>
<td>bf, st</td>
<td>65</td>
</tr>
<tr>
<td>Aug/Sept 96</td>
<td>bf, L</td>
<td>L, S, st</td>
<td>st, L</td>
<td>L, bf</td>
<td>L</td>
<td>51</td>
</tr>
<tr>
<td>November 96</td>
<td>bf, L</td>
<td>L, bf, scc</td>
<td>scc, bf, L</td>
<td>L, scc</td>
<td>scc, bf</td>
<td>23</td>
</tr>
<tr>
<td>Jan/Feb 97</td>
<td>Cu, bf, S</td>
<td>L, s, st, C, S</td>
<td>st, C, bf</td>
<td>BF, scc</td>
<td>scc, bf</td>
<td>18</td>
</tr>
</tbody>
</table>

L = lettuce, S = spinach, C = capsicum, bf = bare fallow, st = weeds, oats or sorghum stubble, Cu = cucumber, scc = sorghum cover crop

Table 6-5 shows that the area of the farm being irrigated varied seasonally. Irrigation was most extensive between May and September. During late spring to early summer the proportion of the vegetable growing area being irrigated was a half or one third of the cooler months.

According to the Salibas, the irrigation frequency and duration was varied according to climatic and soil moisture conditions. Therefore it is reasonable to assume that the soil would rarely have a soil water deficit of greater than 10 mm. If correct, we can assume the soil remains sufficiently wet to allow Stage 1 drying (Cornish, 1983) at a rate equivalent to pan evaporation. The measured irrigation rate was equivalent to approximately 10 mm h\(^{-1}\) and the period of irrigation was variable, but would have typically been 30-60 minutes on any one area of ground.

Figure 6-5 shows a crudely calculated spatially averaged available soil water (\(ASW\)) for the case study farm from December 1995 to May 1996. This was done for this period because 2 contrasting runoff events occurred within this period. The water balance is based on the areas of irrigated, non-irrigated/vegetated (weeds), and bare fallow as extrapolated and interpolated from the farm maps and the set of assumptions outlined in Chapter 5. As mentioned in Chapter 5 the maximum \(ASW\) was assumed to be 70 mm, 6.4 mm above the maximum calculated infiltration rate from the farm-scale monitoring (Chapter 7). This assumption was of little consequence to the calculated spatially averaged \(ASW\).
Figure 6-5. Calculated water balance from 1/12/95 to 9/5/96

It is clearly evident that \( ASW \) differed greatly between non-irrigated/vegetated (weeds) areas and both irrigated and bare fallow areas, particularly during the driest periods (weeks 16-21). Figure 6-7 suggests that runoff characteristics inasmuch as they are affected by antecedent soil water, would differ greatly between the above mentioned categories of land. Unless surface sealing is the major limitation for infiltration, runoff would be generated in the order of irrigated > bare fallow > weeds.

6.4 SUMMARY

For intensive vegetable production, the soil on the case study farm requires management that is likely to produce high rates of soil loss and particulate P. Cultivation has exacerbated the poor structure by oxidising OC. The concentration of soil P was far higher than background concentration. Much of the soil surface on the cultivation area was disturbed due to cultivation and weeding and the degree of soil cover was low. Although fertiliser application rates were reduced several years ago, inputs were still high (414 and 182 kg ha\(^{-1}\) yr\(^{-1}\) for N and P respectively). The soil is dispersive and hard setting and may therefore produce high runoff coefficients. In dry conditions runoff would be generated in the order of irrigated > bare fallow > weeds.

Chapter 6: Land Management on the Case Study Farm
Under these conditions high rates of sediment, N and P loss could be expected in stormwater runoff.
7. RESULTS AND DISCUSSION 2: SUSPENDED SEDIMENT, NITROGEN AND PHOSPHORUS LOSSES IN RUNOFF - JUNE 1995 TO MAY 1997

7.1 INTRODUCTION

The magnitude of sediment and nutrient loss in runoff from market gardens is a contentious issue with environmental, social and economic implications. Data are needed to quantify farm-scale losses. Knowledge of the factors that influence runoff and losses of sediment, N and P should lead to improved farm planning and management and the development of best practice guidelines.

The aims of this chapter are to:

1) quantify soil, N and P losses in runoff from the case study farm; and
2) assess the factors which influenced runoff and associated losses of sediment, N and P.

This Chapter concerns losses of N, P, and suspended sediment in runoff in stormwater. Losses in irrigation runoff were not measured and are likely to be insignificant in comparison to overall losses. The sampling regime and the flow control structure were designed to gain an accurate picture of soil and nutrient runoff losses in conditions when the bulk of exports occur, ie. in larger, less frequent events.

The results are discussed in light of the hydrological and land management conditions that prevailed during each of the major storm events.
7.2 **RESULTS AND DISCUSSION**

7.2.1 **Comparison of rainfall during the monitoring period with long term rainfall records**

Slightly below average rainfall occurred over the 2 year monitoring period. The total rainfall was 1517 mm (av. 758.5 mm yr\(^{-1}\)), approximately 25 mm less than the annual average rainfall for Richmond. The 3 wettest months were January 1996 and 1997, and February, 1997. Rainfall at Richmond is moderately summer dominant (Figure 7-1) although large rainfall events may occur at any time of the year. September is on average the second driest month of the year. However, in September 1995 and 1996, sizeable rainfall events occurred.

![Graph showing monthly rainfall data](image)

**Figure 7-1.** Comparison of measured rainfall at Currency Creek and long term monthly average rainfall for Richmond

Figure 7-1 shows that despite the slightly lower than average annual rainfall, 12 of the 24 months received equal to, or greater than the monthly average. Three dry spells occurred over the duration of the study, from June to August, 1995, from February to April, 1996 and from March to April, 1997.
7.2.2 Hydrological aspects of major runoff events

Table 7-1 includes rainfall events that were in excess of 25 mm and describes rainfall totals, 2 measures of rainfall intensity (I) as calculated from the 2 minute rainfall readings; peak intensity ($I_{\text{max}}$) and the sum of rain that fell at greater than 25 mm h$^{-1}$ ($\sum I>25$ mm h$^{-1}$).

Shown in Table 7-1 is the average recurrence interval (ARI) of each event (based upon rainfall intensity-duration-frequency table for Richmond) average recurrence interval for each event, runoff depth, peak discharge, and runoff coefficient (runoff as % of total rain).

Table 7-1. Hydrological characteristics of major runoff events

<table>
<thead>
<tr>
<th>Event No.</th>
<th>Date</th>
<th>Total [mm]</th>
<th>$\sum I&gt;25$ [mm]$^a$</th>
<th>$I_{\text{max}}$ [mm h$^{-1}$]</th>
<th>ARI [yr]</th>
<th>Depth [mm]</th>
<th>Peak discharge [m$^3$ min$^{-1}$]</th>
<th>Runoff coefficient [%]</th>
<th>Calculated infiltration [mm]$^b$</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>24-25/09/95</td>
<td>94.0</td>
<td>10.2</td>
<td>38.1</td>
<td>1-2</td>
<td>85.3</td>
<td>13.7</td>
<td>91</td>
<td>8.7</td>
</tr>
<tr>
<td>2</td>
<td>19-22/11/95</td>
<td>84.6</td>
<td>0</td>
<td>23.1</td>
<td>&lt;1</td>
<td>49.4</td>
<td>3.1</td>
<td>58</td>
<td>35.2</td>
</tr>
<tr>
<td>3</td>
<td>01/12/95</td>
<td>25.3</td>
<td>4.4</td>
<td>34.0</td>
<td>&lt;1</td>
<td>14.6</td>
<td>4.7</td>
<td>58</td>
<td>10.7</td>
</tr>
<tr>
<td>4</td>
<td>05-6/12/95</td>
<td>24.8</td>
<td>7.9</td>
<td>37.8</td>
<td>&lt;1</td>
<td>23.1</td>
<td>5.8</td>
<td>93</td>
<td>1.7</td>
</tr>
<tr>
<td>5</td>
<td>02-3/01/96</td>
<td>29.1</td>
<td>2.1</td>
<td>32.1</td>
<td>&lt;1</td>
<td>8.8</td>
<td>3.7</td>
<td>30</td>
<td>21.2</td>
</tr>
<tr>
<td>6</td>
<td>06-7/01/96</td>
<td>26.1</td>
<td>0</td>
<td>19.5</td>
<td>&lt;1</td>
<td>16.0</td>
<td>6.6</td>
<td>61</td>
<td>10.1</td>
</tr>
<tr>
<td>7</td>
<td>19-20/01/96</td>
<td>64.5</td>
<td>46.6</td>
<td>115.8</td>
<td>1</td>
<td>28.0</td>
<td>22.2</td>
<td>43</td>
<td>36.5</td>
</tr>
<tr>
<td>8</td>
<td>12/04/96</td>
<td>20.8</td>
<td>0</td>
<td>21.5</td>
<td>&lt;1</td>
<td>12.6</td>
<td>6.7</td>
<td>61</td>
<td>8.2</td>
</tr>
<tr>
<td>9</td>
<td>01-6/05/96</td>
<td>117.6</td>
<td>25.9</td>
<td>67.8</td>
<td>&lt;1</td>
<td>101.0</td>
<td>6.7</td>
<td>86</td>
<td>16.6</td>
</tr>
<tr>
<td>10</td>
<td>27-28/07/96</td>
<td>38.5</td>
<td>1.3</td>
<td>37.6</td>
<td>&lt;1</td>
<td>14.1</td>
<td>5.7</td>
<td>37</td>
<td>24.4</td>
</tr>
<tr>
<td>11</td>
<td>30-31/08/96</td>
<td>81.3</td>
<td>0</td>
<td>13.5</td>
<td>1</td>
<td>44.9</td>
<td>5.3</td>
<td>55</td>
<td>36.4</td>
</tr>
<tr>
<td>12</td>
<td>28-30/09/96</td>
<td>54.1</td>
<td>0</td>
<td>22.0</td>
<td>&lt;1</td>
<td>18.7</td>
<td>4.2</td>
<td>35</td>
<td>35.4</td>
</tr>
<tr>
<td>13</td>
<td>28-30/01/97</td>
<td>115.8</td>
<td>19.5</td>
<td>48.3</td>
<td>1</td>
<td>52.2</td>
<td>4.5</td>
<td>45</td>
<td>63.6</td>
</tr>
<tr>
<td>14</td>
<td>10-12/02/97</td>
<td>164.6</td>
<td>8.1</td>
<td>80.0</td>
<td>5</td>
<td>156.4</td>
<td>19.0</td>
<td>95</td>
<td>8.2</td>
</tr>
<tr>
<td><strong>TOTAL</strong></td>
<td></td>
<td><strong>941.1</strong></td>
<td></td>
<td></td>
<td></td>
<td><strong>625.1</strong></td>
<td></td>
<td></td>
<td><strong>66</strong></td>
</tr>
</tbody>
</table>

$^a$ Rainfall was recorded at 2 minute intervals. $\sum I>25$ is the sum of rainfall that fell at a rate equivalent to greater than 25 mm h$^{-1}$.

$^b$ Infiltration = rainfall - runoff

The largest events were not necessarily the most intense. Event 7 received 46.6 of 64.5 mm at $I_{>25}$ (intensity of greater than 25 mm h$^{-1}$). The same event also had the highest peak rainfall intensity and the highest peak discharge. In terms of total rainfall, the largest event was 14,
though only 8.1 mm fell at $I_2>25$. The February, 1997 event produced the second highest peak discharge at 19 m$^3$min$^{-1}$.

Table 7-1 shows that each runoff event occurred in a unique set of hydrological circumstances. We can consider that there are 2 extremes of events: a short but intense storm, and an extended rainfall event. Between these extremes there is an infinite range of possible events. Rather than describing each event in detail, which would be a lengthy and unnecessary task one event representing each extreme is described.

A long-lasting rainfall event

Event 9 (May, 1996) occurred over 5 days. A total of 117.6 mm fell in a sporadic fashion, with $I_{\text{max}}$ of 67.8 mm h$^{-1}$ and $\Sigma I_2>25$ of 25.9 mm. Figure 7-2 shows discharge over the duration of the event. Rainfall and runoff are shown cumulatively in Figure 7-2b to demonstrate the relationship between rainfall and runoff.

A short, intense event

Event 7 (January, 1996) was in stark contrast to event 9. It was a more abrupt event that occurred over a 10 hour period with higher intensity rainfall. The event comprised 2 almost identical peaks in flow. Rainfall intensity was very similar for the 2 peaks with approximately 20 mm falling in 8 minutes during both peaks. $I_{\text{max}}$ was 115.8 mm h$^{-1}$ and $\Sigma I_2>25$ was 46.6 mm for the 2 peaks combined.
Figure 7-2. (a) Standard hydrograph, and (b) cumulative rainfall and runoff, for event 9 (May 96)
Variations in runoff

Between events 7 and 9

Figure 7-2b shows that runoff exceeded rainfall at the end of event 9, indicating that it was fed by throughflow from the raised vegetable beds. The sporadic falls of rainfall coincided with virtually immediate, but slightly dampened rises in runoff.

Calculated infiltration for event 9 was 16.6 mm. Calculated infiltration for event 7 was 36.5 mm. Differences in infiltration could be due to the effect of: (i) antecedent soil moisture, and/or (ii) surface sealing and/or (iii) spatial variability in antecedent soil water creating source areas, such as the drainage. The water balance presented in Chapter 6 (Figure 6-4) shows that ASW in the week preceding event 7 and event 9 was 46 mm and 47 mm respectively. This suggests that infiltration was restricted in event 9 by surface sealing and/or that runoff was primarily from ‘source’ areas where there was little infiltration. Examination of Figure 6-3 shows that the soil surface was classed as SC2 and SC3 across 100 % of the cultivation area after event 7, compared to less than 30 % after event 9. Therefore it seems reasonable to conclude that the surface seal/crust was disrupted during event 7, thus allowing 20 mm more infiltration. However, the drainage channel would also have been wetter during event 9 due to a more extensive irrigation area (therefore more irrigation runoff in the channel). It is difficult
to conclude exactly what factors were most significant based on this data. However, plot-scale data in Chapter 8 provides an indication of the importance of surface sealing.

**Between all events**

Runoff coefficients varied from 30 % to 95 %, with an average of 66 %. Chapter 6 and Figure 6-4 show that on the case study farm soil moisture varied significantly both spatially and temporally due to variations in evapotranspiration (ET) and soil evaporation ($E_s$), which are functions of land management and prevailing climatic conditions. Soil condition also varied markedly. As a result, runoff coefficients varied greatly and that there was no strong relationship between runoff coefficient and the amount of rain (Figure 7-4). This variability is typical of rural catchments (Sharpin, 1992).

![Figure 7-4. Comparison of rainfall totals and runoff coefficients](image)

**7.2.3 Forms of N and P in runoff**

**Nitrogen**

Figure 7-5 shows the mean sample concentrations of the analysed forms of N in samples grouped into the runoff events. The mean concentrations of total N varied from 16.35 mg L$^{-1}$
in event 12 (September, 1996) to 49.4 mg L\(^{-1}\) in event 9 (May, 1996). The dominant forms of N were generally NO\(_x\) and particulate N. On average NO\(_x\) -N was 59% of total N. Concentrations of NO\(_x\) were considerably higher for events 2 (November, 1995) and 9 (32.78 and 36.44 mg L\(^{-1}\) respectively). Particulate N was 32% of total N. The relative proportions of the different forms of N were variable, which is not surprising given the rapid transformations that occur in both poultry manure and soil. N is also applied in large amounts in different forms (Chapter 6) leading to potential temporary spikes in concentrations of quite different forms of N.

![Figure 7-5. Mean sample concentrations of N forms in runoff events](image)

DON concentrations exceeded particulate N for events 1 and 2 (September, 1995 and November, 1995) when they were 13 and 16% of total N, respectively.

Soluble N was dominated by NO\(_x\), and the amount of dissolved organic N (DON) was relatively small, and NH\(_4^+\) was very small (Figure 7-5). Ammonium (NH\(_4^+\)) is adsorbed to negatively charged particulate matter, so it is not surprising that ammonium was present in
small concentrations in runoff containing an abundance of fine mineral particles. The low levels of DON implies that N cycling in soil was important, and suggests that nitrification occurred in the stockpile and/or after application of poultry manure (the largest source of applied N), converting DON to NO₂ and then NO₃. This possibly occurred even before the manure was delivered to the farm.

**Phosphorus**

Figure 7-6 shows the mean sample concentrations of the analysed forms of P, in samples grouped into the runoff events. Mean concentrations of total P ranged from 0.33 mg L⁻¹ in event 2 to 9.79 mg L⁻¹ in events 5-7. Respectively, these are approximately 3 and 100 times the ANZECC guidelines for the protection of freshwater ecosystems, which is 0.1 µg L⁻¹ (McMahon & Hart, 1996).

![Figure 7-6. Mean sample concentrations of P forms in runoff events](image)

The concentrations of total P varied significantly between runoff events, due largely to variations in particulate P which was as more than 90 % of total P for events where total P concentration was over 2.5 mg L⁻¹. The concentrations of soluble P were relatively stable, though a slight increase in soluble P occurred as time progressed. Samples from events 5-7 contained the highest concentrations of P and the largest proportion of particulate P (98 %).
Event 2 had the lowest concentration of particulate P. The mean concentration of soluble reactive P ranged between 0.1-0.27 mg L$^{-1}$.

7.2.4 Behaviour of sediment, N and P concentrations during one event

Figure 7-7 is a pollutograph for event 7 and shows concentrations of suspended solids, N and P throughout the event. Event 7 was chosen for this purpose because the rising and falling limbs of the hydrograph are clearly defined and also the 2 peaks are very similar, enabling comparison of pollutant concentrations between the peaks.

In the first storm flow, soluble N concentrations showed a combination of the ‘first flush’ effect followed by ‘growth’ on the receding limb of the first flow. During the second flow soluble N showed ‘decay/growth’ behaviour as described by Sharpin (1992). This suggests soluble N loss was not determined by storm intensity.

Although soluble P was very small in comparison to particulate P and does not show any visible pattern, examination of the data shows that a peak in soluble P was 0.46 mg L$^{-1}$, which occurred at 453:36 hours, consistent with coincident behaviour described by Sharpin (1992). Suspended solids, particulate N and particulate P showed a combination of ‘first flush’ (or lead-lag) and ‘coincident’ behaviour as described by Olive and Walker (1982) and Sharpin (1992). In the first runoff peak there was a sharp increase in sediment concentration with discharge and the peak in concentration occurred before the peak in discharge, in first flush fashion. According to Olive and Walker (1982) this response is typical for sediment and generally indicates depletion of detached particles. In the second runoff peak, the peak in sediment concentration coincided closely with the peak in discharge, in coincident fashion, but then followed by an abrupt drop in concentration.
Figure 7-7. Concentrations of a) suspended solids, b) N and c) P through event 7

Chapter 7: Suspended Sediment, Nitrogen and Phosphorus Losses in Runoff

Page 109
Concentrations of suspended solids and particulate forms of N and P were at least double for the second peak, indicating that either conditions were more erosive or the soil was more erodible during the second peak. As pointed out previously, the rainfall intensity was very similar during both peaks, hence erosivity was virtually identical. It would appear the major difference in conditions between the peaks was antecedent soil water, though this had little effect on runoff volume according to similarities between the hydrograph and rainfall for both peaks.

Morgan (1986) describes a process where compression of air ahead of the wetting front, as rainfall penetrates the soil, leads to slaking. In a rainfall simulation study, Rudolf et al. (1993) reported that aggregate breakdown was greater in a one week wetting-drying-wetting cycle (12 mm applied 5 times one week apart) than in one continuous storm equal to the total rainfall (60 mm applied in one storm) applied at the same rainfall intensity. In this case, the cycle was not wetting-drying-wetting, but wetting-draining (in the soil surface)-wetting because storm peaks were separated by only 6 hours. Because wetting of aggregates can cause dispersion (Emerson, 1967) and the soil on the case study farm is moderately dispersive it is possible that the increased concentration of suspended sediment in the second peak was due at least partly to dispersion caused by increased soil moisture from the first storm.

The only conclusion that can really be made is that it is very difficult to determine the cause of the marked increase in concentration of suspended sediment without more detail. In particular, was the sediment eroded uniformly across the farm or was it generated from a ‘source’ area such as the drainage channel or certain beds?

7.2.5 Variations in concentrations of soluble N and P in runoff over the study

Figure 7-8 shows the event concentrations (loss/total discharge) of soluble N and P for the runoff events. Figure 7-8a shows that the concentrations of soluble N varied widely between and during events. Event 9 followed a large application of poultry manure, which seems the probable cause of very high concentrations of soluble N, most of which was NO₃. This
indicates that application of poultry manure was linked with NO$_x$, suggesting that nitrification occurred in the poultry manure at some stage.

Under anoxic conditions denitrification may occur, resulting in conversion of NO$_x$ to gaseous N, thereby causing a decrease in NO$_x$ concentration in runoff after a period of anoxic conditions (Sims & Wolf, 1994). There is no strong evidence of this in Figure 7-8a, although, soluble N decreased from events 13 to 14 and from events 11 to 12.

The concentrations of soluble P were small compared to concentrations of particulate P. There was, however, a steady rise in the concentration of soluble P over the duration of the monitoring period (Figure 7-8b). The slope of the regression ($b = 0.3 \mu g \ L^{-1}$ of soluble P per day after the first runoff event), was statistically significant ($P < 0.01$). This was probably linked with an increase in ‘plant available’ P in the bulked soil samples from 174 mg kg$^{-1}$ at the beginning of the monitoring period to 304 mg kg$^{-1}$ in January 1997 and concomitant decrease in P sorption from 129 to 104 mg kg$^{-1}$ (Table 6-1) over the monitoring period. Sharpley (1995) reported that soluble and bioavailable P in runoff is linked to Mehlich-3 P (an alternative extractant Bray P) content of the soil. He reported that dissolved P ($r^2 = 0.86$) and bioavailable P ($r^2 = 0.85$) in runoff were correlated with P sorption saturation in soil.

Figure 7-8b shows that the concentration of soluble P in runoff varied considerably within a time period of a few days to a few weeks. There was an increase in minimum, maximum, and mean soluble P concentrations between events 13 and 14 (January and February, 1997). A small increase occurred between events 11 and 12 (August and September, 1996). In both cases there were no additions of P-containing fertiliser, and very little activity in terms of tillage, harvesting of crops, or planting of new crops. This indicates that inorganic P was desorbed after the first event and/or organic P was mineralised to inorganic soluble P. A possible further explanation is that saturation of the soil profile created anoxic conditions that resulted in chemical reduction of P from an insoluble to soluble form. A similar but more likely explanation is that P was released from sediment in the drainage channel, due to anoxic conditions. This process would be similar to chemical reduction of P in substrate sediment in constructed wetlands, reported by Mann (1996). It was often observed that after rainfall
events, the drainage channel did remain in a waterlogged state for a more extended period than

![Graph showing nitrogen and phosphorus concentrations over time]

Figure 7-8. Minimum, maximum, and mean concentrations of soluble N (a) and P (b)

7.2.6 Comparison of runoff quality from the case study farm and the Borg farm

A question which arises from this work is how representative is the case study farm of other market gardens in the area, particularly those on a similar soil type? This discussion is included
to provide an indication of the representativeness of the case study farm. This is also a useful comparison to make because the Borg farm has been a market garden for 20 years compared to 12 years for the case study farm. Therefore a comparison of runoff quality from these 2 farms may indicate the influence of farming history.

During event 7, runoff samples (n=4) were collected from the Borg farm that did not contain runon from upstream land uses. These samples represent runoff only from the market garden area of the catchment. Runoff samples (n=6) were also collected from the case study farm within the same period. A comparison of concentrations in runoff from the 2 farms during event 7 is given in Table 7-2.

<table>
<thead>
<tr>
<th>Table 7-2. Comparison of runoff quality between the case study farm (Saliba) and the Borg farm during event 7</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
</tr>
<tr>
<td>----</td>
</tr>
<tr>
<td>Saliba mean</td>
</tr>
<tr>
<td>range</td>
</tr>
<tr>
<td>Borg mean</td>
</tr>
<tr>
<td>range</td>
</tr>
</tbody>
</table>

The most significant difference in runoff from the 2 farms is the concentration of soluble P (TSP) which is in the order of 10 times higher from the farm with the longer market gardening history. This suggests that with continued heavy use of P-containing fertilisers, in particular poultry manure, the apparent increase in concentrations of soluble P over the monitoring period as shown in Figure 7-8b, is likely to continue to rise in the long term. Needless to say, the implications of increasing concentrations of soluble P on receiving water quality are serious. On the Borg farm, concentrations of soluble P alone are well above ANZECC guidelines for the protection of freshwater ecosystems (MacMahon & Hart, 1996).
7.2.7 Calculated sediment, nitrogen and phosphorus losses

Table 7-3 summarises N, P, and suspended solids exports from the major runoff events. Suspended solids are also shown as depth. A most noticeable feature of Table 7-3 is the magnitude of soil loss, which was equivalent to more than 19 t ha\(^{-1}\) yr\(^{-1}\) or 2.5 mm of soil over 2 years. It is clearly evident that infrequent, high intensity events result in very large losses, as widely reported in the literature. However, event 7 which was equivalent to a 1 year ARI event, resulted in nearly twice as much soil loss as event 14 which had an ARI of 5 years.

<table>
<thead>
<tr>
<th>Date</th>
<th>Soluble N [kg]</th>
<th>Particulate N [kg]</th>
<th>Soluble P [kg]</th>
<th>Particulate P [kg]</th>
<th>SS [kg]</th>
<th>Soil depth(^{a, b}) [mm]</th>
<th>SS/runoff [kg m(^{-3})]</th>
</tr>
</thead>
<tbody>
<tr>
<td>19-21/09/95</td>
<td>10.7</td>
<td>0.6</td>
<td>0.1</td>
<td>0.2</td>
<td>266</td>
<td>0.003</td>
<td>0.5</td>
</tr>
<tr>
<td>24-26/09/95</td>
<td>94.1</td>
<td>15.6</td>
<td>1.2</td>
<td>6.6</td>
<td>20 281</td>
<td>0.215</td>
<td>2.8</td>
</tr>
<tr>
<td>19-22/11/95</td>
<td>175.6</td>
<td>3.3</td>
<td>0.9</td>
<td>0.8</td>
<td>976</td>
<td>0.010</td>
<td>0.2</td>
</tr>
<tr>
<td>1-2/12/95</td>
<td>23.1</td>
<td>6.5</td>
<td>0.2</td>
<td>1.8</td>
<td>3 109</td>
<td>0.033</td>
<td>2.2</td>
</tr>
<tr>
<td>5-6/12/95</td>
<td>36.3</td>
<td>3.5</td>
<td>0.3</td>
<td>1.2</td>
<td>1 778</td>
<td>0.019</td>
<td>0.9</td>
</tr>
<tr>
<td>2-16/01/96</td>
<td>68.5</td>
<td>5.6</td>
<td>0.5</td>
<td>1.6</td>
<td>3 553</td>
<td>0.038</td>
<td>1.6</td>
</tr>
<tr>
<td>19-20/01/96</td>
<td>28.5</td>
<td>64.2</td>
<td>0.7</td>
<td>26.3</td>
<td>71 069</td>
<td>0.752</td>
<td>19.7</td>
</tr>
<tr>
<td>27-29/02/96</td>
<td>9.3</td>
<td>10.6</td>
<td>0.2</td>
<td>4.1</td>
<td>6 290</td>
<td>0.067</td>
<td>8.0</td>
</tr>
<tr>
<td>2-6/05/9696</td>
<td>403.0</td>
<td>45.4</td>
<td>1.7</td>
<td>20.3</td>
<td>34 279</td>
<td>0.363</td>
<td>4.0</td>
</tr>
<tr>
<td>30-31/08/96</td>
<td>82</td>
<td>108.4</td>
<td>0.8</td>
<td>21.9</td>
<td>30 223</td>
<td>0.320</td>
<td>7.7</td>
</tr>
<tr>
<td>28-30/09/96</td>
<td>29.6</td>
<td>10.4</td>
<td>0.6</td>
<td>3.9</td>
<td>7 449</td>
<td>0.079</td>
<td>3.9</td>
</tr>
<tr>
<td>26-31/01/97</td>
<td>111.7</td>
<td>53.2</td>
<td>1.0</td>
<td>19.1</td>
<td>36 170</td>
<td>0.383</td>
<td>7.8</td>
</tr>
<tr>
<td>11-14/02/97</td>
<td>151.8</td>
<td>47.6</td>
<td>4.7</td>
<td>17.9</td>
<td>30 585</td>
<td>0.324</td>
<td>2.3</td>
</tr>
<tr>
<td>TOTAL</td>
<td>1224.2</td>
<td>374.9</td>
<td>12.9</td>
<td>125.7</td>
<td>239 490</td>
<td>2.53</td>
<td>4.4</td>
</tr>
</tbody>
</table>

\(^{a}\) Calculated using bulk density @ 1500 kg m\(^{-3}\)  
\(^{b}\) Assuming losses are only from cultivation area (6.3 ha)

The rate of soil loss is many times higher than losses recorded from a 7 ha pasture area in Wagga Wagga (0.02-1.6 t ha yr\(^{-1}\)) reported by Adamson (1974), but is less than losses recorded from a one ha bare field (57, 122, 180 t ha\(^{-1}\)) from individual storms reported by Marston and Perrens (1980) and less than total sediment loss from one ha contour bays on the Darling Downs, Queensland (370 t ha\(^{-1}\) over a 4 yr period) reported by Freebairn and Wockner (1986). Nevertheless, the magnitude of soil loss is high, particularly, considering the slow rates of soil formation in Australia.
Total N loss was very high, in the order of 127 kg ha\(^{-1}\) yr\(^{-1}\), more than 5 times higher than the ‘best’ estimate for CMSS (Marston, 1994). Soluble N loss was 76% of total N loss.

Total P loss was equivalent to 11 kg ha\(^{-1}\) yr\(^{-1}\), within the CMSS range of 2.7-14.3 kg ha\(^{-1}\) yr\(^{-1}\), but well above the ‘best’ estimate of 7.3 kg ha\(^{-1}\) yr\(^{-1}\). Unlike N loss, P loss was more than 90% particulate.

7.2.8 Factors influencing losses of sediment, particulate N and particulate P

Given that the export of suspended solids from the case study farm was large and that much of the P and some of the N was transported as particulate material, it is logical to examine the variables that influenced exports and concentrations of suspended solids. A number of spatial and temporal variables influence soil erosion: rainfall erosivity, soil erodibility (determined by a complex range of factors including land management), topography, and so on.

Sediment

Stepwise regression was carried out for total sediment loss for 7 events where soil cover data was available (events 7 to 14). Four variables were tested against suspended sediment loss:

- \(\Sigma J > 25\) mm h\(^{-1}\) [mm]
- peak discharge [m\(^3\) min\(^{-1}\)]
- runoff [mm]
- spatially averaged soil cover %

Results indicated that \(\Sigma J > 25\) mm h\(^{-1}\) \((R^2 = 0.75, P = 0.011)\) was the only significant variable that influenced suspended sediment (SS) loss, for which the equation was:

\[
SS = 1114 * \Sigma J > 25\) mm h\(^{-1}\) + 13866
\]

Inclusion of peak discharge \((R^2 = 0.49, P = 0.079)\) improved prediction by approximately 5%.

Figure 7-9 shows losses of suspended solids against peak discharge rate [m\(^3\) min\(^{-1}\)] and \(\Sigma J > 25\) mm h\(^{-1}\), for each of the major runoff events (n=11).
\( \sum I > 25 \text{ mm h}^{-1} \text{ [mm]} \) provides a crude qualitative measure of total kinetic energy of rainfall that occurred during each runoff event. Rainfall intensity influences rainfall kinetic energy, i.e., high intensity rain is generally associated with large drop sizes, and drop size determines velocity and therefore kinetic energy. Figure 7-9 shows that event 7 (January, 1996) was characterised by a high peak discharge and a relatively large \( \sum I > 25 \), that combined to produce the highest loss of sediment. The correlation is stronger for \( \sum I > 25 \) (R = 0.868) than for peak discharge (R = 0.702).

![Graph showing suspended solids vs. peak runoff](image)

![Graph showing suspended solids vs. sum of rain > 25 mm/h](image)

Figure 7-9. Comparison of (a) peak discharge [m³ min⁻¹] and (b) \( \sum I > 25 \text{ mm h}^{-1} \text{ [mm]} \), with sediment loss [kg]

---

*Chapter 7: Suspended Sediment, Nitrogen and Phosphorus Losses in Runoff*
Event 7 had the largest $\Sigma I_2>25$ and by far the largest loss of sediment, despite the fact that it was not a very large event in terms of discharge. Event 11 (August, 1996) is the most outlying point from the line of best fit, with approximately 30 t of suspended solids lost even though $I_{\text{max}}$ was only 13.5 mm h$^{-1}$ (Table 7-1). Soil cover on the farm at the time was 19%, but more importantly, examination of the farm survey shows that much of the bare soil on the farm at that time was at the downstream end of the farm (i.e. nearest the monitoring station) and was also in a heavily disturbed condition (SC3). Soil eroded from the beds nearest the monitoring station has less opportunity to be deposited before leaving the farm. This ‘scale’ effect complicates interpretation of the results, and for this reason, correlations between land management factors and runoff losses are less than for smaller field or plot studies. Nevertheless, the event 11 shows that soil cover and soil condition probably explain variability not accounted for by erosivity parameters, viz $\Sigma I_2>25$, peak discharge and $I_{\text{max}}$.

In order to separate the influence of event magnitude so that other parameters can be compared with sediment losses, the total sediment losses can be divided by total discharge to give mean concentrations. Figure 7-10a shows peak intensity [mm h$^{-1}$] plotted against the concentration of sediment [kg ML$^{-1}$] and shows that it had some influence, but only at the highest intensity (event 7). Event 7 had the highest peak intensity and the highest concentration of sediment in runoff. Figure 7-10b compares soil condition with the concentration of suspended solids [kg ML$^{-1}$]. Factors which weakened the relationship between cover and soil loss included the strong influence of antecedent rainfall on settling of the soil surface, heterogenous management at farm scale and too few data.

A high degree of soil disturbance, such as SC3, creates conditions where export of sediment is limited by the capacity of runoff water to transport sediment, commonly termed ‘transport limited’, rather than conditions that normally prevail where the limiting factor is the capacity of rain and runoff to detach soil particles, commonly termed ‘detachment limited’ (Rose, 1993). When cultivation produces fine tilth, soil particles are easily detached or entrained because they are not consolidated. In these conditions, transport of particles from the farm are limited only by the transporting capacity of overland flow, thus producing high sediment concentrations.
Figure 7-10. Comparison of (a) peak rainfall intensity [mm h⁻¹], (b) the area of disturbed soil (SC2 + SC3) [ha], and (c) total soil cover [%], with concentration of suspended solids [kg ML⁻¹].
Farm surveys were done between events 7 and 8 (January and February, 1996), events 11 and 12 (September, 1996), and events 13 and 14 (February, 1997). For each of the 3 pairs of events, the concentration of suspended solids was more than double in the first event than the second (Figure 7-10b). This highlights the difference between transport-limited conditions following cultivation compared to detachment-limited conditions, where the soil surface is settled. For event 7, the high degree of soil disturbance combined with high storm intensity to produce a very high concentration of suspended solids.

The area of soil disturbance is negatively correlated with total soil cover (Figure 6-4). Therefore, high erodibility due to relatively low soil cover would be exacerbated by high soil disturbance that could combine to produce transport-limited sediment export. Figure 7-10c shows total soil cover plotted against concentration of suspended solids. As for Figure 7-10b, Figure 7-10c shows there is a strong influence of antecedent rainfall, most noticeable from event 13 to event 14 (January to February, 1997).

Although the event 9 (May, 1996) was a long, drawn out one, it also produced the 3rd highest $I_{\text{max}}$ and the 2nd highest $\sum I > 25$. The influence of storm intensity of event 9 should have generated high concentrations of suspended solids, but this influence was counteracted by the soil management conditions of relatively low soil disturbance and higher percentage of soil cover. Although there are few data points, Figure 7-10c shows a relationship that is consistent with that reported by others (Costin, 1980; Freebairn & Wockner, 1986) considering the very heterogenous nature of cover on the case study farm.

Because $\sum I > 25$ is a function of rainfall intensity ($I$) and rainfall kinetic energy ($E$) is a function of $I$ (Rosewell, 1986; Kinnell, 1987) this implies that $E$ influenced sediment loss. Further, this implies that soil cover, in the form of crops, weeds and stubble, is an important land management factor that influences sediment loss. Moss (1989) describes the critical role that plant cover plays in absorbing raindrop energy, thereby reducing the impact of drops on the soil surface.
Particulate N and P

Nitrogen

There was a significant correlation between loss of sediment and particulate N (Figure 7-11a). Apart from ammonium-N, N is not strongly associated with sediment, so it is likely that some of the particulate N was organic, derived either from the poultry manure, or from crop residue. To confirm this it would be necessary to separate organic from mineral particulate material in the runoff samples. Event 11 (August, 1996) is conspicuous in its comparatively high loss of particulate N. This may have been due to i) erosion of temporarily stored adsorbed ammonium-N, and/or ii) erosion of organic N from poultry manure. The elevated particulate N loss of event 11 may have been a result of addition of both poultry manure and chemical N fertiliser over the period from February to June, 1996, combined with prevailing N cycling processes. For example, in the ammonification of complex organic and labile organic N in poultry manure described by Sims and Wolf (1994) ammonium is the product which is positively charged and can thus be temporarily adsorbed onto clay particles. Event 11 was by far the biggest in terms of loss of particulate N. The bulk of the poultry manure applied that year was applied prior to event 9 and event 9 resulted in more than double the discharge of event 11. Yet, particulate N export in event 9 was less than half of event 11. This supports the theory of ammonification of organic N in applied poultry manure and temporary storage of adsorbed ammonium.

The high levels of particulate N in event 11 may have also been influenced by spillage of poultry manure within close proximity of the monitoring station.

The export of particulate N was not strongly influenced by total discharge. The loss of particulate N was 5 % less for event 14 (February, 1997) than for event 13 (January, 1997) even though total discharge was 3 times higher in event 14.

Phosphorus

Particulate P can include adsorbed inorganic P and organic particulate P. The chemical analysis did not separate these fractions; this has to be estimated, taking into account the nature of the soil, including its clay and organic matter content. Given the soil was comprised of approximately 25 % clay, has a high but decreasing P sorption capacity, and was relatively
low in organic matter (<1 % OC), it is likely that most of the particulate P was inorganic adsorbed P. The exports of particulate P in the runoff samples were closely correlated with exports of suspended solids (Figure 7-11b).

Figure 7-11. Comparison of exports of suspended solids [kg] with (a) particulate N [kg] and (b) particulate P [kg]

The implication of Figure 7-11b is that loss of mineral particles (soil erosion) is the major mechanism for export of particulate P. Most of the variability of particulate P was due to
variability in loss of sediment, indicating that most of the particulate P was inorganic/adsorbed. Furthermore, event 7 (January, 1996) was the most intense storm, so the highly erosive conditions would have resulted in larger particle sizes, with proportionally less adsorbed P. This explains why event 7 fell below the line of best fit. Similarly, event 11 was the least intensive event ($I_{\text{max}}$ was 13.5 mm h$^{-1}$) so erosion/deposition processes would have been highly size selective, transporting only the finest particles that proportionally have more adsorbed P.

Given that particulate N and P loss was largely dependent on export of suspended solids, the same variables that influenced sediment loss, influenced loss of particulate N and P.

7.2.9 Factors influencing losses of soluble N and P

Figure 7-12 shows export of soluble N compared to discharge. The variability in the total exports (Figure 7-12) and proportions of the forms of N in the runoff samples (Figure 7-5) is not surprising given the dynamic nature of N (due to N cycling processes that can cause rapid transformations from one form to another), the mix of organic and chemical fertilisers, and temporal and spatial variability of N fertiliser application.

![Figure 7-12. Comparison of discharge [m$^3$] and loss of soluble N [kg]](image)
The May 1996 event was significant, because very high NO\textsubscript{x} concentrations and loss of soluble N followed application of the largest single input of poultry manure during the monitoring period. Another event with very high concentrations of NO\textsubscript{x} was November 1996; a time when crops were being harvested and % of the cultivation area under fallow was increasing. The author's field notes did mention that some poultry manure was applied in October 1996, but Mr Saliba questioned this observation in a discussion that took place in September 1997. It would appear that fertiliser applications on the whole resulted in very high exports of N in runoff.

7.3 SUMMARY

1) There were large variations in runoff coefficient that are not accounted for by the total amount of rainfall. The crude water balance calculated in Chapter 6 indicates that the amount of available soil water (\textit{ASW}) influenced the amount of infiltration and thus runoff.

2) Total sediment, N and P losses in runoff from the farm were high; 240 t, 1600 kg, and 139 kg respectively. Soluble and particulate N losses were equivalent to 97 and 30 kg ha\textsuperscript{-1} yr\textsuperscript{-1} respectively, well above CMSS estimates. Soluble and particulate P losses were equivalent to 1 and 10 kg ha\textsuperscript{-1} yr\textsuperscript{-1} respectively in the upper range of CMSS estimates. Sediment loss was equivalent to 19 t ha\textsuperscript{-1} yr\textsuperscript{-1} or 2.5 mm of the soil profile.

3) The influence of soil cover on soil loss was not as obvious as shown in field-scale studies where land management was homogenous. The only variable that significantly influenced export of suspended solids was \( \sum I > 25 \).

The literature shows that rainfall intensity and energy are factors of rainfall erosivity. \( \sum I > 25 \) mm h\textsuperscript{-1} strongly influenced loss of sediment (R=0.87). Because of the relationship between rainfall intensity and energy, this implies that rainfall energy is important. Soil cover should therefore be critical because it can intercept and absorb rainfall kinetic energy. Linear regression showed weak correlations (\( R^2 \approx 0.3 \)) between \( I_{\text{max}} \), spatially averaged soil cover %, area of soil disturbance (SC2/3) and the concentrations of suspended sediment. The poor correlations reflect the complexity of the situation, particularly the high number
of variables and do not necessarily imply that these factors are unimportant. In fact there is strong evidence that soil disturbance (i.e. area of recently cultivated soil) strongly influences sediment loss and this is also supported by the literature.

4) N losses were highest after large applications of poultry manure. N is a more dynamic nutrient than P and, except for ammonium ($\text{NH}_4^+$), is not strongly attracted to (colloidal) soil particles. On the case study farm, N is applied in a range of forms, as poultry manure and a chemical fertiliser (ammonium nitrate). The forms and concentrations of N in runoff were highly variable. The concentrations of total N were very high and were linked to fertiliser applications.

5) P loss was predominantly due to erosion of particulate P. Therefore the key to minimising losses of P is to focus on erosion control, which could be improved on the case study farm as evidenced by the high figures for sediment loss.

6) There was, however, a general increase in the concentration of soluble P over the course of the monitoring period, corresponding with an increase in available soil P from 174 to 304 mg kg$^{-1}$ and a concomitant decrease in P sorption. The relationship suggested between soil P and soluble P in runoff is consistent with the literature. Comparison with soluble P concentrations in runoff from the Borg farm which has a longer history as a market garden, suggest that the trend of increasing soluble P concentrations is likely to continue if high fertiliser inputs are continued. The concentration of soluble P was considerably higher from the Borg farm, which has been a market garden for about 25 years. The observed rise in soluble P raises serious concerns about the impacts of market gardens with a long history of large applications of fertiliser P.

7) The influence of the stockpiled poultry manure on the measured losses of N and P is minimised by the fact that it was largely placed just outside of the catchment. The haphazard way in which it is handled, however, leaves some uncertainty about whether it contributed to spikes in N losses.
8. RESULTS AND DISCUSSION 3: COMPARISON OF LAND MANAGEMENT WITH A RAINFALL SIMULATOR

8.1 INTRODUCTION

8.1.1 Background

Market gardens are rarely, if ever, fully covered by vegetable crops, ie., there is always a percentage of the cultivation area under fallow. Yet, the practice of growing cover/green manure crops is not widely used in the market gardening industry, despite the potential short and long term benefits. The potential short term benefits include: 1) interception of rainfall, particularly in the pre-cultivation phase; and 2) improvement of soil hydrological properties.

Two contrasting summer fallow land management practices were simultaneously employed on the case study farm, bare fallow and sorghum 'green manure'. The green manure crop was planted to a small proportion of the farm and therefore, the farm-scale monitoring was unable to determine the significance of this practice on runoff.

Rainfall simulators can be used to make relatively quick comparisons of runoff from different land management practices, for both research and extension (Hudson, 1993).

8.1.2 Rationale and aims

In this Chapter, the effect of the sorghum green manure on soil hydrological properties is examined. This was not possible with the farm-scale monitoring because the green manure crop occupied a relatively small area of the farm. Used on the case study farm, the rainfall
simulator would also provide educational benefits to Mr Saliba, as rainfall simulators have been used elsewhere as action learning tools.

The aim was to examine the effect of 2 summer fallow land management practices on runoff volume: 1) bare, and 2) sorghum green manure, before and after cultivation. A secondary aim was to make extension observations.

8.1.3 The farmer’s rationale for growing sorghum

Because lettuce is susceptible to disease in warm conditions, it was not grown on the case study farm over summer. As a result, most of the cultivation area was left in bare fallow during summer when the average monthly rainfall is highest.

For about 5 years up to and including the 1995/96 summer, Mr Saliba grew oats as a green manure crop on small areas of the farm. This still left most of the cultivation area bare. Mr Saliba was aware that his farming practices reduce soil organic matter. The reason for growing green manure crops was to “put organic matter back into the soil”. In the 1996/97 summer Mr Saliba planted sorghum so that it could be ploughed-in early in 1997 as green manure. The reason Mr Saliba chose sorghum was that it would “produce a large amount of coarse organic matter and it would take longer to break down than oats.”

The area planted to sorghum was approximately 1.5 ha, about twice the area of oats grown the previous summer. Although still only a small percentage of the cultivation area, the increase in the area allocated for growing a green manure crop indicated that Mr Saliba’s perception of the need for this practice had changed over recent years.

8.1.4 Considerations for using rainfall simulators in an on-farm experiment

Design considerations

It is usually desirable that a rainfall simulator produces artificial rainfall that is similar to natural rainfall. Rainfall parameters that are most important in rainfall simulation include:
- **drop size**: varies to a maximum of 7 mm diameter. Drop size sharply increases with rainfall intensity from 0 to 40 mm h\(^{-1}\) and peaks at about 75 mm h\(^{-1}\). As intensity increases beyond 75 mm h\(^{-1}\), there is a slight tapering off of the median drop size. The median drop diameter varies between 2 and 3 mm (Bubenz, 1979a; Hudson, 1993).
- **fall velocity**: velocity of natural rainfall is determined by drop size (Kinnell, 1987).
- **rainfall intensity**: is the rate of rainfall and is usually measured as mm h\(^{-1}\). It can vary rapidly in natural rainfall, between and within storms (Hudson, 1993).
- **kinetic energy**: is the sum of kinetic energy of individual drops. It is a function of fall height and drop size. Rainfall kinetic energy is measured as J m\(^{-2}\) mm\(^{-1}\). It is an important parameter as it is closely linked to rainfall erosivity (Hignett, 1991; Hudson, 1993). The kinetic energy of rainfall is closely linked to intensity and is known to peak at about 75 mm h\(^{-1}\) (Hudson, 1993; Kinnell, 1987).

Consideration needs to be given to rainfall simulator design if the intended use is field demonstration, including:
- **plot size**: in the literature there is considerable reluctance to use small rainfall simulators (eg with plots less than 50 m\(^2\)) for measuring nutrient loss and extrapolating the results to larger areas (Hudson, 1993; Sumner et al., 1996). The basis of this argument is that the processes controlling sediment detachment and transport, are not adequately represented on small plots (Loch & Donnollan, 1983a). However, a study by Edwards et al. (1996) showed that on row cropped land treated with poultry manure, the composition of runoff reached equilibrium within 3 m (3 m was the smallest length examined) downslope.
- **water supply**: is a major limitation for using simulators in the field because it usually has to be carted.
- **mobility and cost effectiveness**: are essential characteristics. Cost considerations include building/purchasing and the number and expertise of staff required for operation.
- **power supply**: often mains power is not available in field situations. If the simulator requires electricity, a generator will be required.

One of the main advantages of using artificial rainfall over natural rainfall is that the infrequent, high intensity storms can be produced at will. The most severe erosion occurs as a result of high intensity rainfall. For this reason, the emphasis in erosion research has been on
development and use of simulators that produce high intensity rain. For example, Flanagan and Foster (1989) used a range of intensities from 64 to 250 mm h\(^{-1}\).

**Simulator designs**

There are a number of simulator designs ranging in complexity. Their suitability depends on the nature of the work. Simulator designs can be broadly categorised into 2 groups; low pressure drippers and using sprays.

**Drippers**

In drippers water is gravity fed to a grid module of tubes or needles. The size of the tube or needle orifice determines the size of the drop. The velocity of the drops is determined by their size and the height of the module above the ground. A fall height of between 5 and 12 m is needed to deliver drops at or near terminal velocity (Hudson, 1993; Hignett et al., 1995). Hence, unless the module is situated well off the ground, these simulators will produce low energy rain.

**Sprays**

This category includes the most simple forms of rainfall simulation: a watering can, a rose attached to a hose, or an inverted reciprocating garden sprinkler. These may be suitable for some demonstration purposes. A number of more complicated simulator designs use spray generated by high pressure. There are several commercially available spray nozzles with varying drop characteristics. Some of these nozzles produce very high intensity rain and a fault of many spray type simulators is that to produce a lower average intensity, some means is required to disperse the spray over a larger area (Bubenzer, 1979b). In simulators based on the 'rainulator' (eg. Loch & Foley, 1994) the 'veejet' nozzles (produce a flat 'v' shaped spray) are mounted on a shaft which is pointed downwards and rotates back and forth across the plot. The result is that at any point within the plot, intensity is highly intermittent. Grierson and Oades (1997) reduced the intensity of a 'fulljet' nozzle (produces a round 'v' shaped spray) by using a segmented disc rotated below the nozzle. Shelton, von Bernuth and Rajbhandari (1985) reduced the need for intermittent application by injecting air into the water conduit,
thereby reducing the intensity produced by the nozzle. On some simulators the nozzles are pointed upwards, utilising gravity to simulate natural rainfall.

**Rainfall simulators in extension**

Rainfall simulators provide a means of comparing relative erodibility of various soil management practices (Hudson, 1993). In Australia, some use has been made of rainfall simulators in extension.

The Queensland Department of Natural Resources (QDNR) used a rainfall simulator in a farmer education project, to improve farming systems to optimise rainfall capture and minimise soil erosion in the wheat belt. The simulator was used to compare soil management practices *in situ*, with active farmer participation and input into the demonstration process (D. Freebairn, pers. comm., 1997). Importantly, the extension staff played a passive role in the demonstrations, allowing farmers to make their own interpretation. The demonstrations influenced the attitudes of participating farmers. A survey found that 90 % of the farmers who attended felt that the rainfall simulation would influence their farm management decisions. Ninetynine % of the farmers stated that the rainfall simulator was an effective tool for demonstrating runoff/infiltration/erosion (Cawley *et al.*, 1992). This project has attracted considerable interest from extension practitioners. Hamilton (1997) concluded that the rainfall simulator is an effective participatory learning and action research tool.

### 8.2 Materials and Methods

At the outset of this research the rainfall simulator was designed for use on flat ground rather than raised beds. Thus some modifications were required to overcome before it could be successfully used on the case study farm.

#### 8.2.1 Rainfall simulator

The simulator used is described by Loch and Foley (1994). It was fitted with 3 Veejet 80100 nozzles that produce artificial rainfall with kinetic energy of 29.5 J m² mm⁻¹ (Loch & Foley,
1994). A schematic diagram of the rainfall simulator, as it was used in this research, is shown in Figure 8-1.

![Schematic diagram of the rainfall simulator and collection system](image)

**Figure 8-1. Schematic diagram of the rainfall simulator and collection system**

### 8.2.2 Modifications of the rainfall simulator

Some problems were noted with using the simulator on a market garden after using it during a field day. The criticisms included:

1) the runoff plot frame was only 0.75m wide and had to be placed on top of the vegetable bed. This excluded the furrows which are an important hydrological feature of most market gardens;

2) a rainfall event of 40 mm in 30 minutes has a return period of just under 2 years at Richmond. The dramatic appearance of heavy rain washing away copious amounts of soil, understandably seemed to make the market gardeners get a little defensive; and

3) the freshly cultivated bed was not compared with a reduced tillage or no tillage bed, and therefore there was nothing to compare the results with and no great positive message.

In response to the first problem, a larger plot frame was constructed for use on vegetable beds with furrows spaced approximately 1.5m apart. Measuring 2m x 1.5m it could be placed laterally across 2 beds with the collection trough located in the furrow.
It was also apparent that the 'rainulator' style simulator was designed for erosion research in a tropical or subtropical climate. The 80100 nozzle produces high energy, high intensity rain similar to that of tropical thunderstorms. The intensity can be varied by adjusting the dwell time (when the spray is directed to the outside of the plot into collection troughs). While the total kinetic energy of rain could be reduced in this manner, kinetic energy per unit of rain would remain at 29.5 J m$^2$ mm$^{-1}$, which is only typical of natural rain at very high intensities, which occur infrequently (Kinnell, 1987). Another disadvantage of increasing the dwell time is the highly intermittent and unnatural pattern of rain.

For this reason, 3 sets of Veejet nozzles were purchased which produce lower intensity and energy sprays; the 8030, 8050 and 8070. The second 2 digits of the nozzle code (ie. '30', '50', and '70') refers to the size of the nozzle aperture, and indicates roughly the percentage of 'rainfall' produced, compared to the 80100. Drop sizes also decrease with smaller nozzle apertures and therefore, the kinetic energy decreases similarly. Except for the 8070 nozzle, data on rainfall kinetic energy produced from the 8050 and 8030 nozzles are not available. To determine the kinetic energy, each set of nozzles purchased would have to be tested to measure drop size and velocity, so that the kinetic energy per mm of rain could be calculated. If a direct comparison is to be made with the energy of natural rain, kinetic energy values for the nozzles would be necessary. This was not done in this study.

The rainfall simulator was originally fitted with spray nozzles that produced kinetic energy similar to that of a very heavy thunderstorm. Hignett (1991) suggests that lower energy rain is more useful for producing different runoff responses from different land management practices. Because the intensity and kinetic energy of natural rain is highly variable and kinetic energy greatly influences surface sealing (Hignett et al., 1995) it was considered desirable to produce a range of intensities and energies in the simulated rainfall that would represent local natural rainfall. For this reason a number of sets of spray nozzles with different aperture sizes were purchased.

The 8030, 8050, 8070 and 80100 nozzles were calibrated at UWSH to determine their intensities using a dwell time setting of 2.2 seconds. This was done by covering the plot with
a plastic sheet (so there was no infiltration) and measuring runoff. The rainfall intensities are listed in Table 8-1. A visual assessment was made of the spray uniformity of each nozzle at various pressures. Higher pressure was needed for the smaller nozzles to achieve coverage on the plot edges.

<table>
<thead>
<tr>
<th>Nozzle</th>
<th>Pressure [kPa]</th>
<th>Intensity [mm h⁻¹]</th>
<th>Energy [J mm⁻¹ m⁻²]</th>
</tr>
</thead>
<tbody>
<tr>
<td>8030</td>
<td>90</td>
<td>17</td>
<td>N/A</td>
</tr>
<tr>
<td>8050</td>
<td>80</td>
<td>46</td>
<td>N/A</td>
</tr>
<tr>
<td>8070</td>
<td>70</td>
<td>62</td>
<td>19.7⁺</td>
</tr>
<tr>
<td>80100</td>
<td>60</td>
<td>80</td>
<td>29.5⁺</td>
</tr>
</tbody>
</table>

⁺⁺ Loch and Foley (1994).

It was originally intended to use 3 sizes of 'Veejet' nozzles in this experiment. The 8030 nozzle proved problematic in light winds (it delivered a very fine spray) and subsequently was rejected for this experiment. During the calibration experiment, light winds produced unacceptable variability in the rainfall intensity, while the 8050 and 8070 nozzles produced almost linear rainfall intensities in similar conditions (Appendix). This was a major limitation for the 8030 nozzle, as even light winds would make it difficult to interpret results.

Due to time limitations in the field and to the difficulty of collecting rainfall over the plot without wetting the soil, calibrations were not done in the field. To compensate, great care was taken when setting up the simulator for each run in the field to ensure that the spray nozzles were positioned at the same height above the plot, the primary cause of variation in the application rate. The other causes of variation are pressure, which was very carefully adjusted and monitored during each run, and wind. Prevention of pressure variations during each run was assisted by cleaning the water filter before each run and by maintaining a constant water level in the water reservoir.

### 8.2.3 Measurement of runoff

Runoff was collected from the plot in the collection tube. Water level in the tube was recorded at 30 second intervals with a capacitance water level sensor and calibrated to depth (mm) over
the plot. A cumulative runoff chart was generated simultaneously on a laptop computer. This allowed infiltration to be calculated, assuming the calibrated intensity.

8.2.4 Rainfall simulation runs made on bare fallow and sorghum cover/green manure crop

The rainfall simulation runs and treatments are outlined in Table 8-2. Runoff volume was measured on the bare and green manure crop in 2 stages: as a mature cover crop (pre-incorporation), and as green manure (post-incorporation) at 2 rainfall intensities. Two rainfall simulations were run on each treatment, one with the 8070 nozzle (≈62 mm h⁻¹) and one with the 8050 nozzle (≈46 mm h⁻¹) so in all 8 runs were completed.

<table>
<thead>
<tr>
<th>Comparison</th>
<th>Treatment</th>
<th># Run</th>
<th>Soil water assessment</th>
<th>Treatments compared</th>
<th>Treatment</th>
<th># Run</th>
<th>Antecedent soil water assessment</th>
<th>Intensity [mm h⁻¹]</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>bare</td>
<td>1</td>
<td>qualitative⁴</td>
<td>mature sorghum</td>
<td>qualitative⁴</td>
<td>2</td>
<td></td>
<td>62</td>
</tr>
<tr>
<td>2</td>
<td>bare</td>
<td>3</td>
<td>&quot;</td>
<td>mature sorghum</td>
<td>&quot;</td>
<td>4</td>
<td></td>
<td>46</td>
</tr>
<tr>
<td>3</td>
<td>bare</td>
<td>5</td>
<td>quantitative⁵</td>
<td>incorporated sorghum</td>
<td>&quot;</td>
<td>6</td>
<td>quantitative⁵</td>
<td>46</td>
</tr>
<tr>
<td>4</td>
<td>bare</td>
<td>7</td>
<td>&quot;</td>
<td>incorporated sorghum</td>
<td>&quot;</td>
<td>8</td>
<td></td>
<td>62</td>
</tr>
</tbody>
</table>

⁴ Soil water status (McDonald et al., 1984).
⁵ Gravimetric analysis 0-5 cm, 5-15 cm, and 15-30 cm.

Description of the sorghum and bare fallow ‘treatments’

Both ‘treatments’ were under the farmer’s full control and were not directly influenced by the author. Until November, 1996 both treatments had a similar short term management history. In the 1995/6 summer, both had a crop of oats incorporated as green manure. The incorporated oats were left fallow until June, 1996. Two crops of lettuce were grown between June and October. From this juncture, the management differed greatly.
Bare fallow

The bare fallow beds were rotary hoed in mid-November, 1996 and the left until January, 1997, when poultry manure was spread and incorporated, and the beds were reformed (Plate 7).

Sorghum

In mid-October sorghum was sown at 30 kg ha\(^{-1}\), into beds (rotary hoed to 5 cm) that were covered by dead and partly intact weeds. Mr Saliba was dissatisfied with the sorghum because of germination of too much thistle, so he sprayed the young sorghum and thistle seedlings with a contact herbicide. In early November, he re-sowed sorghum. The sorghum was sown in 3 lines per bed and irrigated sparingly. Rainfall was frequent enough in the first half of the life of the sorghum to avoid any need for irrigation. By January, 1997, conditions became drier and Mr Saliba attempted to irrigate the sorghum. By that stage the sorghum was taller than the sprinklers, so only the beds closest to the sprinkler lines received any water. As a result the height of the mature sorghum varied between 1.5 - 2.5 m, depending on the position of beds in relation to the lines.

Poultry manure was applied to the sorghum treatment and incorporated during the process of incorporating the sorghum into the soil (between comparisons 2 and 3), using the same tillage procedure as for the bare fallow treatment.

The rainfall simulations were carried out with minimal disturbance to the soil surface. The mature sorghum was 2 to 3 metres high, so the stalks were trimmed to about 10 cm above ground level. The effect of raindrop interception by the mature sorghum was not tested (because the focus was on soil hydrological properties). This could not be adequately done without a separate experiment and in any case sorghum is not particularly suited to being used soley as a cover crop. In a tall plant such as sorghum where the leaves cause drops to coalesce and fall from height, the impact of these drops on the soil surface can lead to particle detachment and transport (Moss, 1989).
Plate 7. Bare fallow beds with the rainfall simulator in place

Plate 8. The sorghum beds as they were prepared for the rainfall simulation (note the cracks in the furrows between sorghum beds, prior to the January, 1997 event)

Chapter 8: Comparison of Land Management with a Rainfall Simulator
When the first rainfall simulation comparison was done, neither treatment had been irrigated for several weeks and there had not been rainfall since 13.7 mm fell between 17-18/12/96. By the time the first rainfall simulations were done, cracks had appeared in the furrows between the beds of sorghum (Plate 8), but were not present in bare fallow furrows.

8.2.5 Sample collection and handling

Samples were collected by temporarily diverting runoff into a container through a series of taps, to a sealed perspex container. Samples were taken at:

1) 5 minutes after runoff had commenced, and
2) 25 minutes after the rainfall simulation commenced.

Samples were then decanted into 500 ml bottles and labelled and frozen within 6 hours of collection. Analysis was performed by CSIRO using the same methods as described in Chapter 5.

8.2.6 Approach to using the simulator on the farm while maintaining a good rapport with the farmer

High priority was given to maintaining a positive relationship with the case study farmers, hence the experimental procedure was influenced by the needs of the farmer to access his production area. The rainfall simulator and its associated equipment take up a surprisingly large amount of space. It was impossible to use without causing some degree of inconvenience to the farmers. For this reason, the amount of time allocated to using the simulator on the farm was minimised.

The advantage of conducting the experiment in situ was that it allowed interaction with the farmer, and thus the results were readily available and potentially of benefit to the farmer. Thus, the intention was to involve Mr and Mrs Saliba in the rainfall simulation work wherever possible. A flexible approach was taken in terms of when and how they participated. The Salibas were encouraged to "have a look and to ask any questions any time" during the rainfall simulation work.
The Saliba's were obviously curious about the rainfall simulations, particularly Mr Saliba who stopped to talk on a number of occasions. I spent a few minutes explaining how the simulator worked, the measurements I was taking and why I was taking them. This had also been discussed in the weeks leading up to the experiment when I was seeking their approval to carry out the work on their farm, so they were well informed. Notes were made at the end of the day, on any interaction with the farmers.

**Additional soil analysis**

Soil moisture was not measured for the first 2 comparisons. There was an obvious difference in soil moisture during the first comparison evidenced by cracks in the furrows of the mature sorghum. This was due to water uptake and evapotranspiration from the sorghum. After the sorghum was incorporated, the green manure beds had been irrigated, creating 'artificial' discrepancies in soil moisture. For this reason soil samples were taken prior to each run and gravimetrically analysed for moisture.

In May, 1997, soil samples were taken from the beds from the half of the cultivation area which had included the bare and sorghum beds, as well as other bare fallow beds not included in the rainfall simulation experiment. Three composite samples were analysed:

1) bare fallow, where oats had been incorporated in the previous summer;
2) sorghum green manure; and
3) bare fallow, with no recently incorporated green manure

The composite samples were analysed for organic carbon, using the Walkley-Black method, in triplicate.

**8.3 RESULTS**

**8.3.1 Runoff from mature sorghum, green manure sorghum and bare fallow**

Comparisons 1 and 2 (between the mature, trimmed sorghum and bare fallow beds) are shown in Figures 8-2 and 8-3. In both cases runoff was at least 50 % greater than from bare fallow. Comparisons 3 and 4 (between sorghum green manure and bare fallow) are shown in Figures 8-4 and 8-5.
Figure 8-2. Comparison 1: bare fallow and trimmed, mature sorghum at $I = 62$, with dry soil water status\(^a\)

\(^a\) Qualitative field classification from McDonald et al. (1984).

Figure 8-3. Comparison 2: bare fallow and trimmed, mature sorghum at $I = 46$, with moderately moist soil water status\(^b\)

\(^b\) Qualitative field classification from McDonald et al. (1984).
Figure 8.4. Comparison 3: Runoff from bare fallow and incorporated sorghum at $I = 46$ (Antecedent soil water content for bare fallow and sorghum treatments was 11.1 % and 12.2 % respectively.)

Figure 8.5. Comparison 4: Runoff from bare fallow and incorporated sorghum at $I = 62$ (Antecedent soil water content for bare fallow and sorghum treatments was 8.1 % and 8.9 % respectively.)
Calculated ‘steady-state’ infiltration rates

Table 8-3 shows the calculated ‘steady-state’ infiltration rates for each of the rainfall simulation runs, i.e. when runoff is linear. For the purpose of this exercise, steady-state was calculated from 15-25 minutes. This provides an indication of infiltration differences, after surface depressions and cracks were saturated and any surface seal had formed.

Table 8-3. Infiltration rates calculated from 15-20 minutes

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Run</th>
<th>Calculated infiltration rate [mm h⁻¹]</th>
<th>Rainfall intensity (I) [mm h⁻¹]</th>
<th>Soil cover[%]</th>
<th>Antecedent soil water</th>
<th>Soil condition *</th>
</tr>
</thead>
<tbody>
<tr>
<td>bare</td>
<td>3</td>
<td>0</td>
<td>46</td>
<td>0</td>
<td>moist</td>
<td>SC3</td>
</tr>
<tr>
<td>bare</td>
<td>5</td>
<td>0</td>
<td>46</td>
<td>0</td>
<td>11.1 %</td>
<td>SC1</td>
</tr>
<tr>
<td>bare</td>
<td>7</td>
<td>0</td>
<td>62</td>
<td>0</td>
<td>8.1 %</td>
<td>SC1</td>
</tr>
<tr>
<td>bare</td>
<td>1</td>
<td>8</td>
<td>62</td>
<td>0</td>
<td>dry</td>
<td>SC2</td>
</tr>
<tr>
<td>sorghum</td>
<td>4</td>
<td>13</td>
<td>46</td>
<td>3</td>
<td>moist</td>
<td>SC1</td>
</tr>
<tr>
<td>sorghum</td>
<td>2</td>
<td>17</td>
<td>62</td>
<td>5</td>
<td>dry</td>
<td>SC1</td>
</tr>
<tr>
<td>green manure</td>
<td>6</td>
<td>22</td>
<td>46</td>
<td>20-25</td>
<td>12.2 %</td>
<td>SC3</td>
</tr>
<tr>
<td>green manure</td>
<td>8</td>
<td>26</td>
<td>62</td>
<td>20-25</td>
<td>8.9 %</td>
<td>SC2</td>
</tr>
</tbody>
</table>

* SC3 = loose surface, SC1 = settled surface, SC2 = partially settled/loose (see Chapter 5)

Infiltration rate increased in the order of bare fallow < sorghum < green manure/sorghum stubble.

The difference in runoff in comparison 1 (run 1 vs run 2, Figure 8-2) was due to 2 factors:

- earlier runoff on the bare than on the sorghum (after 4 minutes) was due to surface depression storage on the sorghum beds and small cracks in the furrow which delayed runoff; and
- a higher rate of runoff from the bare than from the sorghum, or conversely lower ‘steady-state’ infiltration rate (17 mm h⁻¹ for sorghum and 8 mm h⁻¹ for bare fallow).

In comparison 2 (run 3 vs run 4, Figure 8-3), the calculated infiltration rate from 15-25 minutes was actually 0 mm h⁻¹ for the bare fallow and 13 mm h⁻¹ for the sorghum. The lower infiltration rates in comparison 2 reflect a wetter soil profile and therefore lower available pore...
space. Comparison 2 was made after the January, 1997 runoff event, whereas comparison 1 was made before the runoff event and after a dry period.

In Figure 8-4 runoff from the sorghum commenced at the same time as from bare fallow and the difference in total runoff after 25 minutes was due solely to the difference in steady-state infiltration rate.

In comparisons 3 and 4 the soil water was similar between treatments, but higher in the green manure beds (due to irrigation). At both intensities (in Figure 8-4, \( I = 46 \); and in Figure 8-5, \( I = 62 \)) runoff occurred first on the bare fallow. This indicates that not only was runoff delayed on the incorporated green manure stubble, but infiltration was also higher than on bare fallow.

### 8.3.2 Soil organic carbon

The results of the analyses of surface soil for organic carbon are shown in Table 8-3. The results are considerably higher than the results of the soil survey performed by DLWC, which measured organic carbon levels of between 0.8 and 1.0 %, also using the Walkley-Black method. Differences may be attributed to the fact that analyses were performed in different laboratories by different personnel and also to the fact that samples were collected from the opposite side of the farm.

<table>
<thead>
<tr>
<th></th>
<th>1995/96 summer</th>
<th>1996/97 summer</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bare fallow</td>
<td>Oats incorporated</td>
</tr>
<tr>
<td>Bare fallow</td>
<td>Bare fallow</td>
<td>Sorghum incorporated</td>
</tr>
<tr>
<td>Organic carbon [%]</td>
<td>1.8</td>
<td>2.3</td>
</tr>
</tbody>
</table>

(Walkley-Black method)

The concentration of organic carbon in the beds used for the rainfall simulations was slightly higher in the beds with incorporated sorghum. The organic carbon levels were lowest in the beds which were left as bare fallow for 2 consecutive summers.
8.3.3 Sediment concentrations

The concentration of sediment from each of the treatments at different stages through the rainfall simulation runs is shown in Table 8-5.

Table 8-5. Sediment concentrations from bare, sorghum and green manure with soil condition and rain intensity

<table>
<thead>
<tr>
<th>Treatment</th>
<th>Surface soil condition</th>
<th>Intensity (I) [mm h⁻¹]</th>
<th>Sediment concentration [mg L⁻¹]</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>5 min after runoff</td>
</tr>
<tr>
<td>Bare fallow</td>
<td>SC3</td>
<td>46</td>
<td>20 926</td>
</tr>
<tr>
<td>Bare fallow</td>
<td>SC2</td>
<td>62</td>
<td>13 325</td>
</tr>
<tr>
<td>Bare fallow</td>
<td>SC1</td>
<td>62</td>
<td>8 851</td>
</tr>
<tr>
<td>Bare fallow</td>
<td>SC1</td>
<td>46</td>
<td>6 371</td>
</tr>
<tr>
<td>Sorghum - intact</td>
<td>SC1</td>
<td>62</td>
<td>5 710</td>
</tr>
<tr>
<td>Sorghum - intact</td>
<td>SC1</td>
<td>46</td>
<td>2 204</td>
</tr>
<tr>
<td>Green manure</td>
<td>SC2</td>
<td>62</td>
<td>5 902</td>
</tr>
<tr>
<td>Green manure</td>
<td>SC3</td>
<td>46</td>
<td>14 623</td>
</tr>
</tbody>
</table>

* SC3 = loose surface, SC1 = settled surface, SC2 = partially settled/loose (see Chapter 5)

The highest sediment concentrations occurred from bare fallow with loose surface soil (SC3). Even on the green manure, loose surface soil resulted in the second highest concentrations. Generally where all other conditions were the same, sediment concentrations decreased from SC3 to SC1. Concentrations were also lower for I=46 than I=62. The concentration of sediment after 5 minutes of runoff from the green manure at SC2 and I=62 was less than half the concentration from bare fallow under the same conditions, but by 25 minutes, the concentrations were similar.

8.4 DISCUSSION

8.4.1 Evidence of surface sealing

There is value in looking at what evidence there is or is not for surface sealing, because it is an issue relevant to land management that was unresolved in Chapter 7. Surface sealing poses a potential problem to the market gardener and also with respect to the offsite impacts of market
gardening. The major effect of surface sealing is that it dramatically reduces infiltration and increases runoff. Where surface sealing is caused by raindrop impact, the amount and structure (i.e. the form) of vegetation becomes critical for minimising this deleterious process because of its role in absorbing rainfall energy (Moss, 1989).

One approach to measuring surface sealing is to use infiltration as an indicator, as done by Stern et al. (1991). Wace and Hignett (1991) reported that the infiltration rate after 50 mm of rain was a useful measure of structural degradation, although this related to laboratory studies where soil was completely disturbed to simulate tillage. It appears that surface sealing occurred very quickly even on a loose soil surface (Table 8-3), whereas the farm-scale monitoring indicated that infiltration was higher when the surface seal was disrupted. This anomaly could be due to variation in soil qualities across the farm or differences in rainfall energy and intensity between the rainfall simulator and natural rainfall. The calculated ‘steady state’ infiltration rates for run 2 and run 3 with surface condition of SC2 and SC3 respectively, were 8 mm h\(^{-1}\) and 0 mm h\(^{-1}\). The differences in the calculated ‘steady state’ infiltration rates between the bare and green manure treatments are most likely due to the degree of surface sealing, because the green manure treatment had sorghum residue to absorb much of the energy of rainfall.

Differences in steady-state infiltration between treatments was probably due at least partly to soil cover provided by intact sorghum stalks and the exposed stubble after the sorghum was incorporated as green manure. This cover would have reduced raindrop impact at the soil surface, thus reducing surface seal development.

**8.4.2 Evaluation of sorghum as a conservation measure**

Bare soil is highly erodible. Plant cover greatly minimises the erosive impact of raindrops by intercepting and absorbing kinetic energy of raindrops (Moss, 1989). It is the kinetic energy of rain which often causes surface sealing and reduced infiltration (Wace & Hignett, 1991).Plant material that is in contact with the soil also acts to slow runoff, thereby reducing its erosive power (Moss, 1989; Bradford & Huang, 1994). Farming practices which maximise plant cover result in less soil erosion. A good example is stubble retention. Freebairn and Wockner (1986) found that stubble retention resulted in soil movement equivalent to 5 t ha\(^{-1}\) year\(^{-1}\).
compared to soil movement of 29-62 t ha\(^{-1}\) year\(^{-1}\) from farming systems where stubble was removed.

The long term benefits stem from increasing the level of organic matter in the soil, thereby improving structure and permeability. It is a widely held view that organic C (OC) content strongly influences soil structure. Guerra (1994) and Watts and Dexter (1997) reported that OC correlates with aggregate stability. According to McTainsh (1993) humus, which is derived from organic matter, combines with clays to form a clay-humus complex.

Improving soil structure by building up the level of organic matter is generally regarded as a long term process. There are also short term improvements in soil structure. Fine roots and root hairs physically hold aggregates together. The effect of this should be in an immediate but short term improvement in soil structure. The soil organic carbon results show there has been a short to medium term increase in organic matter that may partly explain higher infiltration rates on the mature and incorporated sorghum. However, the long term improvements in soil structure would be limited by continued tillage, which would accelerate oxidation of organic matter (Rose, 1993).

After the February, 1997 runoff event when 164.6 mm of rain fell, deposition of sediment in the furrows behind the exposed sorghum residue had occurred (Plate 9). The residue had slowed the flow of water in the furrows to extent where a large amount of eroded sediment had been deposited before entering the drainage channel and potentially Currency Creek.

The reasons for reduced runoff in the pre-incorporation phase are likely to include:

1) the sorghum stalks and dead leaves provided some soil cover, thus reducing surface sealing;
2) the sorghum dried out of the soil sufficiently to create cracks in the furrows which absorbed rainfall and delayed runoff; and
3) the fine roots and root hairs physically improved aggregate strength and increased drainage and reduced aggregate breakdown.

The reasons for reduced runoff in the post-incorporation phase are likely to include:
1) residue remaining on the soil surface reduced surface sealing and maintained macropores (thereby increasing infiltration) and reduced particle detachment and transport as well as slowing runoff and trapping some eroded sediment; and
2) after incorporation it added organic matter, improving soil structure.

Plate 9. Sorghum residue provided considerable soil conservation value during the February, 1997 runoff event

There are disincentives for using cover/green manure crops. otherwise the Salibas might presumably apply the practice to the whole cultivation area over summer. The practice requires:

1) Longer fallow period. As time is needed for the green manure crop to grow, and time is then needed before the green manure has broken down sufficiently to plant a vegetable crop into. In the case of sorghum, the beds were not used for growing vegetables for a period of 6 months.
2) Longer term planning. The farmer has to plan ahead so that sufficient land is available for growing income earning crops when he intends to grow them. Ideally, if all of the
cultivation area cannot be covered by a green manure crop, the areas should be rotated each year so that soil structure is maintained on all of the cultivation area.

3) Expenditure with no immediate monetary gain. The costs include: purchase of seed, labour and fuel costs of preparing beds, herbicide, and possibly irrigation.

8.4.3 Extension with the simulator

Simulator design

The simultaneous logging and graphing of runoff on the laptop screen was definitely an advantage for assessing and relaying the results to the farmer. The results were presentable while the simulator was operating. In an ideal situation 2 plots would be tested simultaneously, but there is no easy way around this problem without sacrificing authenticity of the demonstration.

Both the 8070 and 8050 nozzles produced significant amounts of runoff. The main issue in deciding which to use is which one (if any) will better highlight differences in runoff quantity and quality. It would appear that the comparisons done at 46 mm h\(^{-1}\) (with the 8050 nozzles) produced greater differences in runoff between the treatments than at 62 mm h\(^{-1}\) (with the 8070 nozzles). This supports the theory that rainfall energy is an important factor when trying to differentiate the susceptibility of different soils to surface sealing proposed by Wace and Hignett (1991). For this reason, in extension work it is probably best to use nozzles such as the 8050 which produce finer drops (and therefore lower energy) than the standard 80100. The 8050 will provide greater contrast between different soil management practices.

A cheap alternative to using the rainfall simulator is to encourage the farm to trial different management practices on a small number of beds and then to simply compare runoff from irrigation.

Mr Saliba's reaction to the rainfall simulations

Mr Saliba was very interested when told of the contrasting results between the 2 plots. It provided a good point of discussion. We talked about why more of the rain infiltrated into the
soil on the trimmed sorghum. After mentioning the effect of the cracks in the furrows between the sorghum beds in allowing water to enter the soil, Mr Saliba said that he thought the root system of sorghum would reach well below the maximum depth at which his rotary hoe could till the soil and as these roots decay (after the sorghum is ploughed in) subsequent crops would benefit by allowing the root systems of the crop and water, to reach deeper into the soil. He said that this would mean that he would be able to irrigate less for the crop to receive the same amount of water. Mr Saliba said that the cracks in the sorghum furrows were not present in the bare fallow furrows.

8.4.4 Value of the rainfall simulations an extension exercise

The rainfall simulation experiment was a valuable rapport building exercise. Whether or not land management on the farm is influenced by this rainfall simulation work remains to be seen, but it certainly encouraged Mr Saliba to question issues of land management during the summer fallow period. He was able to see tangible results from the rainfall simulations and demonstrated that he was making plausible interpretations that could well influence farm planning in the future. We had conversations about the rainfall simulation experiment weeks after the field work was completed. For example, I gave the him some printouts of the runoff graphs and explained the results to them. I was then able to expand our conversations to related issues of soil erosion and nutrient loss. Mr Saliba mentioned that he had tried legume green manure crops a few years ago, but did not was not impressed. He could not explain why.

I believe this work made an impression on Mr Saliba as he began to see the green manure crop not only as a means to an end: increased organic matter and a more workable soil, but also, the value of sorghum in minimising soil erosion, both as a standing crop and the remaining surface residue after it was incorporated into the soil. Similarly, I the researcher, gained a better understanding of the farmer’s perspective.
8.5 SUMMARY

The results indicated clearly that the practice of growing a sorghum crop for green manure provided immediate benefits by reducing the volume of runoff. The reasons for reduced runoff in the pre and post-incorporation phases are likely to include:

1) the sorghum stalks and residue decreased surface sealing (particularly in the post-cultivation stage);

2) development and maintenance of cracks and macropores; and

3) increased aggregate strength and improved structure (particularly in the pre-cultivation stage).

Surface sealing occurred very quickly even on a loose soil surface.

The farmer was very interested in the rainfall simulation experiment and there is potential to use simulators as an extension tool on market gardens, but design limitations must first be overcome. Alternatively, for extension purposes, runoff could be measured from furrows after irrigation.
9. **FINAL DISCUSSION AND CONCLUSIONS**

9.1 **SEDIMENT, N AND P LOSSES - IMPLICATIONS FOR WATERWAYS**

Concentrations of N and P in runoff from the farm were several orders of magnitude higher than ANZECC guidelines for the protection of Australian freshwater streams (McMahon & Hart, 1996). The concentration of the soluble fraction of P alone was above the ANZECC guidelines which are for total P. The increase in soluble P concentrations observed during the study is most concerning because soluble P is more bioavailable, and more likely to cause algal blooms.

Placement of the monitoring equipment on the boundary of the case study farm, which was also the point of entry of runoff from the farm into Currency Creek, left little doubt that the measured losses accurately represented contributions from the case study farm to Currency Creek. Over 1200 kg of N and 138 kg of P entered Currency Creek from the case study farm. Export of 127 kg ha\(^{-1}\) yr\(^{-1}\) of N was higher than in any other known Australian study of market gardening. Export of 11 kg ha\(^{-1}\) yr\(^{-1}\) of P is at the upper end of estimates based on CMSS (Marston, 1994). Loss of soil from the farm was equivalent to 19 t ha\(^{-1}\).

High concentrations of suspended sediments are especially damaging in slow moving waters such as Currency Creek where smothering of the benthic community and the introduction of oxygen depleting substances is a serious problem. Generally, high suspended sediment concentrations can increase turbidity and decrease light penetration, clog gills and filters of fish and aquatic invertebrates and reduce fish spawning and fish survival (Dennison, 1996).

The case study farm is located several km upstream from the confluence of Currency Creek and the Hawkesbury River. Currency Creek is characterised by slow flows and as such is normally a depositional environment. However, the impact of runoff from the case study farm on the Hawkesbury River cannot be discounted on account of the distance that runoff travels before entering the River. The soluble and colloidal particle size fraction (<2 μm) of runoff

*Chapter Nine: Final Discussion and Conclusions*
from the case study farm has the potential to be transported by Currency Creek to the Hawkesbury River in relatively short time. In high flows, previously deposited material including coarser particles can be entrained and transported downstream. Thus sediment entering Currency Creek from the case study farm has the potential to adversely affect the Hawkesbury River.

Similarly, particulate P entering Currency Creek from the case study farm can change to more bioavailable P. Anoxia in the stream substrate as observed by Mann (1996) in constructed wetlands, could lead to desorption of inert particulate P to re-enter the water column as bioavailable soluble P.

9.2 LINKS BETWEEN MANAGEMENT AND STORMWATER LOSSES OF SEDIMENT, N AND P

9.2.1 The scale issue

The scale issue is particularly important where the objective is to examine the relationship between specific land management practices and runoff. In this study it was difficult to link specific practices with farm-scale runoff data unless the practice was widespread.

The survey of land management on the case study farm revealed that at any particular time land management was heterogenous and often contrasting, which is typical of market gardens in the Hawkesbury-Nepean catchment. For example, on one plot, tillage constituted one pass of the 1.0 m rotary hoe to 5-7 cm, while on an adjacent plot, beds were reformed after several runs of the 2.0 m rotary hoe to 25 cm. Similarly, during the 1996/97 summer, a green manure crop was grown on 2 relatively small plots while most of the farm was left in bare fallow and for much of the time, freshly tilled.

Farm-scale monitoring has a valuable role in assessing the success of whole farm planning and management. In this study, the farm-scale monitoring revealed variations in the quality and quantity of runoff leaving the farm which appear to have been linked with both the storage and application of poultry manure and chemical processes occurring in the drainage channel. The farm-scale monitoring thereby provided an indication of what processes and aspects of land

Chapter Nine: Final Discussion and Conclusions
management are worthy of closer assessment in order to achieve a well balanced understanding of whole farm management.

Farm-scale monitoring provided an accurate indication of export rates of sediment and nutrients from the land use. In this respect, the results have greater value than results from catchment monitoring studies, such as the Monkey Creek study, where market gardening was also monitored, but as a small unit within a 30 km² catchment (Cullen, 1991).

Land management practices were compared at plot-scale in the rainfall simulation experiment. While bare fallow was widespread across the farm over summer, green manure crops were planted to a relatively small proportion of the farm. The rainfall simulator experiment showed that large differences in the quantity of runoff resulted from alternative fallow practices. This could not have been concluded from the farm-scale runoff data. The simulator study also indicated that surface sealing occurred very quickly on cultivated soil, which would indicate that higher infiltration in the January, 1996 event than the May, 1996 event was not due to recent cultivation as indicated by the farm-scale monitoring.

An approach to monitoring and/or extension that would be suitable for use on market gardens, would be to use small flumes or tipping buckets to monitor runoff quantity and quality in furrows, as a substitute for plot work with the rainfall simulator. This would allow farmers and researchers to explore specific land management practices such as green manures. The limitation of this approach is that management of the remainder of the farm, ie. outside of the vegetable beds, would be ignored.

9.2.2 Fertiliser use

During this study fertiliser inputs on the case study farm were high, but not higher than on other market gardens in the Hawkesbury-Nepean catchment. In fact P inputs (182 kg ha⁻¹ yr⁻¹) were well below the 450 kg ha⁻¹ yr⁻¹ average on 28 market gardens as reported by Jinadasa et al. (1997). Fertiliser inputs were reduced on the case study farm from 1993 to 1995, prior to which they were similar to the values quoted by Jinadasa et al. (1997).
Runoff from the case study farm contained increasing concentrations of soluble P over the two year monitoring period. This was explained by excessive application of poultry manure, leading to a reduction in the P sorption capacity in the soil. Soil sampling on the case study farm at the beginning and end of the study as part of the NLP study, and by Jinadasa *et al.* (1997) show that total P in soils on the case study farm is several orders of magnitude higher than P background concentrations. Poultry manure was the only supply of P on the case study farm during the study. Bray P in the soil was high at the beginning of the study (174 mg kg$^{-1}$ in 0-20 cm of soil) but had soared to 304 mg kg$^{-1}$ by the end of the study. Comparison with runoff from a nearby but older market garden showed that the continued heavy use of poultry manure is likely to lead to a continued rise in the concentration of soluble P.

N losses were equivalent to 127 kg ha$^{-1}$ yr$^{-1}$. Of this, 85 % was in soluble form and 15 % was in particulate form. Therefore erosion prevention and control could be effective on only 15 % of total N. The key to reducing N loss lies in reducing application rates, improving the timing and method of application (Barrisas *et al.*, 1978; Baird, 1996). Storage of poultry manure (on the case study farm poultry manure was stored within close proximity of the stream) could be greatly improved. Very high concentrations of N in runoff were linked to extensive annual poultry manure applications across the farm and regular applications of ammonium nitrate.

**Nitrogen and phosphorus budgets for the case study farm**

To put the exports of N and P in runoff into perspective, reasonably crude N and P budgets for the case study farm over the 2 year monitoring period, per ha, are shown in Figures 9-1 and 9-2. The inputs are based on the total fertiliser inputs (Table 6-3) for that part of the cultivation area that fell within the monitored subcatchment, and divided by the area (ha).

The N removed in crops is based on 4 crops of lettuce over 2 years, at 30,000 kg (wet weight) per ha, per crop. Assuming moisture content of 95 %, the total dry weight of removed crop over 2 years was 6,000 kg. The concentration of N and P in lettuce is approximately 3.7 % and 0.7 % respectively (dry weight) (Reuter & Robinson, 1986).
Runoff losses of N accounted for 28% of fertiliser N inputs and coincidentally N removal in harvested crop also accounted for 28%. Over 50% of fertiliser N inputs were unaccounted for (ie. 790 - (222 +222)). Likely N losses are discussed below.

Figure 9-1. N budget for cultivation area over the 2 year monitoring period (kg ha⁻¹)

Figure 9-2. P budget for the cultivation area over the 2 year monitoring period (kg ha⁻¹)
Soil storage of N is unknown. Given that soil organic carbon (OC) increased slightly during the study (from 0.8 % to 0.9 % in 0-20 cm of soil) there may have been an increase in storage of organic N, but this should be viewed with caution as the soil sampling area was a small percentage of the cultivation area. More importantly, the OC concentration in the cultivation area (0-20 cm) was considerably lower than in the adjacent uncultivated pasture (3.0 %), so while there may have been small fluctuations in soil N on the cultivation area, overall all organic N must have been depleted over time due to a marked reduction in the concentration of OC. A significant accumulation of mineral N in soil is also unlikely. That is, the N budget is unlikely to include a significant net storage of N in the soil.

Loss of gaseous N and leaching of NO₃ are the other major unknown variables in this budget. Denitrification is a likely cause of N losses given the frequent irrigation that could lead to anoxia. Assuming that denitrification would lead to lower concentrations of NO₃ in runoff (because it is being converted to gaseous NO₂), there was no consistent pattern in concentrations of soluble N between closely occurring runoff events (when anoxia was most likely) to strongly suggest that the concentration of soluble N was lower after periods of wet weather. That is, a decrease in NO₃ concentrations was not consistently evident in the second of 2 closely occurring runoff events. Given the storage conditions of the poultry manure, gaseous losses of ammonia may account for N loss before the manure was even applied and in the period between being spread and incorporated into the soil.

In the budget, 346 kg of N are unaccounted for, so a potentially large amount of N may find its way into Currency Creek via leaching of NO₃. However, the B horizon is relatively impermeable and throughflow from the cultivation area is likely to be slow. Leaching from the poultry manure stockpile is potentially both a major loss from the farm and a major source of N in Currency Creek.

On the case study farm the inputs of P were grossly excessive considering that only 11 % was removed from the farm in harvested produce. Another 5 % was removed in runoff, most of which (>90 %) was particulate P. The bulk of applied P accumulated in the soil, providing no benefit to the farmer. This was reflected by an increase in Bray P in the soil from 174 mg kg⁻¹ to 304 mg kg⁻¹ and a concomitant decrease in P sorption from 129 mg kg⁻¹ to 104 mg kg⁻¹. A
marked increase in soil P above background concentrations was also reported by Jinadasa et al. (1997). The increase in the concentration of soluble P in runoff over the study period was most likely a result of the decrease in P sorption capacity of the soil.

**Economic incentives for reducing poultry manure use**

These findings suggest that application of P-containing fertilisers above the rate at which P was being removed from the system by crop uptake and soil erosion, was unnecessary and costly. The cost of applying poultry manure was $424 per ha over 2 years. To ‘balance’ the P budget, this could be reduced by 86 %, saving $365 per ha or $2400 over the farm over 2 years. However, this would mean that N input would be reduced, requiring a compensatory increase in ammonium nitrate or other N fertiliser. Other negative impacts of not applying poultry manure would also have to be considered. Poultry manure is a valuable source of organic matter in a farming system that generally removes more organic matter than it creates. Therefore by removing this source without compensating with some other source (eg. green manure), soil organic matter concentrations would decrease further, resulting in other potentially negative impacts on soil structure and cation exchange capacity (CEC).

According to Baird (1996) good fertiliser management should lead to crop uptake of 50-70 % of applied N. In South Australia, Holmes (1979) reported that N lost in runoff from a market garden was equivalent to 19 % of fertiliser inputs. On the case study farm, 28 % of applied N was lost in runoff and a total of 72 % of N was not harvested. This would suggest that N inputs on the case study farm were both excessive and poorly managed. As discussed, there were significant losses from the farm due to the manner in which poultry manure was stored and applied.

If a 50 % fertiliser efficiency could be achieved with mineral N (used to supply the N shortfall after decreasing poultry manure by 86 %), then the required fertiliser would cost $305 per ha or $2016 over the whole farm. The net saving from reducing poultry manure and increasing mineral N would be $384 over the farm. However, by green manuring N-fixing legumes, the need for mineral N could be reduced, while at the same time providing organic matter that was previously provided by poultry manure.
9.2.3 Cultivation and soil cover

Factors which influence soil erodibility include soil cover (Costin, 1980; Freebairn & Wockner, 1986) and cultivation (Hashim et al. 1995). Soil losses from the farm were compared with observations of soil cover and soil disturbance from 7 events. The number of runoff events for which soil cover data were collected are simply too few to conclusively determine the relationship, although it suggests that on the case study farm the relationship was similar to that shown for field studies (Costin, 1980; Freebairn & Wockner, 1986).

The loss of sediment in runoff events was related more to the amount of rainfall that fell at greater than 25 mm h⁻¹ (ΣI > 25) (R = 0.87) than peak discharge (R = 0.70). Rainfall intensity is an important component of ΣI > 25 and rainfall kinetic energy is a function of intensity (Rosewell, 1986; Kinnell, 1987). Soil cover can greatly reduce the impact of raindrops on the soil surface by absorbing energy (Moss, 1989). Therefore soil cover should play an important role in reducing soil erosion on the case study farm. Although difficult to compare with studies where soil cover was more uniform, spatially averaged soil cover was generally below 20 %, well below the 30 % recommended by Freebairn and Wockner (1987) for vertisols on the Darling Downs in Queensland.

The soil surface condition influenced soil loss. The concentration of sediment in closely occurring runoff events separated by 1-4 weeks was markedly lower in the second event due to settling of recently cultivated soil and hard setting. Concentrations of sediment were also higher in runoff from simulated rainfall on loose surface soil conditions.

Runoff volume

A crude water balance model showed that variations in runoff from 2 rainfall events (January and May, 1996) was due to antecedent soil water and the condition of the soil surface. The model indicated that irrigation of vegetable crops maintains a consistently wet soil profile, hence runoff occurred after relatively little rainfall, as in the May, 1996 event. Infiltration in January, 1996 was 20 mm more than in May, 1996 even though spatially averaged antecedent soil water was similar. The difference was attributed to the highly disturbed condition of the soil surface in January compared to May, thereby disrupting the surface seal/crust.
Runoff volume was considerably higher on bare fallow and bare tilled plots in the rainfall simulation study. A large proportion of the case study farm had been previously left in bare fallow over the summer months. The simulator work showed that a temporary cover crop which was later incorporated into the soil as green manure, greatly reduced the runoff volume, both before and after the sorghum was incorporated into the soil. The use of green manure crops offers the farmer a means of increasing infiltration and therefore reducing runoff volume. This should result in less soil erosion and nutrient loss, and an increase in soil organic matter which should improve soil structure. Another potential benefit is that improved drainage will improve irrigation efficiency, so that less water has to be applied to produce the same yield.

A study comparing efficiency of irrigation methods such as overhead sprinklers (which were used on the case study farm) surface drip and sub-surface drip systems at different soil water deficits would be beneficial, particularly if water prices increase. Drip methods offer potential to provide water efficiently to vegetable crops (Hanson et al., 1997).

Implications of soil loss

Loss of suspended sediment from the case study farm was equivalent to approximately 2.5 mm over 2 years. Over 6 cm (almost one third of the A1 horizon) of soil will be removed in 50 years. This is a conservative estimate because:

1) the limited time frame over which the monitoring took place did not include any events with a return period of greater than 5 years, and

2) bed load was assumed to be deposited in the channel before the sampling point and this may not have always been the case, particularly when the channel had not been excavated for a long period of time (ie. the channel was virtually filled in by sediment).

Significant loss in soil depth can cause major losses in soil productivity. The impacts of soil loss/erosion are outlined by Marschke (1988). Soil erosion effectively reduces the amount of water available to plants through reduced root volume and alteration of moisture characteristics of the soil. Similarly, soil erosion reduces the level of available nutrients, particularly those associated with organic matter, and increases mixing of A and B horizons, causing variability across the paddock. Soil loss places the root zone closer to plant-growth-
limiting horizons (eg. B horizon or saline groundwater). The impacts on the case study farm could be high because the A horizon is relatively shallow and overlies a relatively impermeable B horizon.

9.3 REFLECTIONS ON EXTENSION

Runoff monitoring on the case study farm necessitated regular field work on the farm. This provided the opportunity to build a rapport with the farmer and discuss issues related to nutrient runoff and soil erosion with the farmer on a regular basis.

While it is relatively easy to make recommendations about how land should be managed based on runoff and soil data, in order to actually bring about some change it is important to be actively involved and so recognise limitations and opportunities for influencing management. Senn (1996) and Hamilton (1996) based their research on participatory action and learning research (PLAR). This approach is often referred to as action research, where reflection is an integral part of the research process.

9.3.1 Awareness of land and water degradation issues

Mr Saliba demonstrated a good working knowledge of the affects of tillage on soil structure, such as his decision to grow oats and sorghum with the sole aim of increasing organic matter in the soil. Following the rainfall simulator work Mr Saliba commented that he had grown a legume green manure several years ago without much success, but he could not elaborate on the reason for lack of success.

Mr Saliba was careful about cultivating the soil when the moisture content was suitable (ie. not too moist and not too dry). In warmer months he always worked in bare feet to ‘get a good feel for the soil’. If the soil was too dry he irrigated or waited for rain before cultivating. He often delayed cultivating beds for several weeks until weather conditions enabled the soil to dry out sufficiently. These practices are consistent with scientific theory. Structural degradation is accelerated when soils are tilled while near their plastic limit (Utomo & Dexter, 1981).
Mr Saliba demonstrated an eagerness to learn about the behaviour of N and P in the soil. He gave the impression that he did not understand soil chemistry as well as he understood soil physical qualities. He showed a willingness to experiment with crop varieties and fertilisers, but never recorded any results or observations.

When asked about their attitudes to the offsite impacts of market gardening, the Salibas said they wanted to manage their farm to minimise adverse impacts on the river. However, they were much more comfortable when talking on subjects such as improving soil structure, knowing when to apply fertiliser, when the soil was at the right moisture to cultivate, how they managed weeds, and so on. They talked positively about these matters. Offsite environmental issues generally evoked a more defensive, negative response. For example, when we started talking about the offsite impacts of market gardening, Mr Saliba sometimes alluded to the inequity of the water licensing system, which at the time was under review.

9.3.2 Constraints and opportunities

It was obvious that the Salibas were concerned about managing their land well. While they did not want to cause adverse impacts on the receiving waterway, this sentiment did not influence management. An example of this was the poultry manure stockpile, which was positioned about 30 m from Currency Creek. It was apparent they were not overly concerned about soil erosion or applying a bit too much fertiliser as long as the produce continued to be of high quality. This apparent lack of concern is a constraint to bringing about a change in management of the farm.

However, the Salibas were very conscious about the need to farm efficiently to remain profitable. This led to significant changes in their practices in the early to mid 1990’s. Decisions to reduce tillage (Senn, 1996) and fertiliser use were based on the potential to reduce operating costs without compromising productivity.

Changing attitudes about the need to minimise impacts on waterways in order to achieve changes in farm management is a long term process. It would appear that bringing about more immediate changes depends on being able to link them with economic incentives. Better
knowledge of N and P behaviour should improve fertiliser management. For example, any attempt by an extension officer to get Mr Saliba to move the stockpile of manure away from the Creek would probably only agitate Mr Saliba. More success might be achieved if Mr Saliba could be convinced that by changing storage conditions he might conserve the manure’s nutrient content. A better outcome would have been achieved by focusing on the economic incentives rather than environmental impacts of doing nothing.

9.4 RECOMMENDATIONS FOR EXTENSION AND ADOPTION OF IMPROVED LAND MANAGEMENT

This research suggests that 2 major aspects of farm management need to be improved to reduce losses of sediment and nutrients. Firstly, losses can be minimised by improving soil and nutrient management practices. These will reduce soil and nutrient loss at source and improve soil structure in the long term. Secondly, structural measures can be used to intercept and trap potential pollutants in runoff water. This work emphasises the need for whole farm planning, integrating better soil and nutrient management underpinned by an active soil and plant nutrient testing program so that fertilisers are applied according to nutrient requirements of crops, with structural measures which trap eroded sediment and assimilate nutrients.

9.4.1 Source prevention - the primary focus

The need for judicious use of fertilisers based on regular soil and plant testing is highlighted by these results. It would appear that applying poultry manure to provide P was largely unnecessary, and it was probable that any real benefits were related to N and organic matter maintenance. An increase in organic matter could be achieved by other sources such as green manure crops and permanent cover crops (Theunissen, 1997). If legumes are used as green manure, fertiliser N inputs could also be decreased.

Excessive fertiliser use reduced profitability. In the case of P, a saving of up to $2400 in fertiliser costs could be achieved by reducing poultry manure by 86 % (so that P added = P removed). However, N inputs would then be insufficient to maintain yield and there would also be a further decline in soil organic matter. Green manuring with N-fixing legume crops would at least partly correct the shortfall of N and increase soil organic matter, but this option
requires research (see 9.5). However, if legume green manures were not used, additional use of mineral N to the value of $2016 (assuming 50% fertiliser efficiency), would mean that the net saving would be just $384 over the farm in 2 years.

In general, market gardeners need to be more concerned about sustainability, both economic and environmental. Better knowledge of soil and nutrient management including alternative nutrient sources and soil improvement should lead to more sustainable land management. In particular, there is need to base fertiliser application on soil and plant tissue testing, rather than applying current rates. Based on past experiences (Hamilton, 1995; Senn, 1996), an action learning project would facilitate co-learning among farmers and extension staff.

9.4.2 Structural measures - the secondary focus

The use of sediment and nutrient trapping measures holds considerable potential to minimise the offsite impacts of market gardening. Where market gardens are located adjacent to waterways (as the case study farm is) structural measures should be in place to reduce impacts on the waterway.

Soil erosion was an important mechanism for nutrient transport (90% of P and 24% of N was transported as particulate matter) on the case study farm. Measures which trap sediment will also prevent some nutrients from entering waterways. Wetlands and grassed filter strips should ideally be placed below sedimentation dams to maximise biological removal of soluble nutrients. However, given the spatial and maintenance requirements, optimal design of structural works will rarely be possible on market gardens. Practical design farm will only be achieved by assessing farms individually.

On the case study farm either a grassed filter strip or constructed wetland in the drainage line, combined with a sedimentation basin at the downstream end of the drainage line, would greatly increase the trapping of sediment and nutrients removed from the vegetable beds, provided they were appropriately designed. Negative impacts of a sediment basin would need to be considered, particularly land requirements. Such measures are likely to require additional technical and managerial skill, particularly in respect of maintenance of a grassed filter strip or constructed wetland. On the case study farm, both could be used periodically as fodder for
their 2 or 3 beef cattle, thereby harvesting the filter strip or wetland, which the literature suggests is necessary for the long term success of wetlands (Redding et al., 1997).

The practice of stockpiling poultry manure uncovered and close to waterways is an important issue that needs to be addressed. There are 2 concerns: the location of the stockpile, which on the case study farm was adjacent to a receiving waterway; and the manner in which the poultry manure is stored. On the case study farm there were no measures in place to prevent leaching or runoff of nutrients from leaving the site.

9.5 FURTHER RESEARCH

1) This research applies to soil derived from Wianamatta Shales, an important soil type in the region. Further research is needed to assess the effect of current management practices on other soil types in the Hawkesbury-Nepean.

2) The relationship between soil P and soluble P in runoff requires confirmation over longer time periods.

3) The use of farm dams, constructed wetlands and grassed filter strips in market gardening should be examined for both their efficiency in reducing sediment and nutrient loads and for their practicality.

4) The effect of reducing nutrient inputs on yield remains a contentious issue (T. Wells, pers. comm., 1996) and research is required to determine precisely why market gardeners persist with high application rates when current soil tests (eg. Bray) indicate that soil nutrients are excessive on the majority of market gardens.

5) The potential of legume green manures to supplement/replace mineral fertilisers and improve soil organic matter, requires further research and development and this could be incorporated into an action learning extension project (see 9.4)
REFERENCES


References


References


References


*References*  
169


Rosewell, CJ (1993) *SOILOSS A program to assist in the selection of management practices to reduce erosion.* Department of Land and Water Conservation, Gunnedah, NSW.


Senn, AA (1996) *Experiential learning as a basis for extension practices with Maltese vegetable growers of western Sydney.* M.Sc.(hons) thesis, University of Western Sydney, Hawkesbury, Richmond, NSW.


References
*Advances in Agronomy*. 52, 1-83.


Sutherland, RA, Wan, Y, Ziegler, AD, Lee, CT and Elswaify, SA (1996) Splash and wash dynamics - 

Sumner, HR, Wauchope, RD, Truman, CC, Dowler, CC and Hook, JE (1996) Rainfall simulator and 

Theunissen, J (1997) Intercropping in field vegetables as an approach to sustainable horticulture. 

Renewable Resource* *Proceedings of the 4th International Symposium on Livestock Wastes 1980*. 


Wace, SA and Hignett, CT (1991) The effect of rainfall energy on tilled soils of different dispersion 

Watts, CW and Dexter, AR (1997) The influence of organic matter in reducing the destabilization of 

Weaver, DM, Pen, LJ and Reed, AEG (1996) Phosphorus management in the Oyster Harbour 
catchment (Western Australia) to minimise downstream effects. In: H M Hunter, A G Eyles and G E 
Rayment (eds) *Downstream effects of land use*. Department of Natural Resources, Qld.

Wells, T (1996) *Environmental impact of alternative horticultural production systems in the 
Hawkesbury-Nepean catchment*. NSW Agriculture, Gosford, NSW.

commissioned by Australia's rural research development corporations*. Land and Water Resources 
Research and Development Corporation, Canberra.

Young, RA, Onstad, CA, Bosch, DD and Anderson, WP (1994) Agricultural non-point source pollution 
model, version 4.02 AGNPS user’s guide. Minnesota Pollution Control Agency, Minnesota, USA.

APPENDIX

A. Examples of Farm Survey Maps

B. Relationships Between Concentration Suspended Solids vs Particulate N and P in Runoff Samples

C. Calibration Charts for Veejet 8070, 8050 and 8030 Nozzles
A. Examples of Farm Survey Maps

Station 6: Sketch Map

Date: 9/5/96

General Observations:

+ ~ 10% of cropping area bare + 9% bare stubble

+ marked difference in water quality runoff (forbids) from the oats stubble to adjacent bare, reformed seeds during the preceding runoff event.

+ expanding area of lettuce production (see sketch map for 7/5/96)

Figure A-1. Farm survey map from the case study farm
Figure A-2. Farm survey map from the Borg farm (Note the complexity of farm layout)
B. Relationships Between Concentration Suspended Solids vs Total Particulate N and P in Runoff Samples

The relationship between the concentration of suspended solids and other pollutants is of particular interest to modellers. If other pollutants can be predicted from the concentration of suspended solids, modelling can be simplified and done in less time. Similarly, the cost of monitoring pollutants can be drastically reduced because the number of parameters analysed can be reduced, and suspended solids is relatively cheap and easy to measure (no chemistry is required). Figure B-1 shows the relationship between suspended solids and particulate P. Figure B-2 shows the relationship between suspended solids and particulate N. It should be stressed that the relationships are site-specific and may differ markedly to market gardens on other soil types particularly.

![Graph](image1)  
*y = 0.0012x (P<0.01)*  
*R^2 = 0.76*

**Figure B-1.** Relationship between the concentration of suspended solids and total P in runoff samples

![Graph](image2)  
*y = 0.0004x (P<0.01)*  
*R^2 = 0.94*

**Figure B-2.** Relationship between the concentration of suspended solids and particulate N in runoff samples
C. Calibration Charts for Veejet 8070, 8050 and 8030 Nozzles

Figure C-1. Calibration chart for the Veejet 8070 nozzle @70kPa/Dwell time=2.2

Figure C-2. Calibration chart for the Veejet 8050 nozzle @60kPa/Dwell time=2.2

Figure C-3. Calibration chart for the Veejet 8030 nozzle @50kPa/Dwell time=2.2
Links between management of a market garden and stormwater losses of sediment, nitrogen and phosphorus

Eric Hollinger

Faculty of Environmental Management and Agriculture
University of Western Sydney - Hawkesbury

A thesis submitted for the fulfilment of the Degree of Master of Science (Hons)

March 1998
PLEASE NOTE

The greatest amount of care has been taken while scanning this thesis,

and the best possible result has been obtained.
SUMMARY

The Problem
Market gardening is commonly characterised by intensive cultivation, high inputs of both organic and inorganic fertilisers, chemical over/misuse, frequent irrigation, and a low degree of soil cover. While market gardening is readily perceived to be detrimental to waterways, there is remarkably little data to quantify the impacts. Soil and nutrient loss in stormwater runoff varies with soil type, climate and production systems. Therefore local data are needed to determine the impact of market gardening on the Hawkesbury-Nepean. This should lead to a better understanding of how land management influences runoff quantity and quality so that practices can be improved.

Objectives of this research were to:
1) quantify sediment, N and P loss and assess the implications for waterways;
2) relate sediment, N and P losses to specific land management practices and assess their impacts on profitability; and
3) reflect on this research in terms of extension and adoption of better land management.

Methods
An 8.8 ha property with 6.6 ha of market garden was used as a case study in the Hawkesbury-Nepean Catchment. The farm is situated near Freeman’s Reach, between North Richmond and Windsor. Sediment, N and P losses in stormwater runoff were measured for a 2 year period (1995 to 1997). Soil samples were collected at the beginning and end of the study. Sediment core samples were collected from the drainage channel. A rainfall simulator was used to compare runoff volume from green manure and bare fallow beds.

Description of the farm
The soil type, a Brown Podsolic (Db2.41, Db1.41), is moderately to highly erodible, hard setting and is prone to surface sealing. Farm management was consistent with other market gardens situated on poorly drained loams derived from Wianamatta Shale. The main crops grown were lettuce, capsicum, spinach and cucumber. During the study, the fertilisers used were poultry manure and ammonium nitrate.
Key Findings

Environmental implications of sediment, N and P losses in stormwater

Concentrations of N and P in runoff from the farm were several orders of magnitude higher than ANZECC guidelines for the protection of Australian freshwater streams. Sediment loss was equivalent to more than 19 t ha\(^{-1}\) yr\(^{-1}\) or 2.5 mm over 2 years. Total N loss was 1599 kg, equivalent to export of 127 kg ha\(^{-1}\) yr\(^{-1}\). Total P loss was 138 kg, equivalent to export of 11 kg ha\(^{-1}\) yr\(^{-1}\). These rates are very high in relation to the Australian and international data.

Links with management and implications for profitability

Overuse of fertiliser was a major factor contributing to the high losses of N and P. Fertiliser inputs were equivalent to 414 kg ha\(^{-1}\) yr\(^{-1}\) of N and 182 kg ha\(^{-1}\) yr\(^{-1}\) of P. The concentration of soluble P in runoff increased significantly (P<0.01) over the course of the study. Extractable P (Bray) in the soil increased from 174 mg kg\(^{-1}\) in May, 1995, to 304 mg kg\(^{-1}\) in January, 1997. There was a concomitant decrease in P sorption from 129 mg kg\(^{-1}\) to 104 mg kg\(^{-1}\). A comparison with runoff from a nearby but older market garden suggested that the increase in concentration of soluble P in runoff is likely to continue at current rates of fertiliser application.

Excessive fertiliser use reduced profitability. In the case of P, a saving of up to $2400 in fertiliser costs could be achieved by reducing poultry manure by 86% (so that P added = P removed). However, N inputs would then be insufficient to maintain yield and there would also be a further decline in soil organic matter. Green manuring N-fixing legume crops would at least partly correct the shortfall of N and increase soil organic matter. However, if legume green manures were not used, additional use of mineral-N to the value of $2016 (assuming 50% fertiliser efficiency), would mean that the net saving would be just $384 over the farm in 2 years. A rainfall simulation experiment showed that green manure crops provided additional benefits by increasing infiltration in the pre and post-incorporation stages, hence reducing soil and nutrient runoff losses.

Soil erosion was an important mechanism for nutrient transport (90% of P and 24% of N was transported as particulate matter). High soil loss was due to soil disturbance resulting from cultivation, a low degree of soil cover, and very low infiltration rates. Average soil cover was
less than 20% on the cultivation area for most of the study. In addition to better soil management practices, soil loss from the farm could be further reduced if a new dam was constructed to collect runoff from the cultivation area. Alternatively, purpose-built sediment traps could be constructed.

**Recommendations for extension and adoption of improved land management**

In order to reduce sediment, N and P losses in stormwater, the primary focus should be on improving soil and nutrient management, in particular matching fertiliser inputs more closely to nutrient requirements. The secondary focus should be on utilising structural measures, in particular farm dams, to prevent pollutants from entering waterways. The outcome should be decreased costs to the farmer and decreased impacts on waterways.

The use of N-fixing green manures to decrease the use of poultry manure should be explored in further research.
ACKNOWLEDGMENTS

This research has only been possible because of the generous support of a market gardener and his family, who wish to remain anonymous. Their willingness to participate and assist in this work is gratefully acknowledged.

Many thanks to the other members of the NLP project including Professor Peter Cornish, Dr Barbara Baginska (UWSH), Dr David Jones, Mr Ken Riley, Ms Rosemary Wood (CSIRO), Dr George Kuczera (Newcastle University), Ms Linda Henderson (DLWC). Special thanks to Dr Barbara Baginska (UWS Hawkesbury) for assistance with analysis of hydrological data and helpful advice and Ms Linda Henderson (Department of Land and Water Conservation) for assistance with analysis of soil data. Professor Peter Cornish (UWSH) ably lead the NLP project and also was the principal supervisor for this thesis and deserves special thanks.

Thanks to co-supervisor Dr Rob Mann for providing constructive feedback on the manuscript. Thanks also to Dr Steven Riley and Dr Basant Maheshwari for advice and comments.

Thanks to Mr Pat Hanson for assistance with modifying and testing the rainfall simulator, and to Mr Steve Muller and Mr Burhan Amiji who assisted with the rainfall simulation work. The NSW Agriculture Organic Waste and Recycling Unit at Richmond provided the rainfall simulator.
"Today we have farmers that publicly question whether they are farming 'sustainably'... and challenge science to define the land management practices that need to be implemented to be 'sustainable'. And as we grapple with those challenges and what they mean, it appears we need new ways of looking at the world and integrating management and research."

Allen and Bosch, 1996
TABLE OF CONTENTS

Declaration .................................................. i
Summary ..................................................... ii
Acknowledgments ........................................... v
Table of Contents .......................................... vii
List of Tables ............................................... ix
List of Figures .............................................. x
List of Plates ............................................... xi

1. INTRODUCTION
    1.1 Market gardening and the environment - research objectives ................. 1
    1.2 Context of this research .................................................. 5
    1.3 Thesis structure .......................................................... 6

2. PHYSICAL AND SOCIAL FEATURES OF THE MARKET GARDENING INDUSTRY IN THE HAWKESBURY-NEPEAN CATCHMENT
    2.1 Location of market gardens in the Hawkesbury-Nepean catchment ............... 8
    2.2 Market gardeners and their farms ....................................... 10
    2.3 Extension issues ....................................................... 15

3. LITERATURE REVIEW: SEDIMENT AND NUTRIENT LOSS IN AGRICULTURAL RUNOFF
    3.1 Techniques for measuring sediment and nutrient loss in runoff ................. 18
    3.2 Current estimates of nutrient exports from market gardening ................. 22
    3.3 Soil erosion ............................................................ 25
    3.4 Nitrogen sources and transformations in soils on market gardens ............. 42
    3.5 Phosphorus sources and transformations in soils on market gardens .......... 44
    3.6 Transport of nitrogen and phosphorus in runoff ................................ 47
    3.7 Land management practices that reduce offsite impacts of soil and nutrient loss .................................................. 52
    3.8 Summary ............................................................... 55

4. SYNTHESIS OF RESEARCH QUESTIONS
    4.1 Sediment and nutrient loss - implications for waterways .................... 56
    4.2 Links between management and stormwater losses - economic implications .. 56
    4.3 Extension and adoption of improved land management ........................ 58
5. MATERIALS AND METHODS FOR THE CASE STUDY
   5.1 The case study farm .................................................. 60
   5.2 Soil measurements .................................................. 72
   5.3 Collection of land management data .............................. 74
   5.4 Surface water runoff .............................................. 78
   5.5 Rainfall measurements and analysis .............................. 86

6. RESULTS AND DISCUSSION 1: SOIL AND LAND MANAGEMENT ON THE
   CASE STUDY FARM
   6.1 Introduction ....................................................... 87
   6.2 Soil analysis ..................................................... 87
   6.3 Land management data ........................................... 90
   6.4 Summary .......................................................... 97

7. RESULTS AND DISCUSSION 2: SUSPENDED SEDIMENT, NITROGEN AND
   PHOSPHORUS LOSSES IN RUNOFF - JUNE 1995 TO MAY 1997
   7.1 Introduction ....................................................... 99
   7.2 Results and discussion ......................................... 100
   7.3 Summary .......................................................... 123

8. RESULTS AND DISCUSSION 3: COMPARISON OF LAND MANAGEMENT
   WITH A RAINFALL SIMULATOR
   8.1 Introduction ....................................................... 125
   8.2 Materials and methods .......................................... 129
   8.3 Results ........................................................... 137
   8.4 Discussion ......................................................... 142
   8.5 Summary .......................................................... 148

9. FINAL DISCUSSION AND CONCLUSIONS
   9.1 Sediment, N and P losses - implications for waterways ......... 149
   9.2 Links between management and stormwater losses of sediment, N and P 150
   9.3 Reflections on extension ........................................ 158
   9.4 Recommendations for extension and adoption of improved land management 160
   9.5 Further research .................................................. 162

REFERENCES

APPENDICES
LIST OF TABLES

Table 2-1. Summary of climatic variables at Richmond 10
Table 3-1. CMSS estimates for land uses in South East Australia 23
Table 3-2. Suggested K values for a range of texture classes 34
Table 5-1. An extract from the data file from February 1997 81
Table 5-2. Limits of detection for suspended solids and nutrient forms measured 85
Table 6-1. Comparison of electrical conductivity, pH, organic carbon, available P, and P sorption in soils from the case study and adjacent farms, with different land use history 88
Table 6-2. Plant available (Bray) P, P sorption and total P in drainage channel sediment compared to the vegetable beds 90
Table 6-3. Total fertiliser inputs of N and P for the case study farm from June 1995 to May 1997 91
Table 6-4. Total soil cover [%] on the 2 market gardens during selected runoff events 95
Table 6-5. Cropping patterns and % of vegetable growing area under irrigation 96
Table 7-1. Hydrological characteristics of major runoff events 101
Table 7-2. Comparison with runoff quality between the case study farm and the Borg farm during event 7 113
Table 7-3. Export of N, P, and suspended solids (SS) from major runoff events 114
Table 8-1. Nozzle calibrations @ 2.2 sec dwell time 132
Table 8-2. Summary of comparisons made with rainfall simulation runs 133
Table 8-3. Infiltration rates calculated from 15-20 minutes 140
Table 8-4. Levels of organic carbon [%] in adjacent soils with different summer management histories 141
Table 8-5. Sediment concentrations from bare, sorghum and green manure with soil condition and rain intensity 142
LIST OF FIGURES

Figure 2-1. Areas where market gardening is concentrated in the Hawkesbury-Nepean catchment 8
Figure 3-1. Relationship between particle size and velocity of flow 28
Figure 3-2. Idealised fluvial system 29
Figure 3-3. Intensity-energy data and relationship for Canberra 32
Figure 3-4. Conceptual framework of factors affecting soil erodibility 34
Figure 3-5. Soil loss ratio as influenced by cover on vertisols of the Darling Downs compared to the relationship derived from the USLE 40
Figure 3-6. Forms and fate of nitrogen in poultry manure 42
Figure 3-7. General trend of the effect of tillage on losses of N and P 48
Figure 3-9. Relationships between pollutant concentrations and discharge 51
Figure 5-1. The Currency Creek subcatchment, showing the case study farm (stn 6) and Currency Creek 61
Figure 5-2. Schematic of layout of the case study farm 63
Figure 5-3. Schematic diagram of the monitoring equipment at station 6 79
Figure 5-4. Flow diagram showing sample processing methods 84
Figure 6-1. Particle size distribution along the drainage channel and the parent soil 89
Figure 6-2. Soil cover on the cultivation areas of a) the case study farm and b) the Borg farm, as calculated from 4 farm surveys 92
Figure 6-3. Soil condition on the cultivation areas of a) the case study farm and b) the Borg farm, as calculated from 4 farm surveys 93
Figure 6-4. Relationship between soil cover and soil disturbance (SC2+SC3) on the case study farm 94
Figure 6-5. Calculated water balance from 1/12/95 to 9/5/96 97
Figure 7-1. Comparison of actual rainfall at Currency Creek and long term monthly average rainfall for Richmond 100
Figure 7-2. (a) Standard hydrograph, and (b) cumulative rainfall and runoff, for the May 1996 event 103
Figure 7-3. Hydrograph for event 7 (January, 1996) 104
Figure 7-4. Comparison of rainfall totals and runoff coefficients 105
Figure 7-5. Mean sample concentrations of N forms in runoff events 106
Figure 7-6. Mean sample concentrations of P forms in runoff events 107
Figure 7-7. Concentrations of suspended solids, N and P through event 7 109
Figure 7-8. Minimum, maximum, and mean concentrations of soluble N (a) and P (b) 112
Figure 7-9. Comparison of (a) peak discharge [m$^3$ min$^{-1}$] and (b) $\sum_{i=1}^{25} \text{mm} h^{-1}$ [mm], with sediment loss 116
Figure 7-10. Comparison of (a) peak rainfall intensity [mm h$^{-1}$], (b) the area of disturbed soil (SC2+SC3) [ha], and (c) total soil cover [%], with concentration of suspended solids [kg ML$^{-1}$] 118
Figure 7-11. Comparison of exports of suspended solids [kg] with (a) particulate N [kg] and (b) particulate P [kg] 121
Figure 7-12. Comparison of discharge [m$^3$] and loss of soluble N [kg] 122
Figure 8-1. Schematic diagram of the rainfall simulator and collection system 130
Figure 8-2. Comparison 1: bare fallow and trimmed, mature sorghum at $I = 62$, with dry soil water status 138
Figure 8-3. Comparison 2: bare fallow and trimmed, mature sorghum at $I = 46$, with moderately moist soil water status 138
Figure 8-4. Comparison 3: Runoff from bare fallow and incorporated sorghum at $I = 46$ 139
Figure 8-5. Comparison 4: Runoff from bare fallow and incorporated sorghum at $I = 62$ 139
Figure 9-1. N budget for the 2 year monitoring period [kg ha$^{-1}$] 155
Figure 9-2. P budget for the 2 year monitoring period [kg ha$^{-1}$] 155
# LIST OF PLATES

<table>
<thead>
<tr>
<th>Plate</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
<td>View of the case study farm, looking upslope at the drainage channel in dry and wet conditions</td>
<td>62</td>
</tr>
<tr>
<td>2.</td>
<td>Raised beds with recently planted lettuce seedlings and sprinkler irrigation system; Note the neat, weed-free appearance that is typical of the Saliba farm</td>
<td>70</td>
</tr>
<tr>
<td>3.</td>
<td>Finely tilled beds with maximum relief</td>
<td>70</td>
</tr>
<tr>
<td>4.</td>
<td>Coarsely tilled beds with lower relief (Note large clods and weed residue)</td>
<td>71</td>
</tr>
<tr>
<td>5.</td>
<td>Lightly tilled beds with partially intact cover of dead weeds, into which a sorghum green manure crop was sown</td>
<td>71</td>
</tr>
<tr>
<td>6.</td>
<td>Monitoring station 6, on the boundary of the case study farm</td>
<td>80</td>
</tr>
<tr>
<td>7.</td>
<td>Bare fallow beds with the rainfall simulator in place</td>
<td>135</td>
</tr>
<tr>
<td>8.</td>
<td>The sorghum beds as they were prepared for the rainfall simulation</td>
<td>135</td>
</tr>
<tr>
<td>9.</td>
<td>Sorghum residue provided considerable soil conservation value during the February, 1997 runoff event</td>
<td>145</td>
</tr>
</tbody>
</table>