Comparative effectiveness of community drainage schemes and EVTAs in trapping PSII Herbicides from sugarcane farms

Stage 1 Report

ACTFR REPORT NUMBER 10/14

Prepared by Jon Brodie, Jane Waterhouse, Stephen Lewis, Amy Basnett, Aaron Davis, Frances Alexander and Melany Ginders

Australian Centre for Tropical Freshwater Research, James Cook University, Townsville, 4811

Report version 22 August 2011

Table of contents

List of Figures 4

List of Tables 5

Acknowledgements 6

Disclaimers 6

Executive Summary 7

1. Background to the Project 8

Definitions 9

2. Introduction 10

2.1 Sugar industry in Queensland 10

2.2 The Russell-Mulgrave area 14

2.3 Herbicide application and types of usage and management practices 16

2.4 The GBR and herbicide concerns 17

3. Overview of the Babinda Community Drainage Scheme 18

4. Properties of relevant herbicides 21

4.1 Chemical properties 21

4.2 Toxicity 21

4.3 Pesticides used in the sugar cane industry 23

4.4 Pesticides measured in the GBR catchments 24

5. Herbicide use in the area 25

6. Drainage characteristics of the Study Area 26

6.1 Soils 26

6.2 Climate and Hydrology 28

7. Effectiveness of constructed wetlands, vegetated strips and drains in trapping of herbicides (and suspended sediments and nutrients) 34

7.1 Vegetated buffers for reducing pesticide loss in runoff 34

7.2 Pesticide transport 37

7.3 Trapping efficiency 39

7.4 Buffer width 40

7.5 Influence of climatic conditions 45

7.6 Influence of residence times 48

7.7 Predictive models for estimating trapping efficiency 48

8. Relevance to the Babinda Community Drainage Scheme 51

8.1 Theoretical functioning of trapping in riparian vegetation 51

8.2 Effectiveness of different vegetation types 51

8.3 Critical features of riparian vegetation in trapping pollutants 52

8.4 Babinda conditions and likely effectiveness of trapping of the selected herbicides 52

8.5 Other Queensland cane growing regions 53

9. Experimental design **54**
10. References **58**

Note that **Appendix 1** is provided as a separate document and is the complete (tabulated) literature review of the comparative effectiveness of vegetated buffers in trapping pesticides, sediments and nutrients in runoff from agricultural lands.

List of Figures

Figure 2-1. The Great Barrier Reef Catchment Area. 11

Figure 2-2. Sugarcane growing areas in the Great Barrier Reef Catchment area. Source: QLUMP (1999). 12

Figure 2-3. Wet Tropics Coastal area land use. Source: QLUMP (1999); Johnstone catchment QLUMP (2004). 13

Figure 2-4. Sugarcane in the Russell catchment. Source: Google Earth, downloaded October 2010. 14

Figure 2-5. Example of a flood plume at the mouth of Russell-Mulgrave River. Source: GBRMPA. 15

Figure 2-6. Flood plumes from Wet tropics Rivers on the (a) 9th, (b) 11th and (c) 13th February, 2007 moving from inner shelf waters on the 9th to the Coral Sea by the 13th. 15

Figure 2-7. Boom spraying of herbicides on a sugarcane farm. Photo: B. Masters 16

Figure 2-8. The control of weeds using shielded sprayers is being encouraged as an improved management practice in the sugar industry. This approach targets weeds early, reduces the total amount of herbicide applied to the paddock, uses products less susceptible to runoff (e.g. glyphosate), and reduces the need for residual herbicide control. Photo: B. Masters 17

Figure 3-1. Location of the Babinda Community Drainage Scheme in the Russell Mulgrave catchment. 18

Figure 3-2. Boundaries of the Babinda Community and Matthews Road Drainage Schemes. 19

Figure 3-3. Current knowledge of drainage infrastructure within the Babinda Community Drainage Scheme. ... 20

Figure 6-1. Dominant soil types in the Babinda area. Derived from Murtha *et al*. (1996). 27

Figure 6-2. Topography of the Russell Mulgrave catchment area. 28

Figure 6-3. Russell-Mulgrave isohyet rainfall patterns. 30

Figure 6-4. Daily rainfall totals (mm) from the Babinda Post Office BOM climate station for the period 22nd August- 4th October, 2010. 30

Figure 6-5. Daily river height data: Russell River at Bucklands (GS111101D; 1980-present). 32

Figure 6-6. Daily river height data: Babinda Creek at the Boulders (GS111105A; 1980-present). 33

Figure 7-1. The primary pathways of loss of pesticides from agricultural land. Source: USDA (2000). 36

Figure 7-2. How the grass filter strip and riparian buffer zone functions to remove sediments and nutrients from agricultural runoff. Source: The Idea to Here - Lovett and Price (2001). 36

Figure 7-3. How a grass buffer strip functions to trap sediment. Source: The Idea to Here - Lovett and Price (2001). 36

Figure 7-4. Relationship between percent of pesticide trapped and Koc. Source: USDA (2000). 38

Figure 7-5. Buffer width and sediment trapping efficiency. Source: Yuan *et al*. (2009). 41

Figure 7-6. Nonlinear relationship between cumulative rainfall and runon (cm) entering a vegetative filter strip (VFS) versus percent pesticide reduction (%) relative to two different buffer lengths (1.5 and 9.1m) as predicted using the VFSMOD model for the U.S. EPA Illinois corn scenario. Data shown are for a pesticide with organic carbon sorption coefficient, Koc = 100L/kg OC. Source: Sabbagh *et al*., (2010). 46

Figure 9-1. Rainfall simulator use in testing riparian trapping (from Hairsine 1997). 55

Figure 9-2. Sediment fans formed in grass filter strips in the Tarago catchment experiments (from Hairsine 1997). 55

Figure 9-3. Typical layout of paired trials in a actual paddock situation (Experimental set-up consisting of four blocks, each with three treatment plots). Source: Duchemin and Hogue (2009). 56

Figure 9-4. Diagram of a) tilted-bed setup and b) side view catchment and drain area of the tilted bed. Source: Seybold *et al*. (2001). 56

Figure 9-5. Schematic diagram of the experimental supply and monitoring system for artificial runoff. Source: Abu-Zreig *et al.* (2003). 57

List of Tables

Table 2-1. Area of sugarcane land use in Queensland in 1999 (Dwyer, unpublished).. 10

Table 4-1. Chemical properties and toxicity of the four herbicides of interest. 21

Table 4-2. Summary data on the effects of herbicides to marine plants of the Great Barrier Reef. 23

Table 6-1. Outline of the major distinguishing features of the dominant soil types in the Babinda Community Drainage Scheme area. 26

Table 6-2. Daily and Monthly Rainfall Data from Babinda Post Office (BOM site 031004). 29

Table 6-3. Daily and Monthly Rainfall Data from Bellenden-Ker Top Station (BOM site 031141)........................ 29

Table 7-1. Summary of buffer studies measuring trapping efficiencies for specific pesticides. Note: Shaded cells indicate pesticides used in sugar cane application in the GBR catchments. Source: Derived from USDA (2000). 38

Table 7-2. Trapping efficiencies for various buffer widths and rainfall conditions... 41

Table 7-3. Indicative soil losses and design filter widths for two of the six bio-geographical regions of Queensland, for varying rainfall erosivity, soil erodibility, slope and land cover. Source: Karssies and Prosser (1999). 45

Acknowledgements

Funding for this project came from the Queensland Department of Environment and Resource Management (DERM). We would like to thank the staff from DERM and the participants of a workshop held in October 2010 for review and improvement of this report.

Disclaimers

Executive Summary

Recent legislative amendments in Queensland have introduced mandatory requirements for the use of herbicides containing atrazine, diuron, ametryn and hexazinone when undertaking an agricultural Environmentally Relevant Activity (agricultural ERA). The ERAs defined in this legislation are grazing and sugarcane farming in the Wet Tropics, Burdekin Dry Tropics and Mackay Whitsunday catchments. The requirements restrict the use of these herbicides within 20m of all water bodies, although this may be difficult for existing farmers to meet, and alternative management options are suggested. One alternative is for a minimum 5m wide Effective Vegetated Treatment Area (EVTA) adjacent to the crop at the run-off points to be put in place.

Industry representation has identified that farms within the Babinda Community Drainage Scheme (BCDS) are unable to comply with the 20m buffer and are also resisting the installation of 5m EVTAs since they are already using a community drainage scheme that they say results in the same reduction of herbicides in run-off. Additionally, in correspondence to industry representatives, DERM agreed that *compliant* Community Drainage Schemes would be recognised as an equivalent to the alternative EVTA option.

This report has been prepared by the Australian Centre for Tropical Freshwater Research (ACTFR) to assist DERM in assessment of the compliance of this drainage system as an alternative management option. It provides information on the hydrology of the Russell-Mulgrave catchments, the properties of ametryn, atrazine, hexazinone and diuron, a summary of herbicide use in the area, a literature review of the effectiveness of constructed wetlands, vegetated strips and drains in trapping of herbicides (and nutrients and suspended sediments), and assessment of the relevance of this information to the BCDS. Discussion of a suitable experimental design to test the effectiveness of EVTAs compared to the BCDS components in the effective trapping/removal of the above herbicides is also provided.

From the literature it is clear that under suitable conditions vegetated buffers can trap high proportions (>50%) of pesticides from runoff, even for very soluble pesticides with low K_{OC} values. However, trapping efficiency is dependent on several factors including rainfall, water table conditions, solubility of pesticides, slope, and the type and condition of buffer vegetation. The combination of these characteristics in the Babinda area means that buffer widths for effective trapping of the selected herbicides will need to be in excess of 10m (with good grass cover) and in some circumstances (poorer grass cover) in excess of 30m. In addition, for these particular herbicides, while infiltration may occur they may be no further trapping on infiltrated soil and effective sub-surface transport to the closest drain or stream is likely (albeit in some longer timeframe than for surface transport). The final fate of the infiltrated pesticides is unclear but it is quite possible that transport to an adjacent drain or stream could be rapid with little further loss of herbicide, thus minimizing any net trapping. If real input data from a Babinda site could be used within a predictive model of the effectiveness of trapping (reviewed in the report), a quantitative assessment of likely trapping of herbicides with EVTAs could be made.

For the assessment of the BCDS as an herbicide trapping mechanism, it is unlikely that any major trapping of dissolved phase, low K_{OC} pollutants will occur in the mole drains and even less in the sub-surface piped drains. In the major drainage network, no infiltration can occur due to the high water table but some sedimentation (but mostly for particle bound pesticides – not the ones in our study) could occur if residence times were long enough – once again in the order of 5 days or longer. In reality in the 10km length of the drainage, preferred flow path water residence times in low flow conditions will be in the order of < 1 day (at a water velocity of 0.5 m/sec – 6 hours). Therefore, minimal trapping of these herbicides is likely in the drainage scheme in Babinda conditions.

Despite the conclusions relevant to the BCDS, trapping of herbicides using EVTAs may be possible in other sugarcane growing regions of the GBR catchments. In particular, EVTAs and vegetated drains may work effectively in irrigation tailwater and small first flush conditions in the lower Burdekin. Similarly in the less intense rainfall conditions of the Mackay Whitsunday and Burnett Mary cane growing regions, trapping in first flush conditions may occur. However, further studies are required to quantify these suggestions.

1. Background to the Project

The Australian Centre of Tropical Freshwater Research (ACTFR) has been commissioned to undertake this project by the Department of Environment and Resource Management (DERM). The background information presented here is extracted from the Scope and Background documentation prepared by DERM.

The *Chemical Usage (Agricultural and Veterinary) Control Act 1988* was recently amended to introduce new mandatory requirements for the use of herbicides containing atrazine, diuron, ametryn and hexazinone when undertaking an agricultural Environmentally Relevant Activity (agricultural ERA). The ERAs defined in this legislation are grazing and sugarcane farming in the Wet Tropics, Burdekin Dry Tropics and Mackay Whitsunday catchments.

The new mandatory requirements apply to the preparation and use of these chemicals on cane farms in these catchments. One of the new requirements is "*Do not use diuron, ametryn or hexazinone within 20m of all water bodies".* This requirement may be difficult for some growers to meet. The *Chemical Usage (Agricultural and Veterinary) Control Act 1988* also states that a person carrying out an agricultural ERA must comply with the above mandatory requirement or they can choose to meet the above condition through an alternative management practice in an ERMP (also known as a Chemical ERMP) that provides the same reduction in the risk of herbicide run-off. A Suggested Alternative Management Option to the above listed mandatory requirement is:

At the time of applying herbicide products that contain ametryn, diuron or hexazinone, a minimum 5m wide Effective Vegetated Treatment Area (EVTA) adjacent to the crop at the run-off points will be put in place.

Industry representation has identified that farms within the Babinda Community Drainage Scheme (BCDS) are unable to comply with the 20m buffer and are also resisting the installation of 5m EVTAs since they are already using a community drainage scheme that they say results in the same reduction of herbicides in run-off.

Additionally, in correspondence to industry representatives, DERM agreed that *compliant* Community Drainage Schemes would be recognised as an equivalent to the alternative EVTA option.

The Department has found that little research has been completed in Australia on the effectiveness of community drainage schemes, which consist of sub-surface mole drains, agricultural pipes, constructed wetlands and surface paddock and community drains. There is also no quality assurance standards associated with these schemes. Sampling of herbicide content at the beginning and the end of each of these types of drains/structures, where water exits the structure and enters a natural water body, has not been completed. This is the preferable method of proving that the Babinda community drainage scheme is able to fulfill the mandatory requirement of the legalisation and therefore be deemed compliant.

Prior to any sampling being undertaken, DERM has contracted ACTFR to provide an information paper and experimental design (Stage 1) to answer the following specific questions, which will provide DERM with the information needed to assess the compliance of this drainage system:

- How do ametryn, atrazine, hexazinone and diuron move and transform in sub-surface drains, vegetated surface drains and constructed wetlands (infrastructure typical of the BCDS) using the practices typical of these landholders, within the drainage area consisting of the soils similar to those of the BCDS, in low flow events and before they reach a natural water body?
- What is the effectiveness of a 3m, 5m, (possibly 10m) and 20m effective vegetated treatment area (EVTA) in mitigating the impact of run-off of the above herbicides active constituents (and as a second priority, mitigating the impact of nutrient and sediment run-off) in low flow events on soils of the BCDS?
- What would be a suitable experimental design to test the effectiveness of EVTAs compared to the BCDS components in the effective trapping/removal of the above herbicide active constituents/metabolites from low flow discharges off sugarcane fields?

The contract consists of two stages:

- Stage 1 – Provision of an Information Paper and survey design; and
- Stage 2 – Implementation of survey. The Department reserves the right to terminate the contract upon completion of the proof of concept at its sole discretion.

This draft report forms the basis of the product for Stage 1. It is noted that DERM convened a workshop in October 2010 which provided guidance for completion of this Stage 1 report.

Definitions

- **Effective Vegetated Treatment Area (EVTA):** Grassed area with widths of either 3m, 5m, 10m or 20m, of flat (<2% slope), un-compacted (i.e. no evidence of heavy vehicle or machinery use in muddy conditions) permeable soil, vegetated with at least 90% grass cover, between 10 - 15 cm high.
- **Low flow event:** Runoff up to the level where it is directed by the furrow (i.e. the water cannot flow in a different direction to the furrow because it has not overtopped the furrow)
- **Sub-surface drains:** Mole drains and agricultural pipes

2. Introduction

2.1 Sugar industry in Queensland

Sugarcane is grown along the Queensland coast from the Gold Coast to Mossman and specifically in the Great Barrier Reef (GBR) catchment area (Figure 2.1) from Maryborough to Mossman (Figure 2.2). Within the GBR catchment, sugarcane is grown under various levels of irrigation depending on local weather conditions. Table 2.1 (from Brodie 2007) shows the area of sugarcane under irrigation in each mill area, with approximately 15 per cent of this 520 000ha under fallow, and not fertilised (T. Wrigley, CANEGROWERS 2006, pers. comm.).

Table 2-1. Area of sugarcane land use in Queensland in 1999 (Dwyer, unpublished). Source: Brodie (2007).

Mill area	Cane production area (ha)	Percentage irrigated	Area irrigated (ha)
Mossman	15 356	27	4 146
Tablelands	6 712	100	6 712
Mulgrave	18 740	5	937
South Johnstone	20 523	13	2 668
Babinda/Mourilyan	29 015	0	0
Tully	29 302	0	0
Herbert[1]	68 004	15	10 201
Burdekin	84 004	100	84 004
Proserpine[2]	24 716	89	22 000
Mackay[3]	98 324	70	68 827
Sarina	22 398	36	8 063
Bundaberg[3]	53 003	100	53 003
Isis	19 102	88	16 810
Maryborough	15 493	47	7 282
Moreton	9 828	0	0
Rocky Point[4]	6 043	2.3	139
TOTALS	520 563	(average) 54.7	284 792

Figure 2.3 shows the areas of sugarcane cultivation in the Wet Tropics Region. As can be seen from Table 2.1 sugarcane cultivation in the Babinda area and generally in the Russell-Mulgrave Basin is totally rainfed with no irrigation. Babinda is included in the sugar industry's northern production region which spans from Mossman to Tully and includes seven mills including the Babinda Mill. In 2008, ~625,000 tonnes of cane and ~100,000 tonnes of sugar were harvested from ~8,000 hectares (Canegrowers, 2008). The CCS for 2008 was 12.4 (compared to 10.11 in 1998) which was the lowest in the northern production region (Tully for example was 13.0).

Figure 2-1. The Great Barrier Reef Catchment Area.

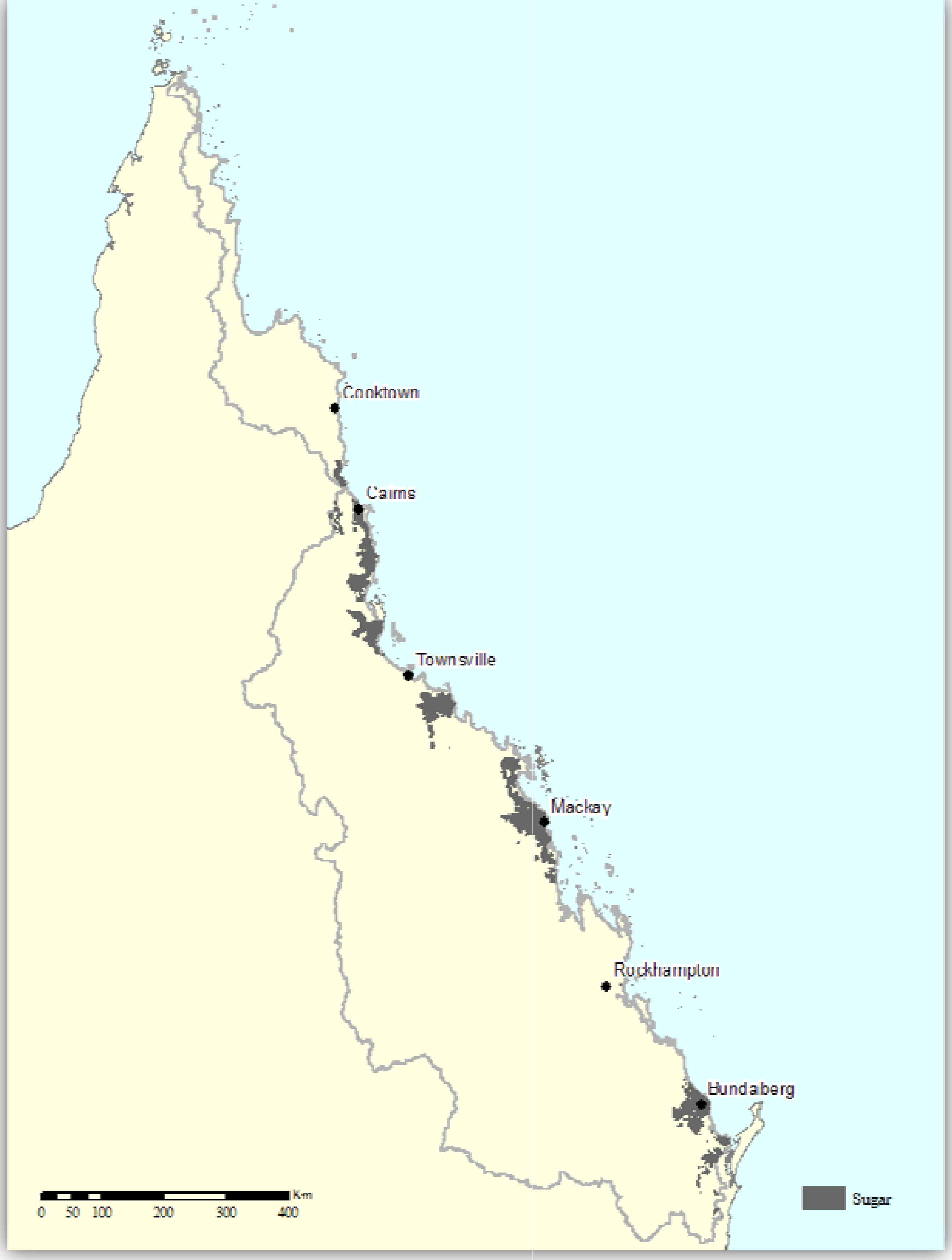


Figure 2-2. Sugarcane growing areas in the Great Barrier Reef Catchment area. Source: QLUMP (1999).

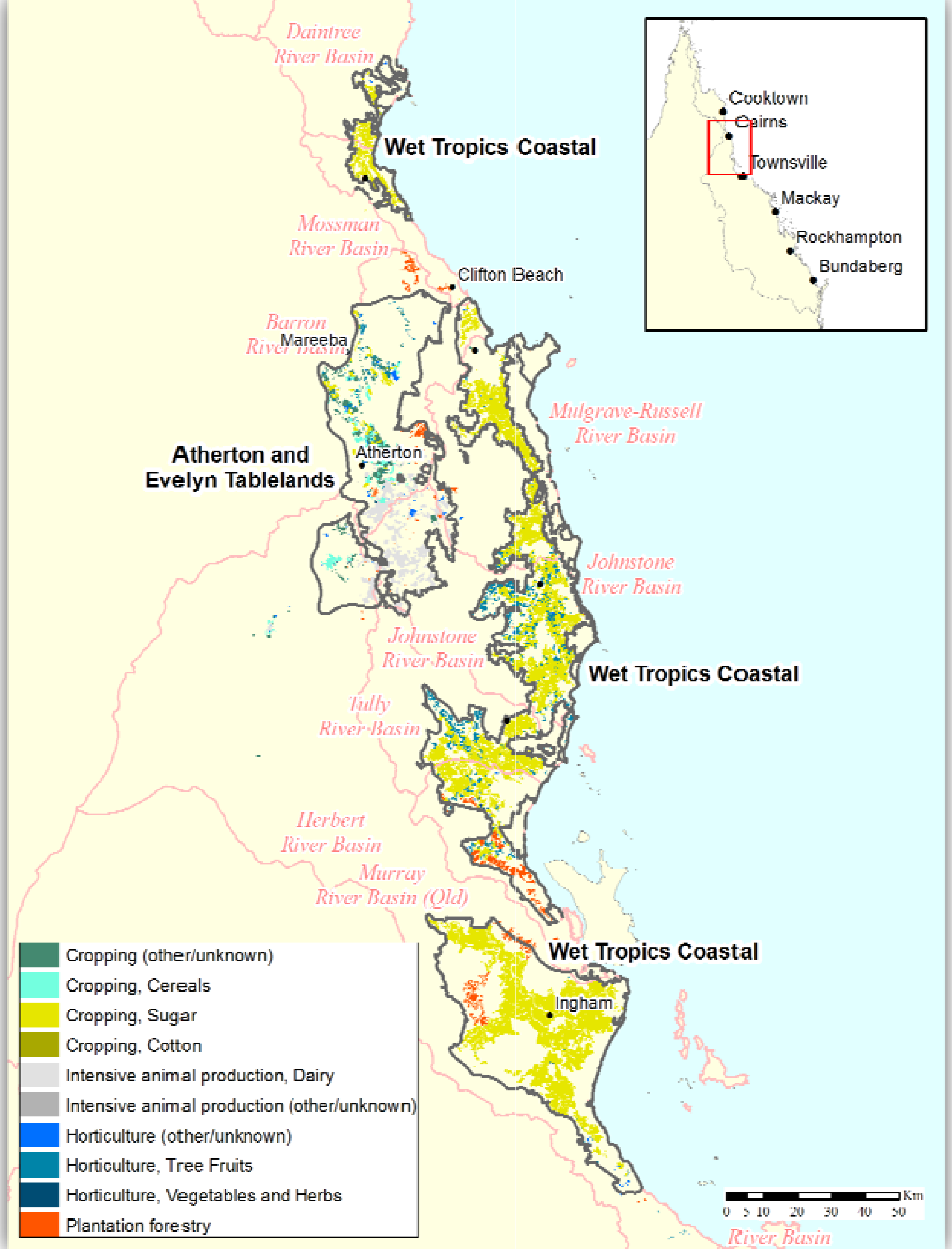


Figure 2-3. Wet Tropics Coastal area land use. Source: QLUMP (1999); Johnstone catchment QLUMP (2004).

2.2 The Russell-Mulgrave area

In the Russell-Mulgrave Basin sugarcane is grown on the coastal plain (Figure 2.4). This area is the wettest area in Australia with high runoff to rainfall ratios and frequent major ‘flashy’ runoff events (see Section 6). Major discharge from the combined Russell and Mulgrave Rivers occurs multiple times per year. The discharge causes frequent flood plumes (Figure 2.5) to form in the GBR lagoon containing the pollutants lost from the land and dispersing them widely in the GBR (Devlin and Brodie 2005). The plumes from the combined Wet Tropics rivers, with their entrained pollutants, disperse completely throughout the inner shelf of the GBR and regularly into the mid and outer shelves as well and occasionally into the Coral Sea beyond the GBR (Figure 2.6).

There have been considerable impacts from land-sourced pollution in this region on coral reefs and marine ecosystems generally. Reefs are degraded compared to those in regions where rivers are less polluted e.g. Cape York (Fabricius et al. 2005; Devantier et al. 2006). Nutrient discharge (from fertiliser losses and soil erosion) has been shown to be linked to outbreaks of the crown of thorns starfish, with the Russell-Mulgrave being a critical river in this situation (Brodie et al. 2005; Fabricius et al. 2010). Inner shelf waters of the GBR are also considered to be eutrophic in this region associated with nutrient discharge at some times of the year (Brodie et al. 2007; Brodie et al. 2011).

Figure 2-4. Sugarcane in the Russell catchment. Source: Google Earth, downloaded October 2010.

Figure 2-5. Example of a flood plume at the mouth of Russell-Mulgrave River. Source: GBRMPA.

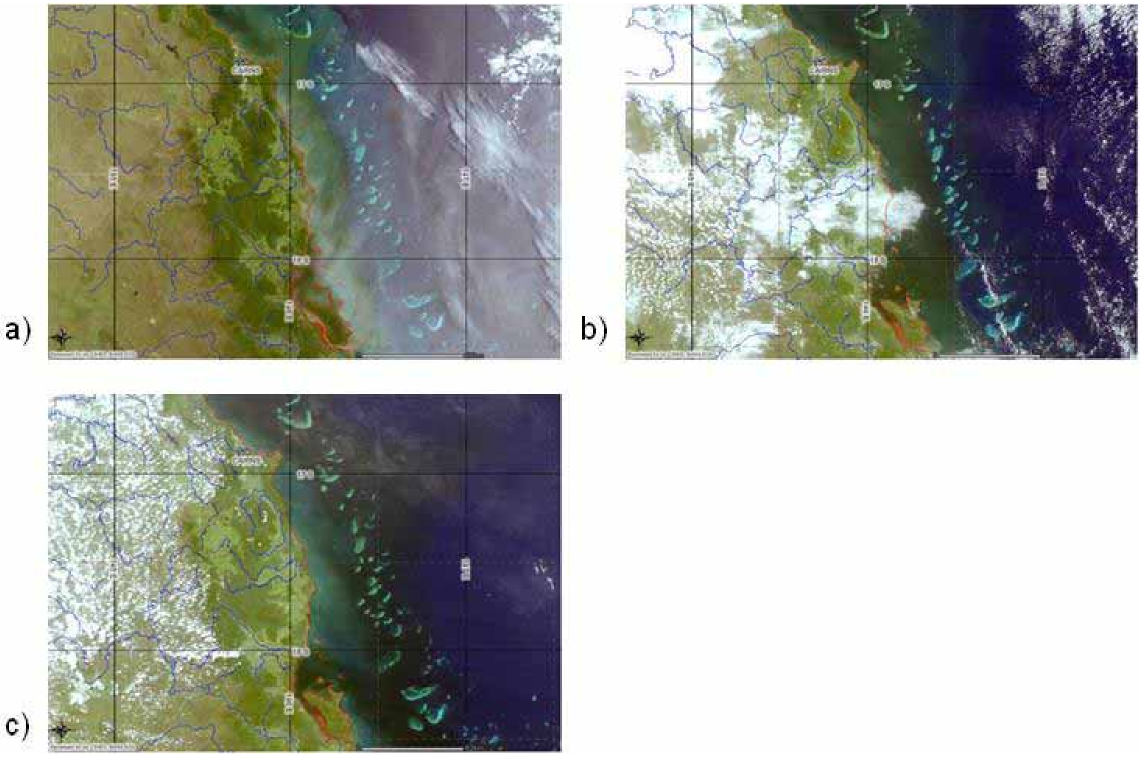


Figure 2-6. Flood plumes from Wet tropics Rivers on the (a) 9th, (b) 11th and (c) 13th February, 2007 moving from inner shelf waters on the 9th to the Coral Sea by the 13th February.

2.3 Herbicide application and types of usage and management practices

The sugar industry is implementing a variety of improved management practices in order to address concerns of potential losses of herbicides to the GBR. These practices include improvements in application methodologies, timing of applications as well as a move towards less environmentally persistent products. The main principle of improved weed management is to control weeds early in the crop cycle – particularly during the fallow period when cheaper and less persistent products such as glyphosate and paraquat can be used. When this principle is combined with other improved farming practices such as minimum tillage and planting operations and the use of legume fallow crops, weed seed germination is greatly reduced when compared to conventional tillage-based systems. This means there is less of a need to rely on residual herbicides due to decreased weed pressure.

One of the methods used to control weeds using less residual herbicide than methods such as boom sprayers (Figure 2.7) is a shielded sprayer (Figure 2.8). Shielded sprayers utilise a shroud to cover spray nozzles in the furrow – meaning that a contact herbicide such as glyphosate or paraquat can be used with minimal potential for crop damage. If the weed pressure warrants the use of a residual herbicide, this can be band sprayed over the crop area and not the furrow, thus minimising total product use over the paddock by up to 60% compared to boom sprayers. The use of such tools allows farmers to achieve both cost savings and environmental improvements while maintaining appropriate levels of weed control.

Figure 2-7. Boom spraying of herbicides on a sugarcane farm. Photo: B. Masters

Figure 2-8. The control of weeds using shielded sprayers is being encouraged as an improved management practice in the sugar industry. This approach targets weeds early, reduces the total amount of herbicide applied to the paddock, uses products less susceptible to runoff (e.g. glyphosate), and reduces the need for residual herbicide control. Photo: B. Masters

2.4 The GBR and herbicide concerns

Prior to the original work by Haynes *et al.*(2000) on the effects of diuron exposure to seagrass species, there was virtually no information on the impact of these herbicides to relevant marine plant species in the GBR lagoon. Over the last 10 years, there have been several laboratory-based studies on the acute (short term) effects of the commonly detected herbicides on species of seagrass (Haynes, Ralph et al. 2000; Ralph 2000), mangroves (Bell and Duke 2005), corals (Jones 2005; Negri, Vollhardt et al. 2005) and algae (Seery, Gunthorpe et al. 2006; Magnusson, Heimann et al. 2008). All of these studies use the pulse amplitude modulation chlorophyll fluorescence (PAM) technique which measures the effective quantum yield of the photosystem of the target plant species. The PAM method has the capacity to measure the lowest concentration that a particular herbicide will have a 'negative effect' on the plant species through its ability to photosynthesise; this measurement is known as the 'lowest observable effects concentration' or LOEC. The data from the grab samples taken from the river water plumes show that some concentrations exceed the LOEC measured for diuron (and to a lesser extent atrazine) on many of the plant species of the GBR (Lewis, Brodie et al. 2009). However, the laboratory experiments showed that many of the plant species appeared to fully recover once removed from herbicide exposure, although certain species also did not display full signs of recovery. This finding implies that, at least temporarily, there are negative effects to some plant species (e.g. seagrass, coral zooxanthellae) from herbicide exposure in the GBR lagoon.

Other studies have examined the effects of herbicide exposure in combination with other potentially relevant environmental influences such as seawater temperature, salinity and sedimentation. One such study showed that diuron attached to sediment particles can produce an enhanced effect on the sedimentation stress on crustose coralline algae (Harrington, Fabricius et al. 2005). Another study examined longer term effects on the exposure of diuron on the reproductive potential of corals (Cantin, Negri et al. 2007).

3. Overview of the Babinda Community Drainage Scheme

The Babinda Community Drainage Scheme is located in the Russell Mulgrave catchment, near the township of Babinda south of Cairns (Figure 3.1). The boundaries of the Babinda Community Drainage Scheme and Matthews Road Drainage Area are shown in Figure 3.2.

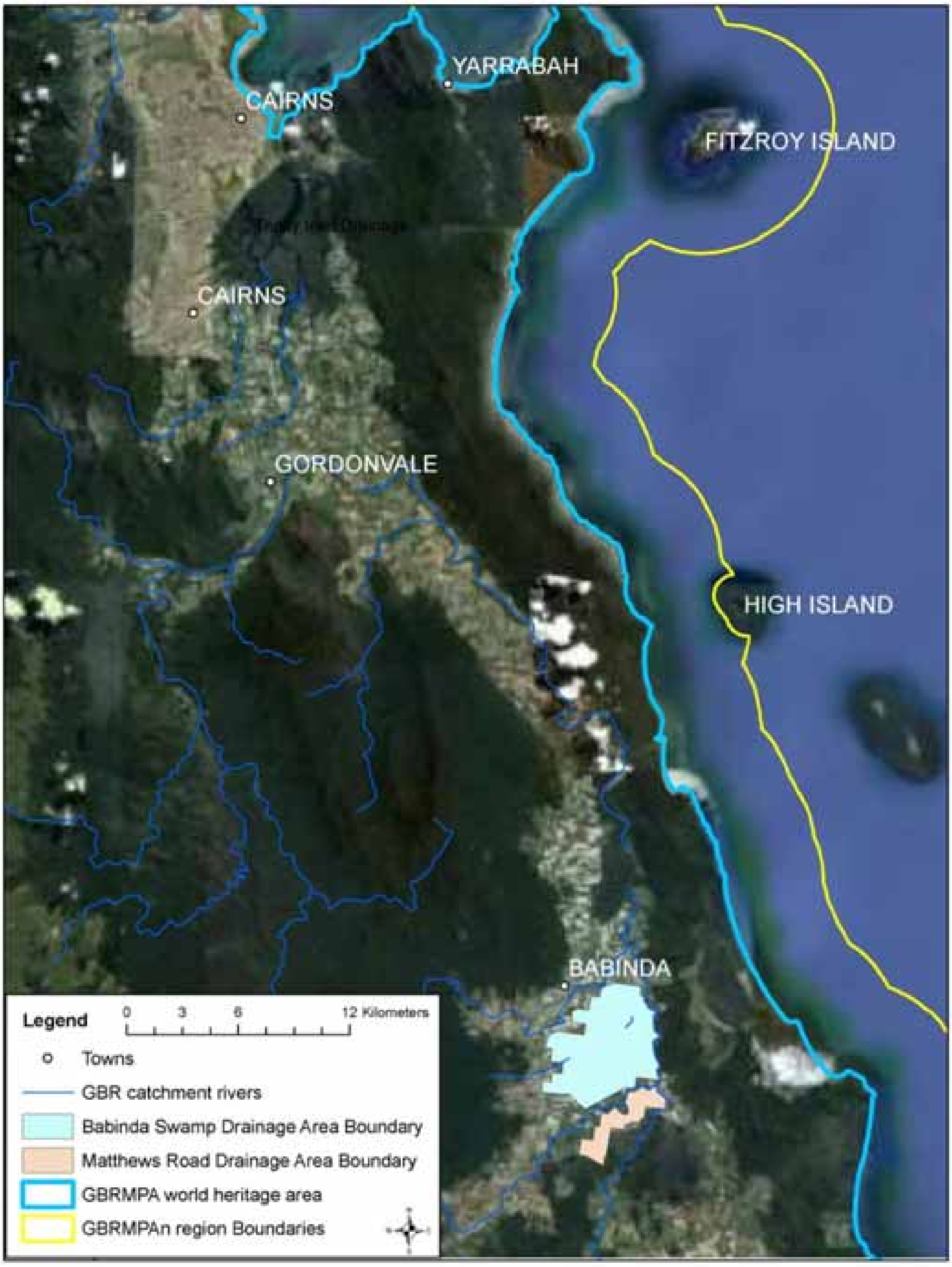


Figure 3-1. Location of the Babinda Community Drainage Scheme in the Russell Mulgrave catchment.

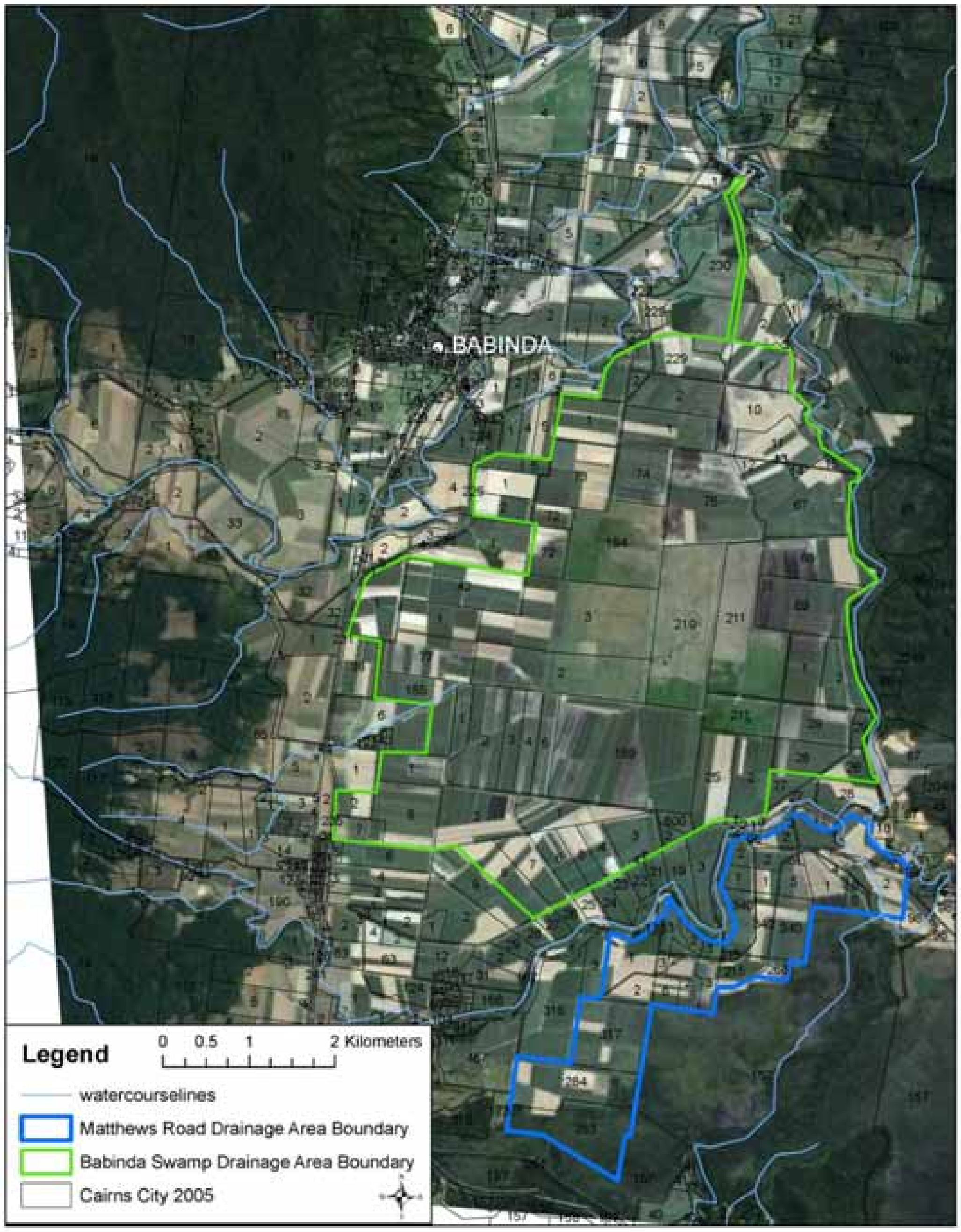
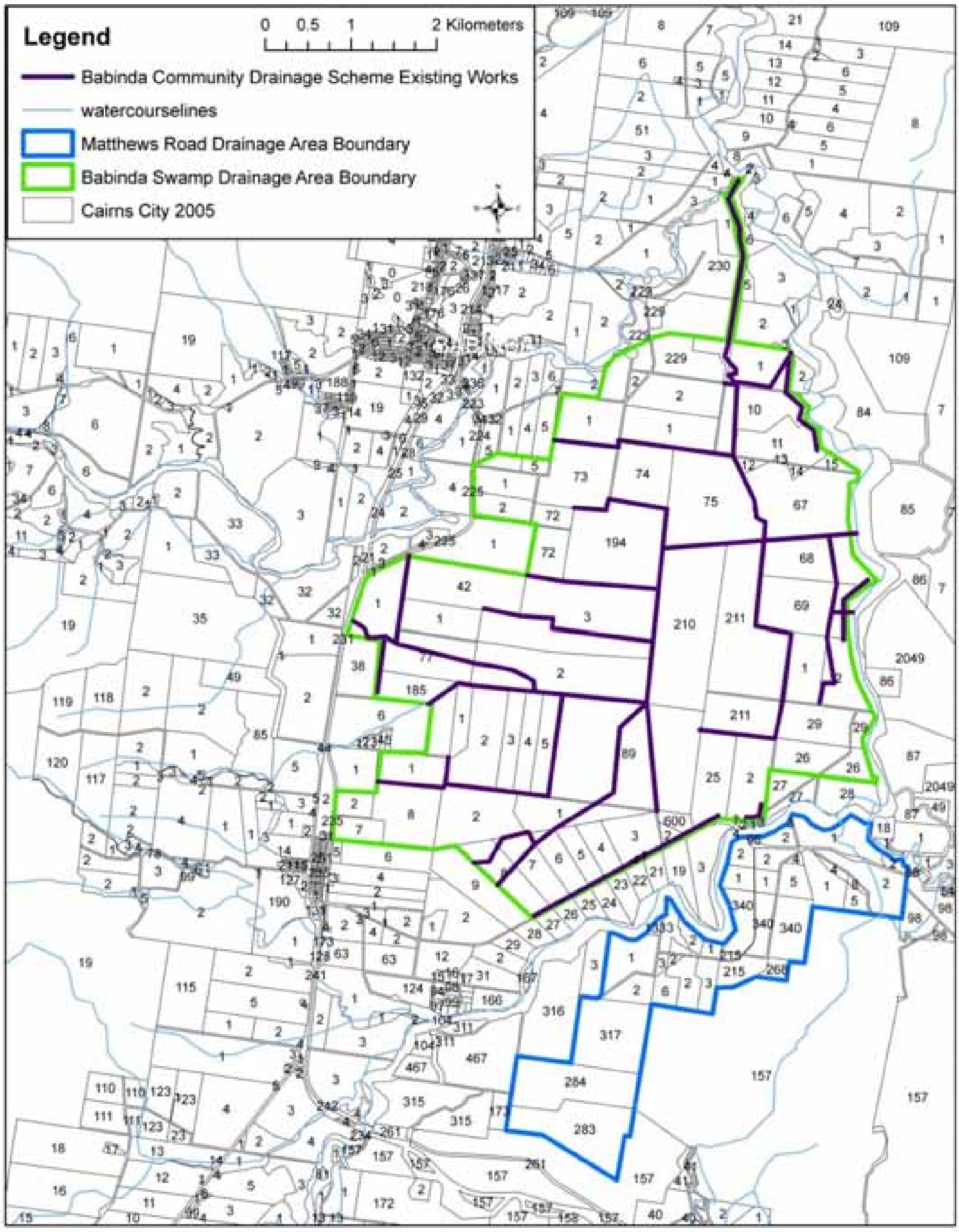


Figure 3-2. Boundaries of the Babinda Community and Matthews Road Drainage Schemes.

A comprehensive drainage map of the area does not presently exist, and part of this project is to develop a map of all of the drainage components of the Scheme including sub-surface drains, surface paddock drains, surface community drains and constructed wetlands. A map showing the primary drainage lines has been prepared from information obtained from the Drainage Boards and is shown in Figure 3.3. However, this information has not been ground truthed and the local consultation necessary to complete this component has not been able to be arranged as yet, pending the outcomes of the upcoming workshop to be convened by DERM.



Important note: This map has been derived from a map provided by the Drainage Board but has not been ground-truthed or checked with the Board.

Figure 3-3. Current knowledge of drainage infrastructure within the Babinda Community Drainage Scheme.

4. Properties of relevant herbicides

Physical and chemical properties of pesticides affect pesticide behaviour and transport. Some pesticides are highly adsorbed to soil particles. Because properly designed buffers are effective in trapping eroded sediment, runoff losses of this kind of pesticide have been consistently reduced by buffers. However, a number of modern pesticides are only moderately adsorbed to soil particles and are carried with runoff primarily in the dissolved phase. To be effective in trapping this type of pesticide, buffers must slow runoff and increase infiltration so that pesticides can be trapped and degraded in buffer soil and vegetation. Many studies have demonstrated pesticide trapping efficiencies of 50 percent or more for this type of pesticide, provided that sheet flow, not concentrated flow, occurs.

4.1 Chemical properties

The chemical properties and toxicity (Table 4.1) of the four herbicides of interest have been compiled using the ‘Footprint’ Pesticide Properties Database (http://sitem.herts.ac.uk/aeru/footprint/en/index.htm) while common product names and method of application were obtained using the APVMA’s Public Chemical Registration Information System (PUBCRIS: http://services.apvma.gov.au/PubcrisWebClient/welcome.do). These herbicides all have a similar mode of action (i.e. inhibit photosynthesis) but have different properties which influence their offsite transport. Hexazinone is by far the most soluble of the four herbicides although all four can be applied either through granular or liquid formulations. Both atrazine and hexazinone are weak bases while ametryn is weakly acidic (pKa). All four herbicides have reported soil half lives which range from ~30 days up to ~100 days; this variability is largely due to different climate regimes (temperature, solar radiation) as well as the composition of the soil (e.g. organic content, mineralogy, microbial communities, pH, oxidation levels). Moreover, all four herbicides are moderately to highly persistent in water. All four herbicides are preferentially hydrophilic (i.e. transported in dissolved phase), although some diuron can adsorb onto sediments and be transported in the particulate phase (K_{OC}: ACTFR, unpublished data).

4.2 Toxicity

The four herbicides have relatively low toxicity to mammals and birds, low-moderate toxicity to fish, aquatic invertebrates and crustaceans and moderate-high toxicity to aquatic plants and algae (relative to concentrations measured in waterways of the Great Barrier Reef catchment area) (Table 4.1). In addition, marine plants of the Great Barrier Reef (seagrass, coral symbionts, phytoplankton, algae) are highly sensitive to all four herbicides, and in particular diuron (Table 4.2).

Table 4-1. Chemical properties and toxicity of the four herbicides of interest.

Active ingredient	Diuron	Atrazine	Hexazinone	Ametryn
Common product names	Diurmax, Diurex, Velpar K4, Krovar, Striker, Zee Uron, Dethrone, Vertex, Diuron	Gesaprim, Atramax, Atradex, Nutrazine, Gesapax combi, AC AXIS 900 WG, Tarazine, Prozine, Atraphos, Atrazine	Velpar, Velmac, Velchem, Vertex, Bobcat Combi, SugarHex, Hexon, Grunt, Grandpar K4, Hexmac, Dymac, Dethrone, Hexazinone	Primatol Z, Reflex, Gesapax combi, Viking, Amesip Krismat, Ametrex, Amigan, Ametryn
Chemical group	Phenylurea	Triazine	Triazinone	Triazine
Mode of action	Systemic, absorbed via roots, acts by strongly inhibiting photosynthesis	Selective, systemic action with residual and foliar activity. Inhibits photosynthesis (photosystem II)	Non-selective with contact action, absorbed through the roots and foliage of plants. Inhibits photosynthesis (photosystem II)	Selective, systemic absorbed through foliage and roots. Inhibits photosynthesis (photosystem II)
Solubility - In water at 20°C (mg l^{-1})	35.6	35	33000	200

Active ingredient	Diuron	Atrazine	Hexazinone	Ametryn
Bulk density (g ml^{-1})/Specific gravity	1.5	1.23	1.25	1.18
Dissociation constant (pKa) at 25°C	No dissociation	1.7	2.2	10.07
Soil degradation (days) (aerobic) - range	75 - 90	29 - 75	90 - 105	37 - 60
Aqueous photolysis DT50 (days) at pH 7	43	2.6	56	N/A
Aqueous hydrolysis DT50 (days) at 20°C and pH 7	Stable	86	56	Stable
Koc - Organic-carbon sorption constant (ml g^{-1})	1067	100	54	316
Metabolites (breakdown products)	1-(3,4-dichlorophenyl)-3-methylurea; 3,4-dichlorophenyl urea; 3,4-dichloroaniline	6-deisopropyl atrazine; deethylatrazine; 2-hydroxyatrazine	3-(4-hydroxycyclohexyl)-6-(dimethylamino)-1-methyl-1,3,5-triazine-2,4(1H,3H)dione; 3-cyclohexyl-6-(methylamino)-1-methyl-1,3,5-triazine-2,4-(1H,3H)-dione; 3-(4-hydroxycyclohexyl)-6-(methylamino)-1-methyl-1,3,5-triazine-2,4(1H,3H)-dione	deethyl ametryne
Toxicity				
Mammals - Acute oral LD50 (mg kg^{-1})	437	1869	1690	1160
Birds - Acute LD_{50} (mg kg^{-1})	1104	4237	2258	5620
Fish - Acute 96 hour LC50 (mg l^{-1})	6.7	4.5	320	5
Fish - Chronic 21 day NOEC (mg l^{-1})	0.41	2	N/A	N/A
Aquatic invertebrates - Acute 48 hour EC_{50} (mg l^{-1})	5.7	85	85	28
Aquatic invertebrates - Chronic 21 day NOEC (mg l^{-1})	0.096	0.25	50	0.32
Aquatic crustaceans - Acute 96 hour LC_{50} (mg l^{-1})	1.1	1	N/A	1.7
Aquatic plants - Acute 7 day EC_{50}, biomass (mg l^{-1})	0.0183	0.019	0.072	0.01
Algae - Acute 72 hour EC_{50}, growth (mg l^{-1})	0.0027	0.059	0.0145	0.0036
Algae - Chronic 96 hour NOEC, growth (mg l^{-1})	N/A	0.1	N/A	N/A

Table 4-2. Summary data on the effects of herbicides to marine plants of the Great Barrier Reef.

Active ingredient	Diuron	Atrazine	Hexazinone	Ametryn
Seagrass LOEC (μg l^{-1})	0.10 μgL^{-1} (2)	10 μgL^{-1} (1)	N/A	N/A
Coral symbionts LOEC (μg l^{-1})	0.30 μgL^{-1} (3,4)	3.0 μgL^{-1} (3,4)	3.0 μgL^{-1} (3)	0.30 μgL^{-1} (3)
Coral symbionts EC$_{50}$ (μg l^{-1})	2.3 μgL^{-1} (4)	45 μgL^{-1} (3)	8.8 μgL^{-1} (3)	1.7 μgL^{-1} (3)
Microalgae LOEC (μg l^{-1})	0.10 μgL^{-1} (9)	1.1 μgL^{-1} (9)	0.07 μgL^{-1} (9)	N/A
Microalgae EC$_{50}$ (μg l^{-1})	2.06 μgL^{-1} (9)	14.2 μgL^{-1} (9)	2.4 μgL^{-1} (9)	N/A
Macroalgae EC$_{50}$ (μg l^{-1})	1.65 μgL^{-1} (6)	N/A	N/A	N/A
Diatoms	0.05 μgL^{-1} (5)	47 μgL^{-1} (8)	N/A	N/A
Crustose coralline algae EC$_{50}$ (μg l^{-1})	2.9 μgL^{-1} (7)	N/A	N/A	N/A

[1]Ralph (2000); [2]Haynes *et al.*(2000); [3]Jones and Kerswell (2003); [4]Jones *et al.* (2003); [5]Bengston Nash *et al.* (2005); [6]Seery *et al.* (2006); [7]Harrington *et al.* (2005); [8]Magnusson *et al.* (2008); [9]Magnusson *et al.* (In press).

4.3 Pesticides used in the sugar cane industry

Insecticides

Chlorpyrifos is the most commonly used insecticide in the sugar cane industry following the banning of organochlorines in 1987. This organophosphate may be applied by two separate techniques: a controlled release formulation (SuSCon®) and a spray emulsion (Lorsban®). The SuSCon® technique is used to control cane grubs while Lorsban® is used to control other pest insects (Cavanagh, 2003). Chlorpyrifos is commonly used between April and December. Other insecticides used in the sugar industry to control pest insects include carbaryl (sugarcane stem borer and controlling adult beetles, common usage from April to December), methamyl (sugarcane stem borer and controlling adult beetles, common usage from April to December) and imidacloprid, although these controls only form a relatively minor component compared to the chlorpyrifos formulations. Chlorpyrifos has not been completely effective in controlling the grey back grub and more recently some farmers have resorted to using imidacloprid (confidor) (E. Shannon pers comm., 2007).

Herbicides

Similarly to the organochlorine insecticides, herbicides were first used in the GBR catchments during the 1940's. 2,4-D was the first herbicide to be used in the Herbert catchment in 1948 while atrazine and diuron have been in use since 1959 (Johnson and Ebert 2000). These herbicides have remained among the preferred measures for weed control in the sugarcane industry and their use has increased considerably in response to expansions in this industry particularly during the 1980s (Johnson and Ebert 2000). In particular, atrazine use increased markedly in the Herbert Region since 1983 (Johnson and Ebert, 2000). Other popular herbicides include hexazinone, ametryn, asulam, fluroxypr, glyphosate, MCPA 500 and paraquat. Unfortunately, no data are available on the historical use of these additional herbicides in GBR catchments. Hamilton and Hayden (1996) provide the most up to date quantitative estimates of the major herbicides used in the sugar industry.

Many of the herbicides may be mixed together to target particular weeds; for example 2,4-D may be mixed with glyphosate (i.e. roundup) or paraquat for grass control (see Makepeace and Williams, 1986). In addition, specific herbicides are applied to target particular weeds (e.g. atrazine is used to control broadleaf weeds, while 2,4-D is used to control vines).

Fungicides

Methoxyethylmercuric chloride (MEMC) is used in the sugar industry to control “pineapple disease” in cane. Fungicide usage in cane is generally restricted to the time of planting (E. Shannon pers comm., 2007).

4.4 Pesticides measured in the GBR catchments

A number of monitoring programs in the GBR catchments and Marine Park incorporate sample collection for pesticide analysis. The following results have been reported as part of the Reef Plan/Reef Rescue Marine Monitoring Program between 2004 and 2010 and are relevant to the Study Area:

- Imidacloprid was the only insecticide detected in a grab sample collected from the mouth of the Russell Mulgrave Rivers in 2010 (Devlin, McKinna et al. 2010) while diazinon, chlorfenvinphos, fipronil and chlorpyrifos were all detected in passive samplers deployed at the mouth of the Russell Mulgrave Rivers in 2005 (Shaw et al., 2010).
- Diuron, atrazine, metolachlor and pendimethalin have been detected at the mouth of the Russell-Mulgrave Rivers in passive samplers deployed in 2004 and 2005 (Shaw, Furnas et al. 2010) while diuron, atrazine, hexazinone and ametryn were detected in passive samplers deployed in 2005/06 and 2006/07 (Kapernick, Shaw et al. 2007; Prange, Haynes et al. 2007). Diuron, atrazine and hexazinone have been detected in grab water samples taken in February 2010 (Devlin, McKinna et al. 2010). Passive samplers deployed off High Island and Normanby Island have detected diuron, atrazine, hexazinone, ametryn, simazine, tebuthiuron (Kapernick, Shaw et al. 2007; Prange, Haynes et al. 2007; Bartkow, Dunn et al. 2008).
- Propiconazole was the only fungicide detected in passive samplers deployed at the mouth of the Russell Mulgrave Rivers (Shaw, Furnas et al. 2010).

5. Herbicide use in the area

A requirement of the contract with DERM for this project is that herbicide use (specifically atrazine, ametryn, diuron and hexazinone) in the area is reported. Ideally, this would include application rates, application timing and compound type used. However, this component has not been progressed at this stage pending further consultation with DERM, industry representatives and the local community.

6. Drainage characteristics of the Study Area

The Russell Mulgrave catchments are located in the Wet Tropics region of the Great Barrier Reef catchments. This section provides an overview of the soils and hydrology of the area, relevant to the objectives of this project.

6.1 Soils

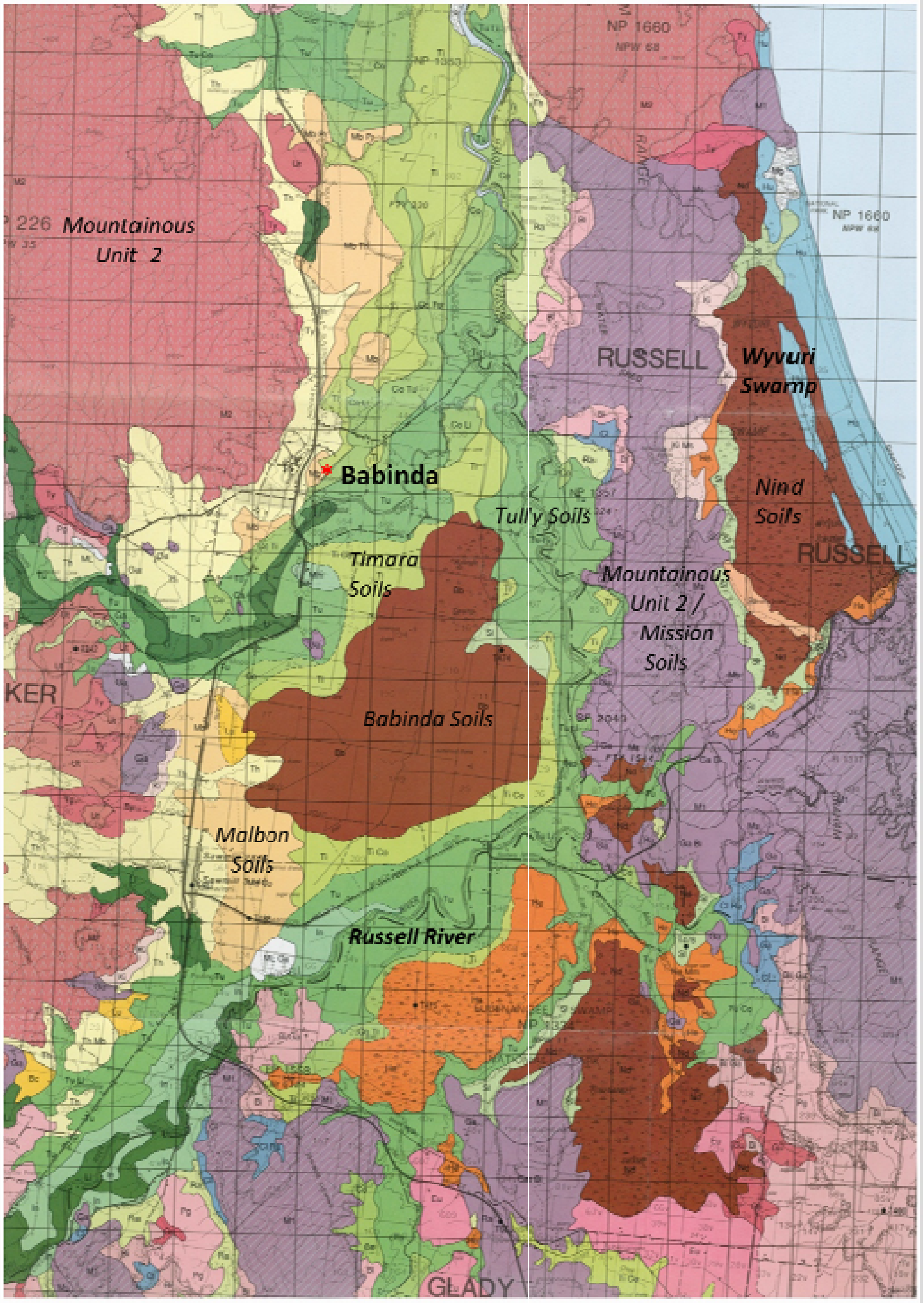
The soil descriptions and mapping units outlined in this report are based predominantly on the soil series recognized in Murtha et al. (1996). Soil classification terminologies follow those outlined in Northcote (1979), Stace et al. (1968) and Isbell (1993). Soil parent material in the mountainous uplands includes a range of granitics (mainly Utchee series: uniform or gradational textured, red structured soils formed in situ) and low grade metamorphics (mainly Galmara series: red, structured, uniform or gradational textured soils formed in situ). Soil geomorphology of the lowland country is very diverse and complex with a range of origins. Soil profile wetness, which is usually site dependent, is typically the overriding determinant of soil morphology in lowland areas (Murtha, Cannon et al. 1996). The map of soil types prepared by Murtha et al., (1996) is not available electronically, although a scanned image identifying the predominant soil types is shown in Figure 6.1 and described in Table 6.1.

Soil morphology in the immediate Babinda Community Drainage Scheme area consist of a mosaic of Tully series brown dermosols; ‘well drained soils formed on alluvium’, Timara series oxyaquic hydrosols; ‘poorly drained soils formed on alluvium’ and Babinda fibric organosols; ‘peats in freshwater swamps’ (Murtha, Cannon et al. 1996). Table 6.1 provides an outline of the major distinguishing features of these three soil series. The alluvial soils have been primarily formed from mixed deposits of granitic and metamorphic parent materials derived from upland areas. Murtha et al. (Murtha, Cannon et al. 1996) noted that the differentiation between well and poorly drained soils is somewhat arbitrary, with soil morphology in many areas forming a transition from one designation to another.

Under natural conditions the water table in the Babinda Swamp would have rarely fallen below 0.3m of the soil surface, and free water at the surface would have been common. The Babinda Swamp region has been drained and cleared for sugarcane cultivation and to a smaller extent improved pastures. Some sections of the Babinda Swamp have been subject to substantial surface shrinkage (sometimes over 1m) as a results of peat shrinkage following removal of water. Although substantial areas have been artificially drained, high water tables are maintained in the profile for much of the year, with water rarely dropping below a depth of one metre from the surface (Murtha, Cannon et al. 1996).

Table 6-1. Outline of the major distinguishing features of the dominant soil types in the Babinda Community Drainage Scheme area.

Series	Landform	Major distinguishing features
TULLY	Stream levees flood plain and terraces	Uniform to gradational texture profile, yellow, strongly structured, silty clay loam to silty clay textures
TIMARA	Backplain	Uniform or gradational textured soil, light grey upper B horizon, strongly structured, saturated for long periods each year
BABINDA	Peat Swamp	Highly organic soils formed from remains of sedges, pandanas and trees. Typically sapric surface layers (0.4-0.6m of completely decomposed, unrecognizable plant remains), underlain by fibric (very fibrous peat with unidentifiable plant remains).



Soil Descriptions:

Babinda: Fibric, Organosol; *Mission*: Red, Dermosol; *Malbon*: Brown, Kandasol; *Mountainous Unit 1*: Dominantly Galmara series with lesser Bicton and Bingil series; *Mountainous Unit 2*: Dominantly Utchee series with Tyson series on small alluvial fans included; *Nind*: Fibric Organosol; *Tully*: Brown Dermosol; *Timara*: Oxyaquic, Hydrosol

Figure 6-1. Dominant soil types in the Babinda area. Derived from Murtha et al. (1996).

6.2 Climate and Hydrology

This study focuses primarily on two watersheds within the broader Russell-Mulgrave catchment; the Russell River and Babinda Creek, both of which are the major drainage lines bounding the Babinda Drainage scheme area. The Russell-Mulgrave catchment in its entirety is one of the larger wet-tropics catchments in terms of area, rainfall and discharge to the Great Barrier Reef (GBR) lagoon (Furnas 2003). The Russell River and Babinda Creek are approximately 65 km and 22 km long, with catchment areas of approximately 560 km2 and 92 km2, respectively (Connolly, Pearson et al. 2007). Both systems drain the eastern escarpment of the Great Dividing Range, with much of the watershed of both systems dominated by the mountain massifs of Mt Bartle Frere, the highest mountain peak in Queensland, which reaches an elevation of 1622m and Mount Bellenden-Ker (1593m elevation). The Russell River and Babinda Creek catchments include a range of landforms, commencing in the mountain range through which the streams have cut deep valleys and gorges. Due to the topographic features of their upper catchments, these systems both traverse some of the greatest altitudinal ranges of any creek or river systems in Queensland. The streams descend rapidly down the steep, mountainous range before abruptly changing slope to flow across a flat and narrow coastal floodplain (Connolly, Pearson et al. 2007). As depicted in Figure 6.2, the combination of steep topography and the close proximity of both mountain peaks to the coastline (<25km) results in very rapid transit times between coastal rainfall and oceanic discharge (minimal residence times).

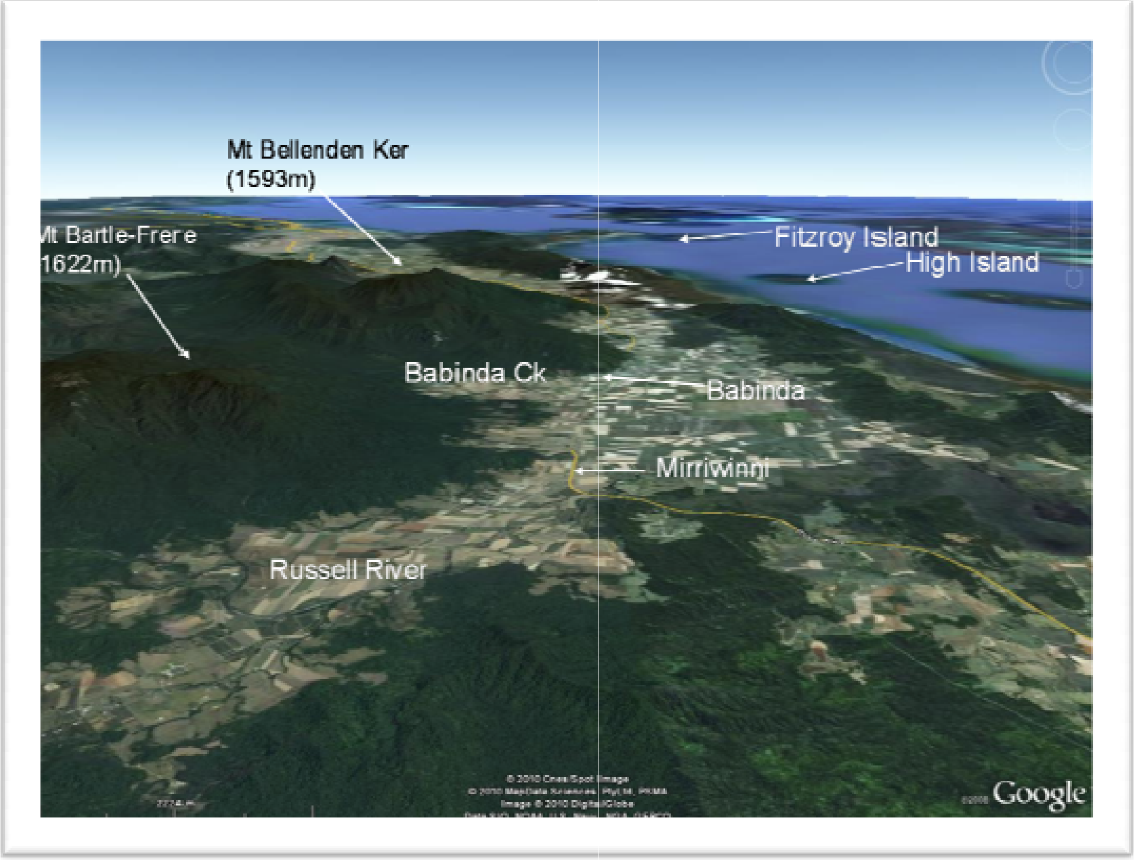


Figure 6-2. Topography of the Russell Mulgrave catchment area.

Rainfall

Regional climate in the study area is warm-humid. Average annual rainfall on the coastal plain in the Russell-Mulgrave catchment consistently exceeds 3000 mm/year. The Babinda Township competes with Tully for the highest annual average rainfall of any Australian town (at 3016 mm/year). Rainfall is seasonal with 60% falling in the summer wet season, occurring through the period December to March. Table 6.2 outlines the average and median monthly rainfall data and highest daily rainfall figures for each month over the period of record (1911 to present) registered at the Babinda Post Office climate station (BOM Station number 31004).

Table 6-2. Daily and Monthly Rainfall Data from Babinda Post Office (BOM site 031004).

Statistic	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	645.6	746.8	792.4	536.7	345.9	204.3	141.9	115.1	124	112.6	181.5	303
Median	561.8	654.4	668.5	449.6	328.3	179.6	130.7	66.9	92.7	72.1	126.7	196
Highest Daily	546.4	521	613.2	407.7	315	227.3	139.2	236.2	174.2	221.7	248	480.6

Because of orographic uplift effects associated with movement of warm, humid air from the coral seas, a substantial rainfall gradient occurs from the mountainous upper reaches of catchments through to the flatter coastal sections of the Russell-Mulgrave watershed. Rainfall can be substantially higher in the upper parts of catchments compared to that falling on lower floodplain environments. While no rain gauge exists on Mount Bartle-Frere, BOM rainfall data sourced from the nearby Mount Bellenden Ker Top Station (BOM Site 031141, data record: 1973-present) at the mountain summit documents an annual average rainfall in excess of 8,000 millimeters/year, making it the wettest meteorological station in Australia (Table 6.3). Bellenden-Ker Top Station has also recorded the highest rainfall in a calendar year of 12,461 mm (490.6 in) in 2000 and the highest rainfall in Australia for a calendar month of 5,387 mm (212.1 in) in January 1979. These climatic figures rank these coastal mountain ranges as one of the wettest places in Australia, if not the world. Figure 6.3 depicts annual rainfall isohyets for the study area. Peak annual rainfall occurs in association with higher terrain elevations, while annual rainfall diminishes rapidly westward of the coastal mountain range.

Table 6-3. Daily and Monthly Rainfall Data from Bellenden-Ker Top Station (BOM site 031141).

Statistic	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Mean	1012	1217	1338	1089	777.7	438.6	380.7	327.1	276.3	288.6	372	556.8
Median	769	963.4	1393	884.8	790.1	445	365.5	238.8	173.9	171	240.4	376.5
Highest Daily	780	410	470	490	354	290	205	277	370	175	630	468

While the data presented in Table 6.3 highlights a summer dominant rainfall pattern, Babinda Post Office rainfall data also highlights that significant falls (>100mm/day) can occur on the coastal plains at any time of year. Figure 6.4 outlines very recent rainfall patterns that highlight the intensity of rainfall that can occur outside of the typical December to March wet season period. Two >100mm/day and two >150mm day rainfall events occurred between 22/08/2010 and 04/10/2010, a period that also coincides with many on-farm management actions (fertilizer and pesticide application) with significant off-site water quality risk associations.

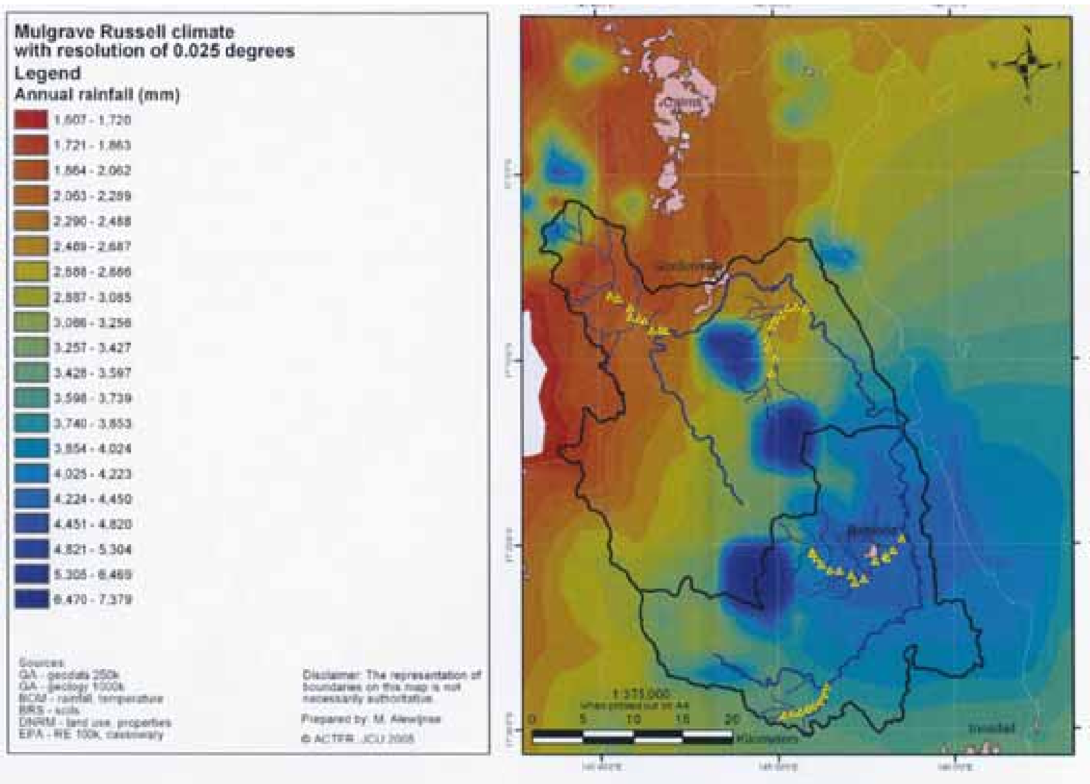


Figure 6-3. Russell-Mulgrave isohyet rainfall patterns.

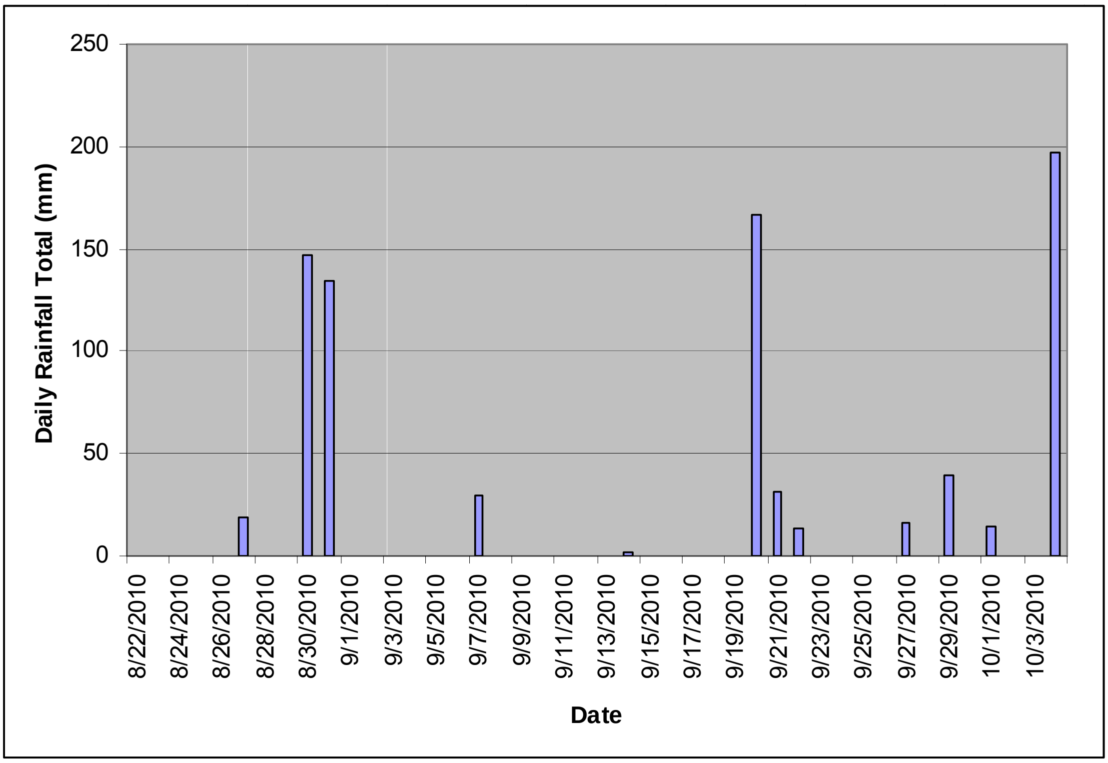


Figure 6-4. Daily rainfall totals (mm) from the Babinda Post Office BOM climate station for the period 22nd August- 4th October, 2010.

Local Stream Hydrology

No flow data is currently available to analyse the specific hydrological behaviour of drainage lines within the Babinda Drainage Scheme. To obtain a surrogate measure for likely streamflow characteristics of the drainage scheme, the stream gauging data record for the past 30 years were obtained from two DERM gauging stations close proximity to the Babinda Community Drainage Scheme. Gauging Station GS 111101D (Russell River at Bucklands) is located on the Russell River approximately 5km downstream of the Bruce Highway Bridge (Latitude 17:23 S; Longitude 145:58, catchment area 315 km^2). Gauging station GS111105A (Babinda Creek at the Boulders) currently operates on the upper section of Babinda Creek, and is located downstream from the Boulders near the base of the range (Latitude 17:21 S; Longitude 145:52, catchment area 39 km^2). Daily river height data was plotted in conjunction with Bureau of Meteorology flood classification (major and moderate floods) to highlight frequency and duration of local flood events in the Babinda area (Figure 6.5 and 6.6).

Daily river height data for both Russell River and Babinda Creek demonstrate that large flow events occur multiple times annually and that these large flows persist only for short periods of time. Data also show that although large flow events commonly occur during the typical December to March wet season period, minor and moderate flow events can occur throughout the year.

The amount of water available for run-off in a catchment is influenced by an array of factors such as rainfall intensity, surface slope, soil type and ground cover (Furnas, 2003). Due to a combination of hydrological variables (i.e. high annual rainfall-high intensity rainfall events and catchment topography) over 60% of average annual rainfall in the Russell-Mulgrave catchment is converted to surface runoff that leaves the basin (1968-1994 average run-off; Furnas, 2003). The small catchment areas and steep topography of streams such as the Russell River and Babinda Creek also make systems very responsive to high intensity rainfall events, with very rapid (‘flashy’) changes in river height associated with rapid transmission of floods through drainage systems.

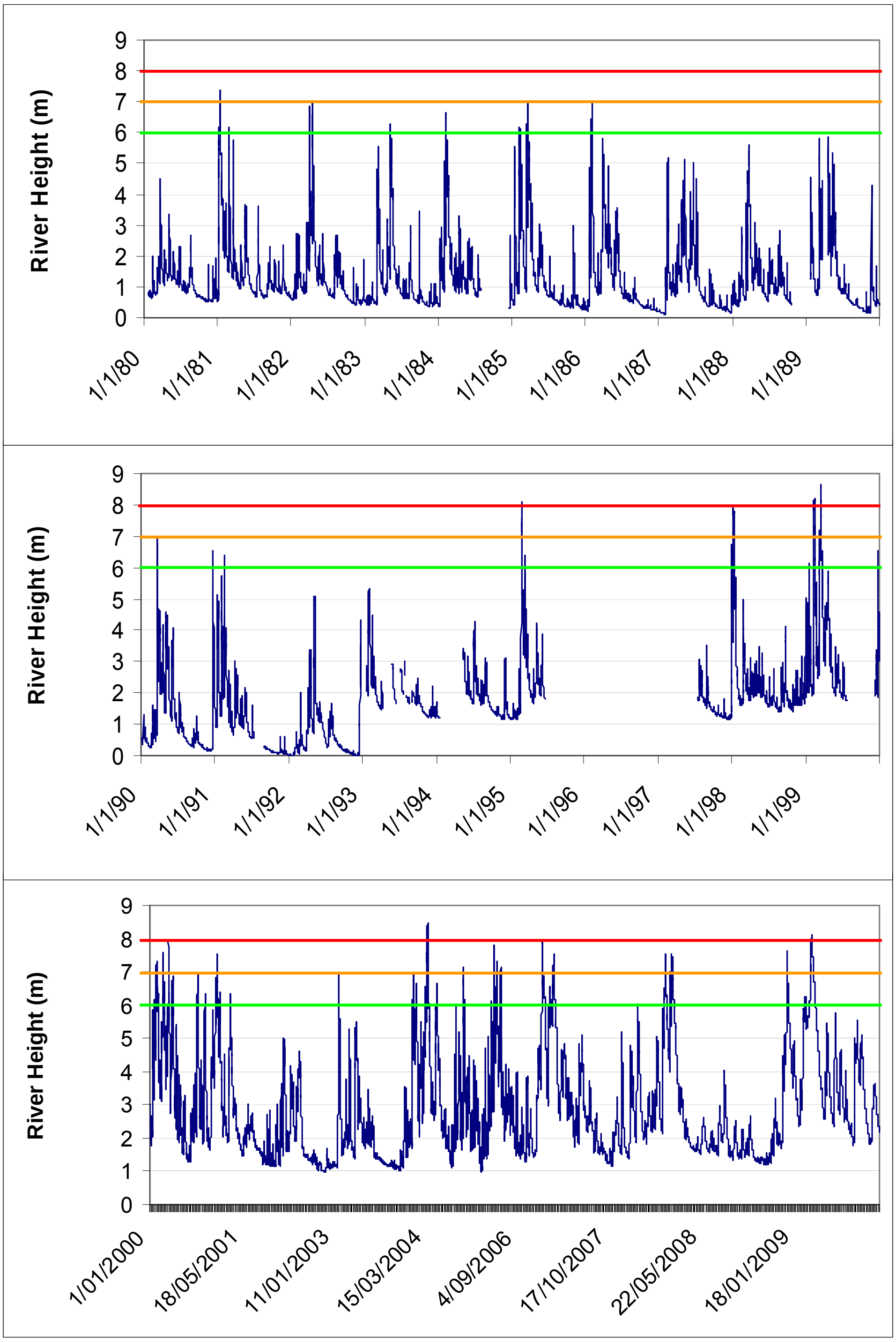


Figure 6-5. Daily river height data: Russell River at Bucklands (GS111101D; 1980-present).

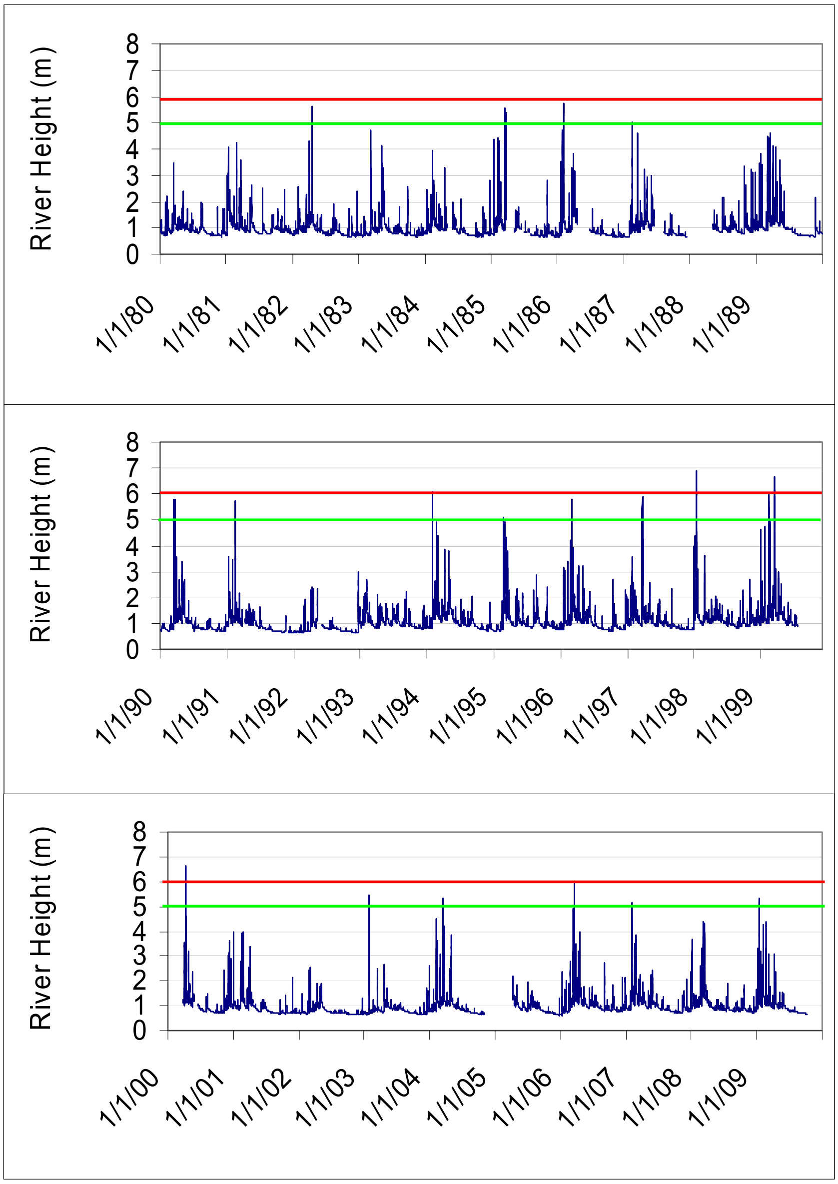


Figure 6-6. Daily river height data: Babinda Creek at the Boulders (GS111105A; 1980-present).

7. Effectiveness of constructed wetlands, vegetated strips and drains in trapping of herbicides (and suspended sediments and nutrients)

A key part of this project involves a literature review of the effectiveness of constructed wetlands, grassed vegetated areas (strips, riparian areas and drains) and sub-surface drains in effective trapping of herbicides (emphasis on PSII herbicides and their metabolites) lost from agricultural lands in low flow events. A comprehensive summary of the literature reviewed is tabulated in Appendix 1. The following section provides an overview of the findings relevant to this project.

7.1 Vegetated buffers for reducing pesticide loss in runoff

Vegetated areas including vegetated or grassed strips, riparian areas and drains and constructed wetlands can be used as ‘buffers’ in a systems approach to manage soil, water, nutrients, and pesticides for sustainable agricultural production, while minimising environmental impact. The buffers are usually grasses, situated down slope of cropping areas or animal production facilities to filter sediment and other pollutants from agricultural runoff.

Vegetated buffers can improve incoming runoff by changing the flow hydraulics, which increases the opportunity for infiltration of surface runoff, deposition of suspended solids, filtration of suspended sediment by vegetation, adsorption on soil and plant surfaces and absorption of soluble pollutants by plants (Smith, Melvin et al. 1992; Norris 1993; Parson 1994; Hatfield, Mickelson et al. 1995; Arora, Mickelson et al. 1996; Dawson 1997; Hairsine 1997; Loch, Espigares et al. 1999; Schmitt, Dosskey et al. 1999; Wong, Breen et al. 1999; Dosskey 2001; Jin and Römkens 2001; Seybold, Mersie et al. 2001; Boyd, Baker et al. 2003; Mickelson, Baker et al. 2003; Mickelson, Helmers et al. 2004; Selim, Naquin et al. 2004; Haarstad, Braskerud et al. 2005; Krutz, Senseman et al. 2005; Grismer, O'Geen et al. 2006; Rose, Sanchez-Bayo et al. 2006; Pätzold, Klein et al. 2007; Helmers, Isenhart et al. 2008; Budd, O’Geen et al. 2009; Gregoire, Elsaesser et al. 2009; Hoffmann, Kjaergaard et al. 2009; Poletika, Coody et al. 2009; Fox, Muñoz-Carpena et al. 2010).

The effectiveness of buffers is determined by:

- Structure/species of vegetation (Sullivan ; Smith, Melvin et al. 1992; Norris 1993; Parson 1994; Hairsine 1997; Lowrance, Vellidis et al. 1997; Patty, Réal et al. 1997; Snyder 1998; Loch, Espigares et al. 1999; USDA 2000; Rankins Jr, Shaw et al. 2001; Reungsang, Moorman et al. 2001; Seybold, Mersie et al. 2001; Mersie, Seybold et al. 2003; Lin, Lerch et al. 2004; Mickelson, Helmers et al. 2004; Bouldin, Farris et al. 2005; Krutz, Senseman et al. 2005; Polyakov, Fares et al. 2005; Rankins Jr, Shaw et al. 2005; Fares and Ryder 2006; McKergow, Prosser et al. 2006; Popov, Cornish et al. 2006; Rose, Sanchez-Bayo et al. 2006; Pätzold, Klein et al. 2007; Reichenberger, Bach et al. 2007; Liu, Zhang et al. 2008; Moore, Denton et al. 2008; Otto, Vianello et al. 2008; Rose, Crossan et al. 2008; Budd, O’Geen et al. 2009; Yuan, Bingner et al. 2009).
- Length/gradient/shape of runoff area (width and slope) (Smith, Melvin et al. 1992; Norris 1993; Srivastava, Edwards et al. 1996; Hairsine 1997; Loch, Espigares et al. 1999; Jin and Römkens 2001; Boyd, Baker et al. 2003; Mickelson, Baker et al. 2003; Krutz, Senseman et al. 2005; Polyakov, Fares et al. 2005; Blankenberg, Braskerud et al. 2006; Grismer, O'Geen et al. 2006; Popov, Cornish et al. 2006; Pätzold, Klein et al. 2007; Liu, Zhang et al. 2008; Moore, Denton et al. 2008; Pinho, Morris et al. 2008; Budd, O’Geen et al. 2009; Yuan, Bingner et al. 2009; Fox, Muñoz-Carpena et al. 2010; Sabbagh, Fox et al. 2010).
- Length/gradient/slope upstream (Smith, Melvin et al. 1992; Norris 1993; Hairsine 1997; Boyd, Baker et al. 2003; Liu, Zhang et al. 2008; Page, Dillon et al. 2010),
- Rate of surface water flow (Norris 1993; Barling and Moore 1994; Dawson 1997; Hairsine 1997; Loch, Espigares et al. 1999; Dosskey 2001; Jin and Römkens 2001; Boyd, Baker et al. 2003; Mersie, Seybold et al. 2003; Stearman, George et al. 2003; Brønnum, Jørgensen et al. 2004; Weaver, Zablotowicz et al. 2004; Bouldin, Farris et al. 2005; Haarstad, Braskerud et al. 2005; Pot, Simunek et al. 2005; Popov, Cornish et al. 2006; Fox and George 2009; Gregoire, Elsaesser et al. 2009; Poletika, Coody et al. 2009; Boutron, Margoum et al. 2011).

- Depth of surface water versus vegetation height and density (Barling and Moore 1994; Loch, Espigares et al. 1999; Jin and Römkens 2001; Popov, Cornish et al. 2006), hydraulic conductivity and holding capacity of buffer zone soil (Asmussen, Hauser et al. 1977; Smith, Melvin et al. 1992; Norris 1993; Grismer, O'Geen et al. 2006).
- Soil properties (Smith, Melvin et al. 1992; Hairsine 1997; Krutz, Senseman et al. 2003b; Fares and Ryder 2006; Grismer, O'Geen et al. 2006; Popov, Cornish et al. 2006; Sabbagh, Fox et al. 2009; Fox, Muñoz-Carpena et al. 2010).
- Initial soil water content (Smith, Melvin et al. 1992; Grismer, O'Geen et al. 2006; Otto, Vianello et al. 2008; Pinho, Morris et al. 2008; Sabbagh, Fox et al. 2010).
- Rainfall characteristics (total amount and intensity) (Asmussen, Hauser et al. 1977; Smith, Melvin et al. 1992; Norris 1993; Barling and Moore 1994; Elsenbeer, West et al. 1994; Parson 1994; Snyder 1998; Boyd, Baker et al. 2003; Weaver, Zablotowicz et al. 2004; Bouldin, Farris et al. 2005; Polyakov, Fares et al. 2005; Grismer, O'Geen et al. 2006; Popov, Cornish et al. 2006; Pätzold, Klein et al. 2007; Liu, Zhang et al. 2008; Fox and George 2009; Hoffmann, Kjaergaard et al. 2009; Lizotte, Shields et al. 2009; Poletika, Coody et al. 2009; Yuan, Bingner et al. 2009; Caron, Lafrance et al. 2010; Fox, Muñoz-Carpena et al. 2010; Sabbagh, Fox et al. 2010).

The main ‘trapping’ mechanisms of pollutants (pesticides and nutrients) (Figure 7.1) are:

- Infiltration (Asmussen, Hauser et al. 1977; Arora, Mickelson et al. 1996; Robinson, Ghaffarzadeh et al. 1996; Klöppel, Kördel et al. 1997; USDA 2000; Reungsang, Moorman et al. 2001; Seybold, Mersie et al. 2001; Boyd, Baker et al. 2003; Krutz, Senseman et al. 2003a; Krutz, Senseman et al. 2004; Syversen and Bechmann 2004; McKergow, Prosser et al. 2006; Popov, Cornish et al. 2006; Pätzold, Klein et al. 2007; Otto, Vianello et al. 2008; Pinho, Morris et al. 2008; Poletika, Coody et al. 2009; Fox, Muñoz-Carpena et al. 2010).
- Sedimentation (Asmussen, Hauser et al. 1977; Smith, Melvin et al. 1992; Barling and Moore 1994; Parson 1994; Arora, Mickelson et al. 1996; Robinson, Ghaffarzadeh et al. 1996; USDA 2000; Dosskey 2001; Rankins Jr, Shaw et al. 2001; Seybold, Mersie et al. 2001; Boyd, Baker et al. 2003; Grismer, O'Geen et al. 2006; Popov, Cornish et al. 2006; Pätzold, Klein et al. 2007; Rose, Crossan et al. 2008; Poletika, Coody et al. 2009; Fox, Muñoz-Carpena et al. 2010).

These trapping mechanisms are relatively effective dependent on those points identified above.

Highly soluble pesticides or nutrients are lost via infiltration (however this just takes a different flow path through the subsoil and groundwater) (Robinson, Ghaffarzadeh et al. 1996; Klöppel, Kördel et al. 1997; USDA 2000; Boyd, Baker et al. 2003; Runes, Jenkins et al. 2003; Syversen and Bechmann 2004; Pätzold, Klein et al. 2007; Sabbagh, Fox et al. 2009) (Figure 7.2). Pesticides adsorbed onto particulates and nutrients in particulate form are lost via sedimentation (Smith, Melvin et al. 1992; Barling and Moore 1994; Rankins Jr, Shaw et al. 2001; Boyd, Baker et al. 2003; Runes, Jenkins et al. 2003; Popov, Cornish et al. 2006; Pätzold, Klein et al. 2007; Budd, O’Geen et al. 2009) (Figure 7.3). This requires a long holding time, firstly allowing sediments to drop out (Smith, Melvin et al. 1992; Norris 1993; Barling and Moore 1994; Nguyen, Downes et al. 1999; Wong, Breen et al. 1999; Hunter and Lukacs 2000; Dosskey 2001; Boyd, Baker et al. 2003; Stearman, George et al. 2003; Pot, Simunek et al. 2005; Hoffmann, Kjaergaard et al. 2009; Poletika, Coody et al. 2009) and secondly, to allow for pesticide residence times - to ensure that biodegradation occurs (Norris 1993; Barling and Moore 1994; Dosskey 2001; Kao, Wang et al. 2002; Boyd, Baker et al. 2003; Stearman, George et al. 2003; Brønnum, Jørgensen et al. 2004; Pot, Simunek et al. 2005; Grismer, O'Geen et al. 2006; Pinho, Morris et al. 2008; Gregoire, Elsaesser et al. 2009; Poletika, Coody et al. 2009; Page, Dillon et al. 2010).

- “infiltration was the only mechanism that significantly reduced herbicide loads at runon depths between 160 and 800mm” (Popov, Cornish et al. 2006).
- “dissolved contaminants and colloid-bound contaminants initially retained by water infiltration can continue to move to streams as subsurface flow through macropores” (Pinho, Morris et al. 2008).

Organic matter is another major trapping mechanism for pesticides (Kao, Wang et al. 2002; Selim, Zhou et al. 2003; Selim, Naquin et al. 2004; Popov, Cornish et al. 2006; Pätzold, Klein et al. 2007; Rose, Crossan et al. 2008; Sabbagh, Fox et al. 2009; Dousset, Thévenot et al. 2010; Page, Dillon et al. 2010).

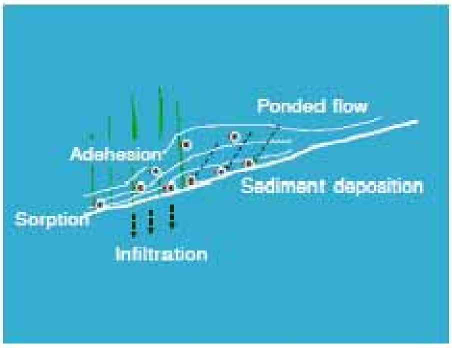


Figure 7-1. The primary pathways of loss of pesticides from agricultural land. Source: USDA (2000).

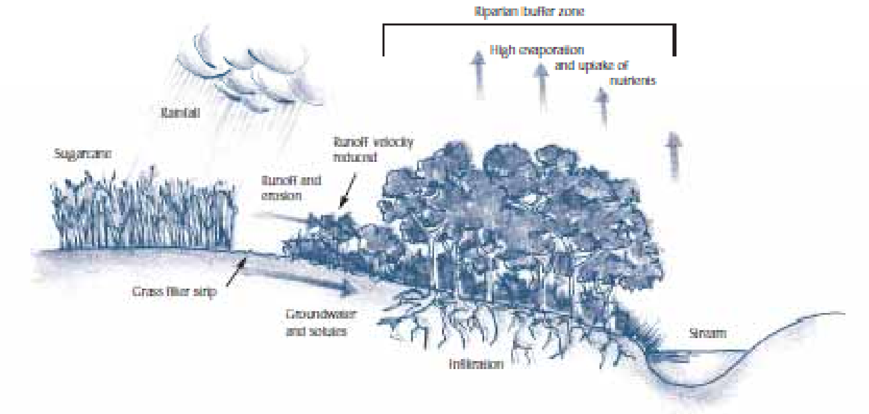


Figure 7-2. How the grass filter strip and riparian buffer zone functions to remove sediments and nutrients from agricultural runoff. Source: The Idea to Here - Lovett and Price (2001).

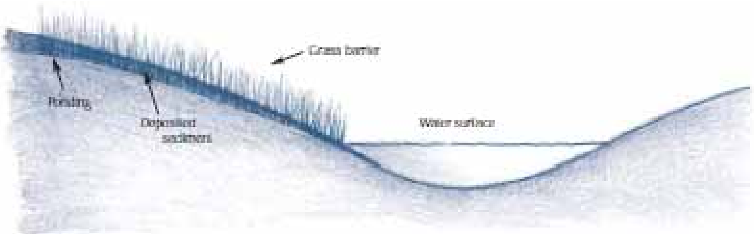


Figure 7-3. How a grass buffer strip functions to trap sediment. Source: The Idea to Here - Lovett and Price (2001).

Vegetated buffer zones are effective at removing pesticides and some nutrients on property scale, but not broad catchment scale without being part of a strategic 'maze' of filters strips installed across the catchment (Norris 1993; Polyakov, Fares et al. 2005; Reichenberger, Bach et al. 2007). However, this still does not account for pollutants lost via infiltration to groundwater.

As one of the main forms of herbicide trapping is via sediment trapping the long-term performance of buffers in sediment trapping is very relevant. Researchers have observed that the effectiveness of grass filter strips may decrease over time as the strip becomes inundated with sediment or as the ground becomes saturated with runoff. For example, in an experiment in Virginia, researchers demonstrated that a filter strip which initially removed 90 percent of the sediment was removing only 5 percent of the sediment after six trials (Dillaha, Reneau et al. 1988). Buffers may be most effective at removing large particles such as sand, but may be less effective at removing small clay particles. In Arizona, researchers found that sand particles could be removed by grass buffers within a fairly short distance from the field edge (as little as 3m), while the removal of silt particles required a buffer of 15m (Wilson 1967). Filter strips 100 to 130m feet wide were required to remove clay particles.

Many factors influence the ability of the buffer to remove sediments from land runoff, including the sediment size and loads, slope, type and density of riparian vegetation, presence or absence of a surface litter layer, soil structure, subsurface drainage patterns, and frequency and force of storm events (Osborne and Kovacic 1993). Riparian buffers must be properly constructed and regularly monitored in order to maintain their effectiveness (Mitsch 1992; Norris 1993; Qiu and Prato 1998; Snyder 1998; Wong, Breen et al. 1999; USDA 2000; Polyakov, Fares et al. 2005; Rankins Jr, Shaw et al. 2005; Grismer, O'Geen et al. 2006; Liu, Zhang et al. 2008; Otto, Vianello et al. 2008; Smith 2008). Probably the most important consideration is the maintenance of shallow sheet flow into and across the buffer. Where concentrated flow paths begin to form or deep sediments begin to accumulate, the buffer can no longer maintain its filtering ability (Norris 1993; Barling and Moore 1994; Hatfield, Mickelson et al. 1995; Jin and Römkens 2001; Krutz, Senseman et al. 2003b; Polyakov, Fares et al. 2005; Yuan, Bingner et al. 2009; Fox, Muñoz-Carpena et al. 2010). Maintaining shallow sheet flow into the buffer can be especially troublesome in areas where slopes are steep and surface flows tend to concentrate.

7.2 Pesticide transport

A common method for alleviating pesticide loading to nearby surface water bodies is the use of riparian buffers or vegetated filter strips or buffers at the paddock boundary or adjacent to waterways (Lowrance, Vellidis et al. 1997; Qiu and Prato 1998; Popov, Cornish et al. 2006; Reichenberger, Bach et al. 2007; Smith 2008). These buffers reduce pesticide movement to streams by reducing runoff volumes through infiltration in the filter strip's soil profile, through contact between dissolved phase pesticide with soil and vegetation in the filter strip, and/or by reducing flow velocities to the point where eroded sediment particles, with sorbed pesticide, can settle out of the water. As shown in Section 4, pesticides vary in how tightly they are adsorbed to soil particles which is particularly relevant to understanding the efficiency of buffers in retaining pesticides. Degree of soil binding is measured by binding coefficients, or K values. K_{oc} (K of organic carbon) is a measure of adsorption to the organic matter or carbon content of soil, with higher values indicating more binding. While pesticides are also bound to clay particles, binding to organic matter is a useful predictor of pesticide behavior and movement in soil. K_{oc} values can be used to predict whether a specific pesticide will be carried primarily in the sediment or dissolved phase of paddock runoff.

Example K_{oc} values for specific pesticides are shown in Table 7.1 and range from 2 for dicamba (which is held loosely in the soil) to 1 million for paraquat (which is bound tightly to soil). K_{oc} values greater than 1,000 indicate that pesticides are highly adsorbed to soil and examples of these typically used in sugar cane in the GBR catchments include paraquat, chlorpyrifos, glyphosate and diuron (and probably MEMC). These pesticides tend to be carried off paddocks on eroded soil particles. Thus, if buffers are effective in trapping the sediment particle sizes that transport the pesticides, they have potential to effectively trap this type of pesticide. Of these pesticides, only diuron has been identified as a focus within this project. Pesticides with lower K_{oc} values (generally less than 500) tend to move more in water than on sediment and examples of these typically used in sugar cane in the GBR catchments include ametryn, atrazine, 2,4-D, hexazinone, imazapic, imidochloprid, metolachlor and metribuzin. The remaining pesticides being considered in this project fall within this category.

Most researchers agree that filter strips trap highly sorbing pesticides in the same manner that they trap sediment. Spatz (1999) suggests that pesticide attached to eroded sediment becomes the dominant transport mechanism only for strongly sorbing (i.e., $K_{oc} > 1000$ L kg–1) pesticides (Reichenberger, Bach et al. 2007; Arora, Mickelson et al. 2010). For low to moderately sorbed pesticides, runoff must infiltrate while in the filter strip or pesticide can be removed from solution through contact with the soil or vegetation that may adsorb pesticides in the filter strip (USDA 2000; Rose, Sanchez-Bayo et al. 2006; Pätzold, Klein et al. 2007; Gregoire, Elsaesser et al. 2009; Arora, Mickelson et al. 2010; Dousset, Thévenot et al. 2010). Concentrations carried on sediment are higher than concentrations in water, but because water quantities running off paddocks are so much greater than eroded soil quantities, water accounts for the majority of chemicals leaving paddocks.

The relationship between Koc and the percent of pesticide trapped is shown in Figure 7.4.

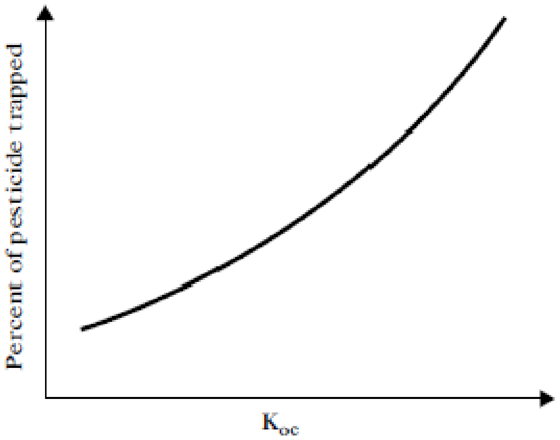


Figure 7-4. Relationship between percent of pesticide trapped and Koc. Source: USDA (2000).

In contrast to most pesticides, nitrate is water-soluble and not readily adsorbed by soil particles. Usually nitrate is not in runoff because in enters the soil quickly. Rather, nitrate that is not taken up by plants may leach to ground water and be carried to streams by subsurface flow. Significant losses of nitrate in surface runoff can occur in certain situations, such as heavy rainfall after surface application of nitrogen fertilizer. To trap nitrate effectively, roots of conservation buffer plants need to intercept this subsurface flow. Conditions for denitrification present in this biologically active zone also reduce nitrate reaching streams. Similarly, some weakly adsorbed pesticides may leach to shallow ground water in small amounts. Although subsurface flow may carry small quantities of pesticides to streams, quantities present in surface runoff are usually much greater.

Table 7-1. Summary of buffer studies measuring trapping efficiencies for specific pesticides. Note: Shaded cells indicate pesticides used in sugar cane application in the GBR catchments. Source: Derived from USDA (2000).

Pesticide	K_{oc}	Study Reference	Percent pesticide trapped (%)
Highly adsorbed pesticides			
Chlorpyrifos	6,070[1]	Boyd et al., 1999	57-79
		Cole et al., 1997	62-99
Diflufenican	1,990[1]	Patty et al., 1997	97
*Diuron	1,067[2]		
Glyphosate	21,699[2]		
Lindane	1,100[1]	Patty et al., 1997	72-100
Paraquat	1,000,000[2]		
Trifluralin	8,000[1]	Rhode et al., 1980	86-96
MEMC (methoxyethylmecuric chloride),			
Moderately adsorbed pesticides			
Acetochlor	150[1]	Boyd et al., 1999	56-67
Alachlor	170[1]	Lowrance et al., 1997	91
*Ametryn	316[2]		
*Atrazine	100[1]	Arora et al., 1996	11-100
		Barfield et al., 1998	90
		Boyd et al., 1999	52-69
		Hall et al., 1983	91

Pesticide	K_{oc}	Study Reference	Percent pesticide trapped (%)
		Hoffman, 1995	30-57
		Lowrance et al., 1997	97
		Mickelson and Baker, 1993	35-60
		Misra et al., 1996	26-50
		Patty et al., 1997	44-100
		Popov et al., 2006	40-85
Cyanazine	190[1]	Arora et al., 1996	80-100
		Misra et al., 1996	30-47
2,4-D	20[2]	Asmussen et al., 1977	70
		Cole et al., 1997	89-98
Dicamba	2[1]	Cole et al., 1997	90-100
Fluormeturon	100[1]	Rankins et al., 1998	60
		Rankins et al., 2001	59
*Hexazinone	54[2]		
Imazapic	137[2]		
Imidochloprid	225[2]		
Isoproturon	120[1]	Patty et al., 1997	99
Mecoprop	20[1]	Cole et al., 1997	89-95
Metolachlor	200[1]	Arora et al., 1996	16-100
		Misra et al., 1996	32-47
		Popov et al., 2006	44-85
		Webster and Shaw, 1996	55-74
		Tingle et al., 1998	67-97
Metribuzin	60[1]	Webster and Shaw, 1996	50-76
		Tingle et al., 1998	73-97
Norflurazon	600[1]	Rankins et al., 1998	65
		Rankins et al ., 2001	63-86

Notes: * indicates the pesticides of focus in this project. K_{oc} values listed for each pesticide are from [1] the NRCS Field Office Technical Guide, Section II Pesticide Property Database and [2] 'Footprint' Pesticide Properties Database (http://sitem.herts.ac.uk/aeru/footprint/en/index.htm).

7.3 Trapping efficiency

Data from a range of studies included in Appendix 1 has enabled the authors to make a number of broad conclusions regarding the trapping efficiency of sediments, nutrients and pesticides in buffers. For example:

- A 60-90% reduction in sediment can be expected as runoff filters through a grassed filter strip.
- Approximately 50-90% reduction in nutrients can be expected, depending on the nutrient species and the species of grass, e.g.
    - 60-90% reduction in Phosphorus (Polyakov, Fares et al. 2005); and
    - 47-100% reduction in nitrate (Patty, Réal et al. 1997).
- Table 3 includes results from studies showing the pesticide trapping efficiencies for a range of pesticides. Highly adsorbed pesticides were trapped at rates of from 62 to 100%. Trapping of moderately adsorbed pesticides was more variable and ranged from 11 to 100%. Lowest percent pesticide retention by buffers occurred when buffer soil was saturated due to previous rains. Many studies found pesticide trapping efficiencies of 50% or more.
- As runoff velocity increases, the ability of the filter strip to remove or trap pollutants decreases (Poletika, Coody et al. 2009). Results in Boutron et al. (2011) suggest that runoff velocities of 2 cm/s resulted in the filter strip removing less pesticides than an increased runoff velocity of 7 cm/s. It was hypothesised however, that pesticide removal would decrease once the filter strip reached saturation; at high velocities saturation is achieved quickly.

- Schmitt et al. (1999) found that the dilution of runoff by rainfall was the most significant mechanism reducing the concentration of dissolved contaminants and that infiltration reduced the volume of runoff leaving the filter strip by 36 to 82%.
- Reichenberger et al. (2007) is an excellent review paper on the efficiencies of different trapping mechanisms (wetlands, grassed filter strips, riparian filter strips etc). It concludes that sub-surface drains are an effective mitigation measure for pesticide runoff losses from slowly permeable soils with frequent waterlogging.
- Performance of constructed wetlands in removing pollutants is influenced by area, length to width ratio, water depth, rate of wastewater loading and retention time. Removal efficiency – organic material and SS 80%, nutrients <60% (Shutes 2001).
- Wetlands are effective in reducing concentrations of pesticides as a result of retention time, sedimentation, adsorption onto organic matter/organic carbon, and plant uptake. 33–51% reduction in diuron and 20–60% in simazine (Runes, Jenkins et al. 2003; Reichenberger, Bach et al. 2007; Rose, Crossan et al. 2008; Page, Dillon et al. 2010).

Studies have also been conducted to determine the effect of pesticide dosage on grass strips. For example, Popov and Cornish (2006) tested the tolerance of four native and introduced grass species (in New South Wales, Australia) to long term low dose atrazine in runoff and found that they may be successfully included by farmers when designing new or maintaining existing buffers. These alleviate potential concerns regarding the effect of pesticide runoff on the health of the actual buffers.

7.4 Buffer width

Appropriate widths for buffers are debatable. Widths are defined here as flow length across the buffer. Buffer per unit area is affected by runoff flow rate and depth as well as by conditions within the buffer, such as soil type and antecedent moisture that affect water infiltration. Amount of runoff is affected by source area size and properties as well as rainfall intensity and quantity.

Selecting an appropriate buffer size often involves consideration of several desired functions, site conditions, and what is economically or politically practical. Many studies have investigated sediment trapping efficiency of grass buffers (Robinson, Ghaffarzadeh et al. 1996; Patty, Réal et al. 1997; Snyder 1998; Tingle, Shaw et al. 1998; Mickelson, Helmers et al. 2004; Otto, Vianello et al. 2008; Sabbagh, Fox et al. 2009; Yuan, Bingner et al. 2009). For example, in a recent review by Yuan *et al*. (2009), it was concluded that although sediment trapping capacities are site- and vegetation-specific, and many factors influence the sediment trapping efficiency, the width of a buffer is important in filtering agricultural runoff and wider buffers tended to trap more sediment. Sediment trapping efficiency is also affected by slope, but the overall relationship is not consistent among studies. Overall, sediment trapping efficiency did not vary by vegetation type and grass buffers and forest buffers have roughly the same sediment trapping efficiency.

Wider buffers tended to trap more sediment, but other factors also influence efficacy. Overall, the sediment trapping efficiency to buffer width relationship can be best fitted with logarithm models (Figure 7.5). According to this relationship, a 5 m buffer can trap about 80% of incoming sediment (Yuan, Bingner et al. 2009). Table 7.2 provides a summary of some findings regarding trapping efficiency at various buffer widths. Note that the rainfall conditions for the study areas are incorporated for comparison.

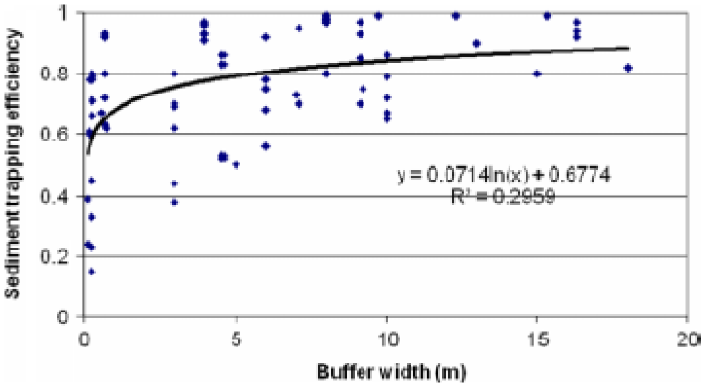


Figure 7-5. Buffer width and sediment trapping efficiency. Source: Yuan et al. (2009).

Table 7-2. Trapping efficiencies for various buffer widths and rainfall conditions.

Reference	Conclusions	Most efficient buffer width	Rainfall
Boutron et al., (2011)	Filtering ability of filter strip increases with increasing surface flow velocity. It was hypothesised however that once saturation was achieved filtering ability would diminish.		Simulated surface flows of 2cm/s and 7 cm/s.
Schmitt et al., (1999)	Doubling the filter strip width from 7.5 to 15 m doubled infiltration and dilution. Infiltration is an important mechanism for the removal of dissolved chemicals. Soils with low infiltration rates could be ineffective when constructed for the purpose of removing dissolved pesticides from runoff	15m	Simulated - 25.4mm in 30 mins
Dillaha et al., (1989)	30- and 15-foot strips of orchard grass trapped 84 and 70 % of incoming solids, respectively. The source area of runoff was 60 feet, or 4 times as wide as the 15-foot buffers.	10m	Simulated – 50 mm/hr. Virginia
Magette et al., (1989)	30- and 15- foot strips of fescue trapped 75 and 52 % of incoming solids, respectively. The source area was 72 feet deep, or 4.8 times as wide as the 15-foot buffers.	10m	Simulated – 48.25 mm/hr.
Castelle, et al., (1994)	A range of buffer widths from 10 to 650 feet were effective, depending on site-specific conditions. A buffer width of at least 50 feet was necessary to protect wetlands and streams under most conditions.	>15m	Various
(USDA 2000)	Draft NRCS Conservation Practice Standard for Filter Strips requires a minimum flow length of 30 feet for the purpose of reducing sediment and sediment-adsorbed contaminant loadings. It also sets ratios of filter strip area to field area based on Universal Soil Loss Equation R factor values (rainfall amount and intensity) of regions.	>10m	Various – USA conditions.
(USDA 2000)	Typical buffer widths of about 15 metres can be effective in reducing pesticide runoff by at least 50 % if sheet flow is maintained, depending on a number of factors as described previously (USDA, 2000).	15m	Various – USA conditions.

Reference	Conclusions	Most efficient buffer width	Rainfall
(Hook 2003)	More than 97% of sediment was trapped in the rangeland riparian buffer area with a 6 m buffer in any of the experimental conditions they studied. Retention was not affected by stubble height.	6m	Simulated. Montana foothills, USA
(Yuan, Bingner et al. 2009)	Buffers of 3–6 m wide have greater sediment trapping efficiency than buffers of 0–3-m wide, and buffers of greater than 6 m wide have greater sediment trapping efficiency than buffers of 3–6 m wide. Thus, wider buffers are likely to be more efficient in trapping sediment than narrower buffers.	>6m	
(Pätzold, Klein et al. 2007)	12-m wide grass filter strips provided an almost optimal reduction of herbicide output from arable fields via surface run-off.	12m	Annual rainfall 1072 mm. Annual rainfall over study period was 988 mm (1997), 1309 mm (1998) and 1236 mm (1999).
(Pinho, Morris et al. 2008)	Reductions in solution concentrations and mass retention of P and 2 herbicides (atrazine and picloram) which were observed for simulated flow within 10m wide forested filter strips across a range of slopes and organic horizon conditions.	10m	Simulated wet & dry. Coastal Piedmont of Georgia – annual ~ 1300mm, summer ~500mm, max daily 250mm.
(Patty, Réal et al. 1997)	Grassed buffer strips are effective in restricting pollutant transfer in runoff; those with widths of 6, 12 and 18 m reduced runoff volume by 43 to 99%, suspended solids by 87 to 100%, lindane losses by 72 to 100% and loss of atrazine and its metabolites by 44 to 100%.	6m, 12m, 18m	Various
(Parson 1994)	8 m filters strips were more effective than 4m filter strips in removing all potential contaminants from the runoff water; but doubling the filter length almost never doubled the grass or riparian filter effectiveness for removal of any constituent.	8m	Kinston, North Carolina – annual average ~1200mm
(Otto, Vianello et al. 2008)	Vegetated Filter Strips 6m wide are very effective in reducing runoff volume and concentration during both wet and dry years, in comparison to 3m wide strips.	6m	Annual average ~805mm (Nth Eastern Italy)
(Norris 1993)	Small controlled runoff plots with buffers 5-10m in width are successful in removing a variety of pollutants from overland flow (sediment, nutrients, chemicals).	5-10m	Simulated
(Snyder 1998)	Tested the effectiveness of different tillage systems and buffer widths. Most effective was no till, 45ft vegetated filter – reduced runoff volume (91%), suspended sediment (99%), nitrate-N (97%) and atrazine (98%).	~14m	Kentucky - annual average rainall 1350mm
(McKergow, Prosser et al. 2004b)	On planar slopes grass buffers strips were able to trap >80% of the incoming bedload; TP, TN and TSS were reduced by 25-65% within the first 15m of the buffer. Loads leaving the buffer were often higher than those entering due to seepage as a result of prolonged or high frequency rainfall. During these conditions the function of the buffer is erosion control rather than a	>15m	Innisfail Qld, ~ 3585m average annual rainfall.

Reference	Conclusions	Most efficient buffer width	Rainfall
	trap for sediment and nutrients. Dense grass riparian buffer strips >15m wide may be able to trap significant quantities of bedload. Trapping is more successful when infiltration occurs. On steep slopes buffer strips would best be installed at the ends of crop rows, where contributing areas are smaller.		
(Mickelson, Helmers et al. 2004)	The first 5m of the vegetative filter strip plays a significant role in removal of suspended solids (>40μm). Length >10m does not significantly improve vegetative filter strip performance. Infiltration is the only mechanism that allows for removal of smaller size sediments (<40μm). Vegetative cover helps to reduce velocity of runoff and increase residence time for water to infiltrate. High vegetation density leads to less erosion and less transport capacity of the runoff and therefore greater settling of sediments. Non-submerged vegetation allows for the greatest flow retardation and minimum sediment transport capacity. Perpendicular planting may be an effective means of managing non-uniform or concentrated flow by slowing down flow velocity. Time elapsed between the time of pesticide application and rainfall event has an important role in pesticide losses. Pesticide losses in vegetative filter strips are reflected by adsorption properties of the pesticides.	5-10m	Study sites were within the Rock Creek Watershed (Newton), Iowa.
(Robinson, Ghaffarzadeh et al. 1996)	The initial 3.0m of the vegetative filter strip removed more than 70% of the sediment from runoff while 9.1m of the vegetative filter strip removed 85%. There was little decrease in sediment concentration was observed with greater vegetative filter strip widths Slopes of 12% grade had greater runoff and soil losses at all vegetative filter strip widths than the 7% grade. Vegetative filter strips promoted infiltration, reduced runoff volumes, and decreased runoff sediment concentration.	3.0-9.1m	Study sites were in north-east Iowa, U.S.
(Sabbagh, Fox et al. 2009)	Filter strip width was not a statistically significant parameter in the empirical model.	N/A	Various.
(Tingle, Shaw et al. 1998)	Differences in parameters were significant between filter and no filter strips regarding filter strip width. Filter strips regardless of width reduced cumulative runoff and sediment loss at least 46% and 83% respectively. Herbicide losses, runoff amounts, and sediment amounts, both within events and cumulative, were regressed in linear, quadratic, logarithmic, and exponential form against filter strip width. Highest surface runoff from unfiltered treatment. Sediment losses reduced 98-99% with filter strip.	4m	Study sites were in Mississippi, U.S.

Several site characteristics may dictate wider buffers, especially when trying to maximize water infiltration and trapping of dissolved pesticides. For example, fine textured soils generally have lower water infiltration rates; or a high water-table underlying buffers may limit infiltration. Iowa studies found that water infiltration and trapping of dissolved herbicides by buffers was least effective when previous rains saturated soils. Vegetation within the buffer improves surface soil conditions, improving infiltration rates and internal soil drainage. Slope also has a significant influence on trapping efficiency.

However, it has also been presented that while site characteristics, such as large source areas or slow permeability soils, may dictate larger buffers for high pesticide trapping efficiency; relatively small buffers can provide significant water quality benefits. Wider buffers may provide greater protection than narrow buffers in many settings, but where space or cost considerations limit buffer widths, a narrow buffer is better than no buffer at all. Narrow buffers have sometimes trapped pesticides effectively. The specific pesticide studies included in Appendix 1 found that buffers as narrow as 0.5 metres could be effective in trapping significant quantities of pesticides. Increasing buffer width did not always significantly improve pesticide trapping. Tingle *et al*. (1998) compared tall fescue buffers measuring 0.5 to 4 metres wide placed below ~20 metres long soybean plots. No significant differences in pesticide trapping efficiencies were found between buffer widths. Runoff loss of metribuzin was reduced by at least 73 percent, and runoff loss of metolachlor was reduced at least 67 percent by all buffer widths.

Given the information presented in Table 7.2 and additional discussion in this section, it is evident that a buffer width of at least 6 metres and in many cases, 10-15 metres, provide the most efficient trapping of sediments, pesticides and nutrients in most climates. However, a majority of the cases presented are relevant to temperate rainfall conditions where rainfall does not exceed 1500mm per year. Limited examples of trapping efficiency in rainfall typical of tropical environments are available; however, the work of McKergow and others provides important information regarding the effectiveness of vegetated buffers in trapping materials in a high rainfall area. This work is summarized in Section 7.6 below. Karssies and Prosser (1999) have also determined indicative soil losses and design filter widths the six bio-geographical regions of Queensland, for varying rainfall erosivity, soil erodibility, slope and land cover. The results for the Wet Tropics and Burdekin Regions are shown in Table 7.3. It is clear from this information that buffer widths in the Wet Tropics (800-5000mm annual rainfall) in areas where there is poor cover (C = 0.2) must be at least 30 metres to minimize soil loss. In areas with good cover (C = 0.01), buffer widths between 2 and 12 metres are required, depending on the site characteristics (rainfall erosivity, soil erodibility and slope). These results are assumed to be similar for particle bound pesticides but are not relevant to dissolved materials.

Table 7-3. Indicative soil losses and design filter widths for two of the six bio-geographical regions of Queensland, for varying rainfall erosivity, soil erodibility, slope and land cover. Source: Karssies and Prosser (1999).

Region (annual rainfall) (mm/y)	Rainfall Erosivity[1]	Soil erodibility[2]	Slope	Poor cover[3] soil loss (t/ha/y)	Filter width	Good cover [4] soil loss (t/ha/y)	Filter width (m)
WET TROPICS 800-5000	High	Medium	Low	17	7	1	2
			Medium	41	26	2	2
			High	74	>30	4	2
		High	Low	25	15	1	5
			Medium	61	>30	3	5
			High	112	>30	6	7
	V.High	Medium	Low	29	15	1	2
			Medium	71	>30	4	2
			High	130	>30	7	2
		High	Low	44	27	2	5
			Medium	107	>30	5	7
			High	195	>30	10	10
	Extreme	Medium	Low	38	20	2	2
			Medium	92	>30	5	2
			High	167	>30	8	2
		High	Low	57	>30	3	5
			Medium	138	>30	7	7
			High	251	>30	13	12
BURDEKIN 500-1200	High	Low	Low	8	2	0	2
			Medium	20	13	1	2
			High	37	24	2	2
		Medium	Low	17	7	1	2
			Medium	41	26	2	2
			High	74	>30	4	2
		High	Low	25	15	1	5
			Medium	61	>30	3	5
			High	112	>30	6	7
	V.High	Low	Low	15	5	1	2
			Medium	36	23	2	2
			High	65	>30	3	2
		Medium	Low	29	15	1	2
			Medium	71	>30	4	2
			High	130	>30	7	2
		High	Low	44	27	2	5
			Medium	107	>30	5	6
			High	195	>30	10	10

Notes:

[1] Rainfall erosivity R: low = 850; medium = 2000; high = 4000; very high = 7000; extreme = 9000.

[2] Soil erodibility K: high = 0.045; medium = 0.030; low = 0.015

[3] Slope S: high = 9%; medium = 6%; low = 2%

[4] Poor cover C = 0.2 (traditional tillage practices, bare soil for some periods, partially covered with crop for remainder of year; good cover C = 0.01 (improved tillage practices, mostly permanent cover)

7.5 Influence of climatic conditions

Heavy rainfall events causing storm runoff are always associated with the production of extremely large volumes of water in a short time. In many circumstances, these large water volumes may not be retained by any of widely employed buffer strip, and erosion channels formed during these conditions may further jeopardize the positive effect of buffer zones. This ‘hydrological dilemma’ may result in unavoidable pesticide contaminations of surface waters specifically under conditions where other measures are not applicable or do not provide the necessary benefit (Sabbagh, Fox et al. 2009).

In an experimental study Popov, Cornish *et al.*, (2006) showed that biofilters were still effective in conditions of intense runoff (simulated rainfall) from cropland (runon depths of 160, 320, 800mm over 0.7 and 0.8h). Buffer width was 1.25m x 4m. Pot *et al*. (2005) simulated several rainfall intensities (0.070, 0.147, 0.161, 0.308 and 0.326 cm/hr). At the highest rainfall intensity, multiple porosity domains and multiple permeabilities of preferential flow were probably active and there was rapid flow through macropore pathways, slower flow through a mesoporosity and no-flow in remaining micropores. At lower rainfall intensities, macropore flow was not active anymore.

Using input data from field trials in the United States (Sabbagh, Fox et al. 2009), Sabbagh *et al.*, (2010) showed a nonlinear relationship between total water input (rainfall plus runon) during a storm event and percent pesticide reduction using a predictive model (see also Section 7.8). It was also identified that differences in soil moisture affect pesticide reduction where lower reductions are recorded with higher soil moisture content, i.e. lower infiltration capacity. Other factors controlling the range of responses for each filter length are linked to the range of rainfall intensities and durations that resulted in differences in sediment characteristics (particle size distribution) in runon from the source area. For example, for a 9.1 metre long buffer, pesticide trapping or reduction was generally greater than 60% unless the total water input (rainfall plus runon) during a storm event exceeded 10 cm (Figure 7.6 below). Note that the K_{oc} included in the study of 100L/kg is the same as for atrazine.

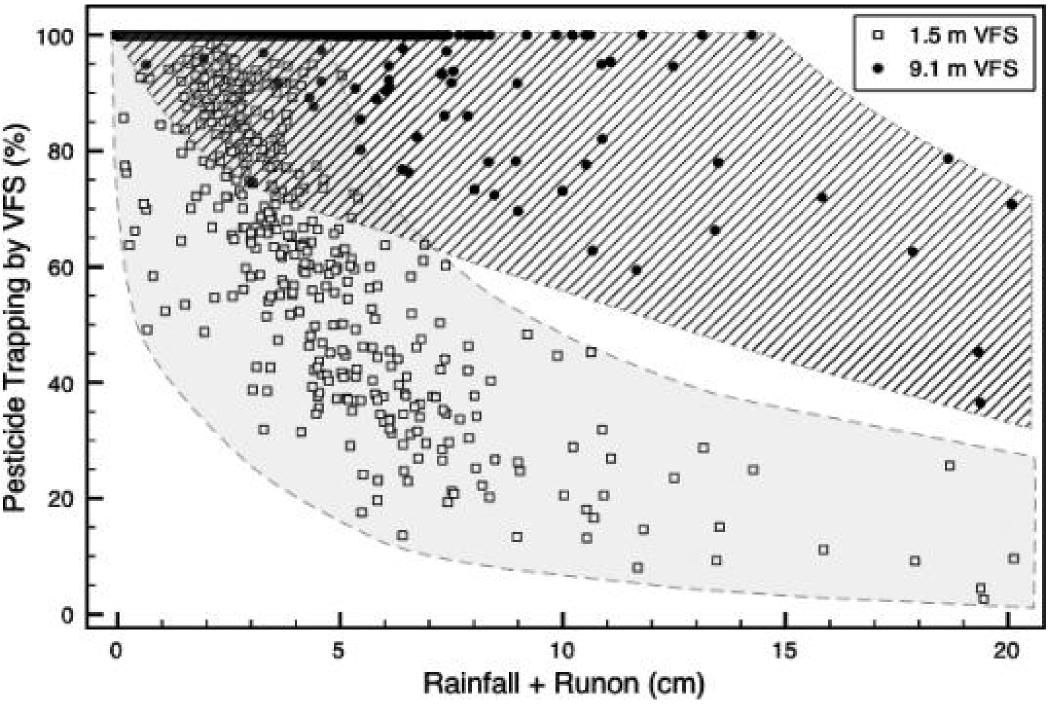


Figure 7-6. Nonlinear relationship between cumulative rainfall and runon (cm) entering a vegetative filter strip (VFS) versus percent pesticide reduction (%) relative to two different buffer lengths (1.5 and 9.1m) as predicted using the VFSMOD model for the U.S. EPA Illinois corn scenario. Data shown are for a pesticide with organic carbon sorption coefficient, Koc = 100L/kg OC. Source: Sabbagh *et al.*, (2010).

In the wet tropics of Far North Queensland, rainfall conditions are extreme and this is highly significant in assessing the effectiveness of buffers in removing materials from agricultural runoff in these environments. For example, McKergow, Prosser *et al.*, (2004a; 2004b) have shown that intense cropping, high intensity rainfall and a steep landscape reduce the effectiveness of riparian vegetated buffer strips. The study area included hillslopes in the banana and sugar cane growing area of wet tropical Far North Queensland. The study site was located in the North Johnstone River catchment, which reaches the coast at Innisfail. Average rainfall at Innisfail is 3585mm during the wet season, December to April. During the wet season peak rainfall events can result in 533mm falling over a 65 hour period, as recorded in 1998-1999.

Riparian buffers at four sites were monitored. Soil at these sites was classified as krasnozems, derived from basalt, with high permeability and available water storage capacity.

- Site 1: 15m wide riparian buffer at Gallagher's Grass, planted with signal grass (*Brachiaria decumbens*) draining a 0.2ha area on a 7% gradient.
- Site 2: 15-20m remnant rainforest riparian buffer at Gallagher's Tree, with no understorey, draining a 0.2ha area on a 7% gradient.
- Site 3: 50m wide signal grass buffer with 4 vetiver grass hedges (*Vetiveria zizaniodes*) located 5, 10, 25 and 45m below the plume at Dunne's Extreme, which drained 5ha on a 13% gradient.
- Site 4: 60m of gently sloping hollow covered with dense signal grass, which drained 0.3ha on a 3% gradient at Dunne's Moderate.

On planar 7% slopes peak discharges from slope lengths of 200m were <30L/s. However, peak surface water discharge from a 5 hectare convergent catchment was >350L/s. As a result the grass buffer and vetiver hedges in this catchment were unable to reduce runoff velocity or disperse flow sufficiently in a large event and a channel was scoured in through the buffer. High discharges and channelized flow decreased the buffers effectiveness.

This study showed that infiltration of surface runoff is unlikely to be an important factor for riparian buffers in the wet tropics. During small events runoff did infiltrate into the riparian soils, however, during large events infiltration is limited due to the high surface runoff velocities which may reduce the ability of runoff to infiltrate. During large rainfall events, significant runoff still reaches the streams due to the large runoff volumes. However, it should be noted that the sites on planar slopes (Site 1,Site 2 and Site 4) were able to withstand peak discharge events as the planar slopes allow flow to disperse, assisted by the grass vegetation on a low gradient slope reduces flow velocity and depth.

Saturation overland flow, return flow and seepage increased the volume of surface runoff flowing through the buffer strips, where soil depth was shallow. In these conditions, it appears that buffer strips best function as erosion control.

Further work assessing trapping efficiency (see McKergow, Prosser *et al*.(2004b)) found that on planar slopes, even with high soil loss, grass buffers strips were able to trap >80% of the incoming bedload. Total Phosphorus, Total Nitrogen and suspended sediments were reduced by 25-65% within the first 15m of the grass buffer. However, loads leaving the buffer were often higher than those entering due to seepage as a result of prolonged or high frequency rainfall. During these conditions the function of the buffer is erosion control rather than a trap for sediment and nutrients. However, results show that riparian buffer strips on planar and moderately converging slopes could be effective at trapping nutrients and sediment in the extreme conditions of Far North Queensland although it is clear that trapping is more successful when infiltration occurs. In addition, dense grass riparian buffer strips <15m wide may be able to trap significant quantities of bedload if the area is maintained appropriately. On steep slopes, buffer strips would best be installed at the ends of crop rows, where contributing areas are smaller.

Several factors limit riparian performance in these conditions including exfiltration, flow channelization, scour and low vegetation density limit riparian buffer performance. The type of vegetation is also important as the riparian rainforest buffer was not successful and became a contributor of suspended solids as material was not permanently trapped and was released during subsequent runoff events. It was therefore concluded that rainforest buffers should contain a grass buffer upslope of a rainforest buffer.

The outcomes of this work are also supported by the information reported by Karssies and Prosser (1999) in Table 7.3.

7.6 Influence of residence times

The residence time of water in trapping mediums is an important measure of likely trapping effectiveness. Sufficient time allows sedimentation of particles with attached pollutants (and finer particles need most time), infiltration of water with contained dissolved pollutants, sorption of dissolved pollutants onto soil and vegetation and in some cases, for short half life pesticides, chemical breakdown of the pesticide. Residence times increase with treatment area but decrease with water volume and flow velocity. In grassed buffers infiltration may occur in minutes, sedimentation of coarse particles (sand) in minutes, sedimentation of silt in hours but sedimentation of fine sediment (clay) may require days. Chemical breakdown (by bacteria, light, etc) of most pesticides however will require weeks to years depending on the half life statistics of the particular chemical.

In high rainfall/runoff conditions (e.g. in the Wet Tropics) residence times for both surface and sub-surface water will be low and trapping will be limited as noted by McJannet *et al*. (CESE abstract and pers. com.) for trapping of nitrate in Kyambul lagoon on the Tully Murray floodplain (essentially no reduction in nitrate loads) and by Connor *et al*. (pers. com very early results) for riparian trapping of nitrate in the Mulgrave catchment.

7.7 Predictive models for estimating trapping efficiency

A number of modeling approaches have been developed to examine riparian trapping (Barling and Moore 1994; Newham, Rutherford et al. 2005; McJannet 2007; Fox and George 2009; Sabbagh, Fox et al. 2009; Winchell and Estes 2009; Arora, Mickelson et al. 2010; Page, Dillon et al. 2010). Predictive models exist that can be used to determine appropriate buffer widths. For example, Sabbagh *et al.* (2009; Sabbagh, Fox et al. 2010) have developed a predictive model that can be run under different physical and hydrological conditions. The model can also be combined with a pesticide exposure model developed by the US EPA (PRZM) which simulates pesticide fate and transport.

In this model, the empirical equations are based on runoff reduction / infiltration, sediment reduction, a phase distribution factor, and the percent clay content of the incoming sediment (Poletika, Coody et al. 2009; Sabbagh, Fox et al. 2009):

$$\blacktriangle P = a + b(\blacktriangle Q) + c(\blacktriangle E) + d\ln(Fph + 1) + e(\%C)$$

where $\blacktriangle P$ is the pesticide removal efficiency (%), $\blacktriangle Q$ is the infiltration (%) defined as the difference between total water input to the buffer (i.e., rainfall plus inflow runon) minus the runoff from the buffer, $\blacktriangle E$ is the sediment reduction (%), $\%C$ is the clay content of the sediment entering the buffer, F_{ph} is a phase distribution factor (i.e., ratio between the mass of pesticide in the dissolved phase relative to the mass of the pesticide sorbed to sediment), and *a*, *b*, *c*, *d*, and *e* are regression parameters (i.e., 24.8, 0.54, 0.53,-2.42, and -0.89, respectively) with $R^2 = 0.86$. Mathematically, F_{ph} was written as the following:

$$F_{ph} = Qi / K_d E_i$$

where Q_i and E_i are the volume of water (L) and mass of sediment (kg) entering the buffer, and K_d is the distribution coefficient defined as the product of the organic carbon sorption coefficient (K_{oc}), and the percent organic carbon in the soil, divided by 100 (Sabbagh et al., 2009). Parameters within this equation were used to represent some of the processes within the filter strip, including infiltration ($\blacktriangle Q$), sedimentation ($\blacktriangle E$), and sorption (F_{ph}). Degradation processes were not simulated in the buffer due to the assumption of a small residence time during typical rainfall runoff events. The focus was on immobilisation of the pesticide by the buffer due to the assumption that the most significant surface water loading threat was due to surface runoff in the immediate runoff event.

It is proposed that a model of this type could be used to predict the pesticide removal efficiency of buffers in the Study Area as part of the experimental design.

7.8 Nutrient removal in grassed buffers and forested riparian areas

Nitrogen

Most studies support the hypothesis that the primary mechanism for nitrate removal by riparian forests is denitrification. Denitrification is a process whereby nitrogen in the form of nitrate (NO_3^-) is converted to gaseous

NO_2 and N_2 and released into the atmosphere. In order for denitrification to occur, certain soil conditions must be present:

1. A high or perched water table;
2. Alternating periods of aerobic and anaerobic conditions;
3. Healthy populations of denitrifying bacteria; and
4. Sufficient amounts of available organic carbon (Lowrance, Leonard et al. 1985; Lowrance, Vellidis et al. 1995).

Denitrification offers an important means for the permanent removal of excess nitrogen from the riparian area because nitrates are converted to nitrogen gas and released to the atmosphere.

Other mechanisms for nitrate removal include uptake by vegetation and soil microbes and retention in riparian soils (Evanylo 1994; Beare 1997). Plants can take up large quantities of nitrogen as they produce roots, leaves, and stems. However, much of this is returned to the soil as plant materials decay. For example, scientists in Maryland estimated that deciduous riparian forests took up 69 pounds of nitrogen per acre annually, but returned 55 pounds (80 percent) each year in the litter (Peterjohn and Correll 1984). In North Carolina, researchers estimated that only 3 percent to 6 percent of the nitrogen passing through an alluvial swamp forest was taken up and stored in woody plant tissues (Brinson, Bradshaw et al. 1984). Nevertheless, Correll (1996) suggested that vegetative uptake is still a very important mechanism for removing nitrate from riparian systems, because vegetation (especially trees) removes nitrates from deep in the ground, converts the nitrate to organic nitrogen in plant tissues, then deposits the plant materials on the surface of the ground where the nitrogen can be mineralized and denitrified by soil microbes.

Grass buffers may also reduce nitrogen levels from agricultural runoff. For example, scientists in the Piedmont of North Carolina found that both grass and grass/forest riparian buffers reduced total nitrogen by 50 percent (Daniels and Gilliam 1996). On experimental plots at Blacksburg, Virginia, orchard grass buffers 30 feet wide reduced total nitrogen by 76 percent (Dillaha, Reneau et al. 1988). However, scientists in England reported that although both grass and forested buffers can effectively remove nitrogen, forested buffers may be more efficient (Haycock, Pinay et al. 1993). They found that a buffer of poplars adjacent to cereal croplands could remove 100 percent of the nitrate that entered the buffer, even in the dormant season, compared to a perennial ryegrass buffer which removed only 84 percent. They attributed the difference to the larger amount of carbon available year-round in the forested buffer. Likewise, a study in central Illinois comparing the ability of a mixed hardwood riparian forest and a reed canary grass filter strip to filter nutrients found that both were effective filters for nitrate-nitrogen, but on an annual basis, grass was less effective than the forest (Osborne and Kovacic 1993). The scientists suggest that this may be associated with the form of carbon available in the forested buffer for denitrification.

Current studies in the Ridge and Valley region of Pennsylvania suggest that neither grass nor forest provides a consistently more favourable environment for denitrification (Schnabel, Shaffer et al. 1997). Rather, it is the presence of certain soil and hydrological conditions which promote denitrification. However, their study confirmed the importance of carbon in fuelling denitrification processes; denitrification rates increased on both the grass and forested sites when they were amended with additional carbon. Likewise, studies conducted on Virginia's Eastern Shore by the U.S. Geological Survey suggest that the mere presence of forested buffers may not significantly decrease nitrogen loads to streams (Speiran and Commission 1996). Here, soil texture, organic matter content, and groundwater flow paths were reported to be the most important factors influencing the fate of nitrogen.

Phosphorus

Riparian areas can be important sinks for phosphorus; however, they are generally less effective in removing phosphorus than either sediment or nitrogen (Parson 1994). For example, only half the phosphorus entering a riparian forest in North Carolina was deposited within the forest (Cooper and Gilliam 1987). Lowrance (Lowrance, Todd et al. 1984) reported only a 30 percent reduction of phosphorus by a hardwood riparian forest in Georgia. Yet, in Maryland, scientists found that deciduous hardwood riparian buffers removed nearly 80 percent of the phosphorus from agricultural runoff, primarily particulate phosphorus (Peterjohn and Correll 1984). The riparian buffer had little effect on phosphorus in the form of dissolved phosphate.

The primary mechanism for phosphorus removal by riparian buffers is the deposition of phosphorus associated with sediments (Brinson, Bradshaw et al. 1984; Walbridge and Struthers 1993). In addition to the settling of particulate phosphorus, dissolved phosphorus may also be removed from runoff waters through adsorption by clay particles, particularly where there are soils containing clays with high levels of aluminium and iron (Cooper and Gilliam 1987). Some have suggested that because clays tend to accumulate in riparian soils, riparian areas play an important role in the removal of dissolved phosphorus (Walbridge and Struthers 1993). However, others have found that soils are limited in their capacity to adsorb large loads of phosphorus, and in areas where excessive phosphorus enrichment occurs; soils become saturated within a few years (Cooper and Gilliam 1987; Mozaffari and Sims 1994). Unlike nitrogen, phosphorus absorption is reduced in soils with high organic matter (Sharpley, Daniel et al. 1993; Walbridge and Struthers 1993).

Some phosphorus may be taken up and used by vegetation and soil microbes, but like nitrogen, much of this phosphorus is eventually returned to the soil. For example, researchers estimated that less than 3 percent of the phosphate entering a floodplain forest in eastern North Carolina was taken up and converted to woody tissue, while scientists in Maryland reported a deciduous riparian forest buffer took up 8.8 lb/A/yr phosphorus but returned 7 lb/A/yr (80 percent) as litter (Brinson, Bradshaw et al. 1984; Peterjohn and Correll 1984). In some riparian areas, small amounts of phosphorus (0.05-2.14 lb/A/yr) may be stored as peat (Walbridge and Struthers 1993).

Grass buffers may reduce phosphorus levels as well as forested buffers. Researchers in Illinois compared the ability of a mixed hardwood riparian forest and a grass filter strip to reduce phosphorus loads from agricultural runoff (Osborne and Kovacic 1993). They found that while the forest buffer removed more phosphorus initially, the forest buffer also released more phosphorus during the dormant season. On an annual basis, the grass buffer was a more efficient sink for phosphorus than was the forest buffer. Studies in the Coastal Plain of North Carolina suggest that grass buffers can reduce phosphorus loads by as much as 50 percent to 70 percent (Daniels and Gilliam 1996). Studies by Dillaha *et al.* (1988) at Virginia Tech reported similar results; orchardgrass buffer strips 30 feet wide removed 89 percent of the phosphorus from runoff, while filter strips 15 feet wide removed 61 percent. However, their research also suggests that grass buffers may only trap particulate phosphorus temporarily, then release it during later storm events.

8. Relevance to the Babinda Community Drainage Scheme

8.1 Theoretical functioning of trapping in riparian vegetation

The effectiveness of riparian vegetation in filtering pollutants depends on the nature of the pollutant. Retention of sediments is usually higher than retention of sediment-bound pollutants, because most sediment-bound pollutants are usually attached to finer particles which are more difficult to retain; and dissolved contaminants are reduced the least (Karssies and Prosser 1999). Riparian vegetation mainly filters sediments through the following mechanisms (Karssies and Prosser 1999; Mander, Hayakawa et al. 2005):

- by enhancing infiltration (i.e. reducing runoff volume) and increasing surface roughness (i.e. reducing runoff velocity), which favour sediment settling out – with effectiveness depending on many factors, such as rainfall characteristics and riparian topography;
- by protecting the stream banks and riparian soils from direct erosion;
- by filtering solid particles;
- by adsorbing pollutants onto the vegetation and the soil;
- by biologically taking up nutrients before they reach the watercourse.

Infiltration is by far the most important mechanism filtering incoming hillslope surface flows. However, when subsurface flows are sizeable, seepage and saturation flows can hinder infiltration (McKergow, Prosser et al. 2004a). In areas of high infiltration no real trapping of dissolved pollutants may be occurring (McKergow, Prosser et al. 2006).

The effectiveness of riparian vegetation in trapping sediments depends on many factors, such as incoming flow rates, sediment particle size, hydrologic and topographic settings of the riparian area, and vegetation cover and type (Karssies and Prosser 1999).

8.2 Effectiveness of different vegetation types

Density, height and type are the most important characteristics affecting the capacity of vegetation to retain sediments in riparian land (Karssies and Prosser 1999).

The density of the vegetation is important, particularly at ground surface, because the vegetation stems offer resistance to overland flow, thus reducing flow velocity and favouring particle settling. Vegetation should be uniformly dense; stoloniferous grasses (those spread by lateral stems, called stolons, which creep over the ground and give rise to new shoots along their length) and creeping grasses are the best, whereas tussocks may concentrate flow (Karssies and Prosser 1999). A minimum of 45 percent ground cover is recommended for effective buffers. Vegetation height should be at least 10 to 15 cm; it must be high enough to avoid submergence from overland flow – difficult in high rainfall areas.

The effect of vegetation type is more controversial. Grass may be more effective than woody vegetation in reducing bank erosion and trapping sediments, but grass requires active management because succession processes tend to favour woody vegetation (Lyons, Trimble and Paine, 2000). Grass filters colonize new sediments quickly so they are not removed by subsequent runoff; grass filters should be perennial, resistant to flooding and drought, able to grow after partial inundation, and not invasive of other ecosystems (Karssies and Prosser 1999).

Unless undergrowth is dense, forest is considered the least effective buffer because stems are dispersed and flow often gets concentrated into rills, thus becoming more erosive. Litter works only as a temporary store: it traps sediments, but these are flushed out by subsequent runoff (Karssies and Prosser 1999; McKergow, Prosser et al. 2004a; McKergow, Prosser et al. 2004b). However, trees and shrubs can provide other benefits to streams, such as shade and control of water temperature, which affect primary production and in-stream habitat (Lyons, Trimble et al. 2000). Forest should therefore be bordered by a grass strip to trap sediments from adjacent fields. For the southeastern United States, Sheridan et al (1999) recommended forest riparian buffers composed of three zones: a grass filter strip adjacent to fields, whose main function is to spread surface runoff as sheet flow; a first forested zone where infiltration and sedimentation occurs; and a second forested zone to protect and stabilize stream banks.

8.3 Critical features of riparian vegetation in trapping pollutants

From the literature it is clear that under suitable conditions vegetated buffers can trap high proportions (>50%) of pesticides from runoff, even for very soluble pesticides with low K_{OC} values. We will now explore the critical features of such buffers which allow them to work well and document conditions under which they are unlikely to work well.

From the discussion in Section 7 and above it is clear that critical parameters for effective trapping of pollutants in riparian vegetation, especially in high rainfall areas, include:

- Grass is generally better than trees but a mixture of both maybe the most effective.
- Suspended sediments are most easily trapped, especially the coarser fractions.
- Pollutants attached to particulate matter are not as easily trapped as the suspended sediments as most sediment-bound pollutants are usually attached to finer particles which are more difficult to retain.
- Dissolved materials are normally only trapped through infiltration and this may not be a ‘real’ trapping if the infiltrated pollutants make their way to the stream via sub-surface transport and no further pollutant loss occurs during this process.
- In very wet areas infiltration may not be able to occur where the water table is at or near the surface. Several factors limit riparian performance in these conditions including ex-filtration, flow channelization, scour and low vegetation density.
- Required widths for trapping dissolved materials in wet areas are generally in excess of 10m for buffers with dense and high grass cover and >30m with sparse grass cover.
- Required widths for trapping fine particulate matter (on which most sediment-bound pollutants are associated) in wet areas are also generally >10m for buffers with dense and high grass cover and >30m with sparse grass cover.
- Residence times of at least several days are required to trap fine particulate matter in riparian vegetation, vegetated drains or vegetated wetlands, times which are unlikely to occur in high rainfall areas.
- Pesticides strongly sorbed to fine particulate matter (i.e. generally with high K_{OC}) are likely to be much more effectively trapped than weakly sorbed types (low K_{OC}).
- Pesticides with low K_{OC} will also be less likely to be trapped when moving through the soil in sub-surface transport (e.g. in mole drains).

8.4 Babinda conditions and likely effectiveness of trapping of the selected herbicides

In our Babinda case study many conditions are quite different than for the locations and conditions of most of the reviewed literature. The main points of difference are as follows:

1. Extreme rainfall and runoff (Section 6). Rainfall is in the range of 3000 – 4000 mm in the area compared to rainfall of 500 – 1500 in other study areas.
2. High water table conditions (Section 6) preventing infiltration and potential leading to exfiltration.
3. Only relatively low K_{OC} value herbicides are considered in the study. Thus the herbicides of concern in this study (atrazine, diuron, ametryn and hexazinone) which are also reasonably water soluble will predominantly move in the dissolved phase rather than be particle bound.
4. Fast surface flows to streams and possibly relatively fast sub-surface flows as well (Section 6) thus transporting pesticides to stream even after infiltration.
5. Low slopes in the drainage scheme area (probably always <2%) compared to many of the reviewed study sites where slopes tested ranged from 0 to 20%.

Even in the period of most risk for pesticide loss from the paddock i.e. the application period from about July to November, rainfall and runoff can be high (Section 6). For pesticides with low K_{OC}, infiltration is more likely to ‘remove’ pesticides than sedimentation. Other pesticides used in sugarcane like chlorpyrifos and paraquat will be more likely to be trapped by sediment retention but are not the focus of this study.

The combination of these factors means that in Babinda buffer widths for effective trapping of these pollutants will need to be in excess of 10m (with good grass cover) and in some circumstances (poorer grass cover) in excess of 30m. In addition, for these particular herbicides, while infiltration may occur they may be no further trapping on infiltrated soil and effective sub-surface transport to the closest drain or stream is likely (albeit in some longer timeframe than for surface transport).

Another factor mitigating against effective trapping through infiltration in the BCDS is the generally high water table (Section 6) which prevents infiltration and may instead promote exfiltration, returning previous upslope drainage to surface flow, as seen in the Johnstone catchment studies of McKergow et al. (2004a; McKergow, Prosser et al. 2004b).

As yet unpublished studies of Connor and others in the Mulgrave catchment, and McJannet and others in the Tully-Murray catchment, also show that with the extreme rainfall and hydrological conditions present in these areas (similar to Babinda) trapping of sub-surface flow nitrate (a dissolved phase pollutant) through riparian areas or small wetlands is minimal. These findings are relevant for the dissolved phase pollutants of our study.

Overall it is likely that low proportions (< 10%) of these dissolved phase herbicides will be trapped in 5m grassed buffer strips in any flow conditions in the Babinda situation and this will occur mostly through infiltration. Trapping may improve to perhaps 30% where buffer widths are increased to 20m but will still mainly occur by infiltration. The final fate of the infiltrated pesticides is unclear but it is quite possible that transport to an adjacent drain or stream could be rapid with little further loss of pesticide, thus minimizing any net trapping (McKergow, Prosser et al. 2006). If real input data from a Babinda site could be used with one of the models discussed in Section 7, a quantitative assessment of likely trapping of herbicides with EVTAs could be made.

For the assessment of the BCDS as an herbicide trapping mechanism, it is unlikely that any major trapping of dissolved phase, low K_{OC} pollutants will occur in the mole drains and even less in the sub-surface piped drains. In the major drainage network, no infiltration can occur due to the high water table but some sedimentation (but mostly for particle bound pesticides – not the ones in our study) could occur if residence times were long enough – once again in the order of 5 days or longer. In reality in the 10km length of the drainage, preferred flow path water residence times in low flow conditions will be in the order of < 1 day (at a water velocity of 0.5 m/sec – 6 hours). So minimal trapping of these herbicides is likely in the drainage scheme in Babinda conditions.

8.5 Other Queensland cane growing regions

Despite the conclusions relevant to the BCDS, trapping of herbicides using EVTAs may be possible in other sugarcane growing regions of the GBR catchments. In particular, EVTAs and vegetated drains may work effectively in irrigation tailwater and small first flush conditions in the lower Burdekin. Similarly in the less intense rainfall conditions of the Mackay Whitsunday and Burnett Mary cane growing regions, trapping in first flush conditions may occur. However, further studies are required to quantify these suggestions.

9. Experimental design

The requirements for the proposed survey design were stated in the contract documentation and are included below.

Aim of survey:

- What is the effectiveness of all the components of the BCDS and EVTAs in trapping atrazine, ametryn, hexazinone and diuron and/ their metabolites?
- An overall risk assessment can be made of the effectiveness of EVTAs for the trapping/removal of ametryn, atrazine, hexazinone and diuron in the specific rainfall/runoff and hydrological conditions in Babinda, North Queensland versus the removal of the above active constituents from waters entering a natural water body by the components of the BCDS.
- The study should be compatible with and add value to related monitoring, modelling and management practice effectiveness studies in the Reef Plan Paddock to Reef Program.
- Proposed survey design must also include:
    - Details of proposed methodology (include sampling points and times for 1 year; experience of key personnel and support staff, ability to provide delivery within one year).
    - Schematic mud map of sample points.
    - Analysis of the transferability of the project methodology and potential findings to other community or regional drainage schemes.
    - Proposed costings and timeframes and milestone deliverables for the research project.

9.1 Initial thoughts on experimental design

The design for testing the EVTAs is likely to be very different to that for the BCDS. We believe testing the EVTAs is best done through an experimental constructed arrangement with fluming (Figure 9.2) and automatic sampling, lysimeters for leachate and a number of paired treatments with, for example, different widths, different grass cover and possibly different soil types (see Figure 9.3 for a typical layout used elsewhere). Rainfall simulators could also be used (as in the Reef Plan Paddock to Reef Integrated Monitoring, Modelling and Reporting Program) (Figure 9.1) although given the frequent rainfall/runoff events in Babinda natural rainfall systems may be adequate. A range of rainfall types and timings will be required especially if ‘natural’ pesticide runoff is used in the tests rather than deliberately applied for the experiment. Concentrations of pesticides in runoff are likely to be quite different (higher) in the ‘drier’ months (May to November) which coincide with pesticide application than in the ‘wetter’ months when less application occurs and greater dilution is likely.

Ultimately a completely artificial system is also possible such as those shown in Figure 9.4 and 9.5 where all parameters can be more completely controlled, leachate and runoff easily collected and various pre-selected concentrations of the pesticides trialled.

For testing the BCDS, a series of grab sampling sites positioned progressively through the drainage system would be used. Runoff from furrows would be sampled along with discharge from sub-surface pipes, small drains, major drains and the final entry point to Babinda Creek/Russell River. Only actual rainfall driven events would be sampled in a variety of circumstances but certainly including significant events in the May – November period (the ‘dry’) as well as January to April (the wet). A combination of automatic sampling and manual sampling is envisaged and passive samplers would be considered.

Further detail of the survey design including selection of study sites, costings, timeframes and milestone deliverables would be undertaken in consultation with DERM and the local community following further advice from DERM on the requirements for the scope of the survey and availability of resources.

Figure 9-1. Rainfall simulator use in testing riparian trapping (from Hairsine 1997).

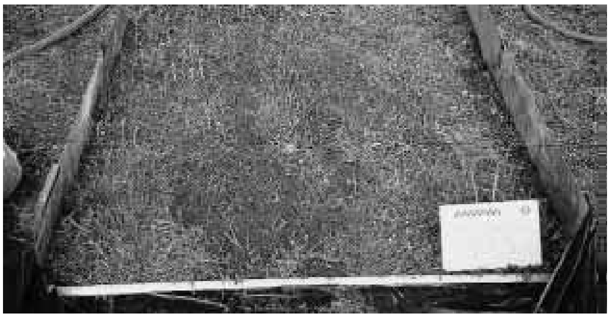


Figure 9-2. Sediment fans formed in grass filter strips in the Tarago catchment experiments (from Hairsine 1997).

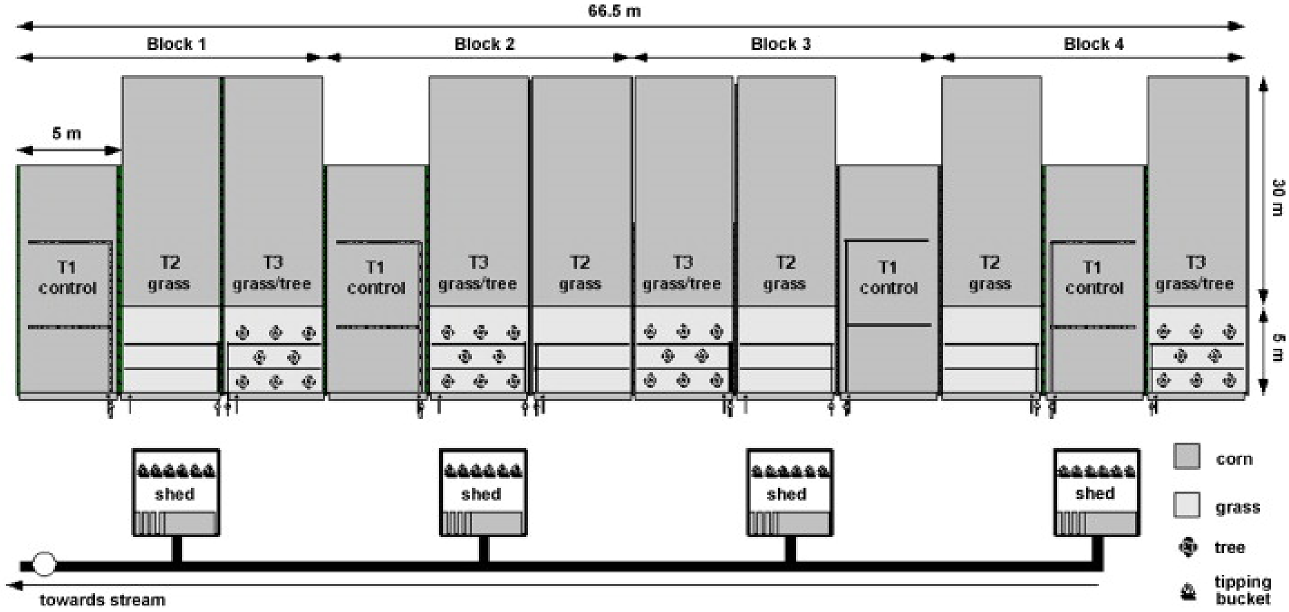


Figure 9-3. Typical layout of paired trials in a actual paddock situation (Experimental set-up consisting of four blocks, each with three treatment plots). Source: Duchemin and Hogue (2009).

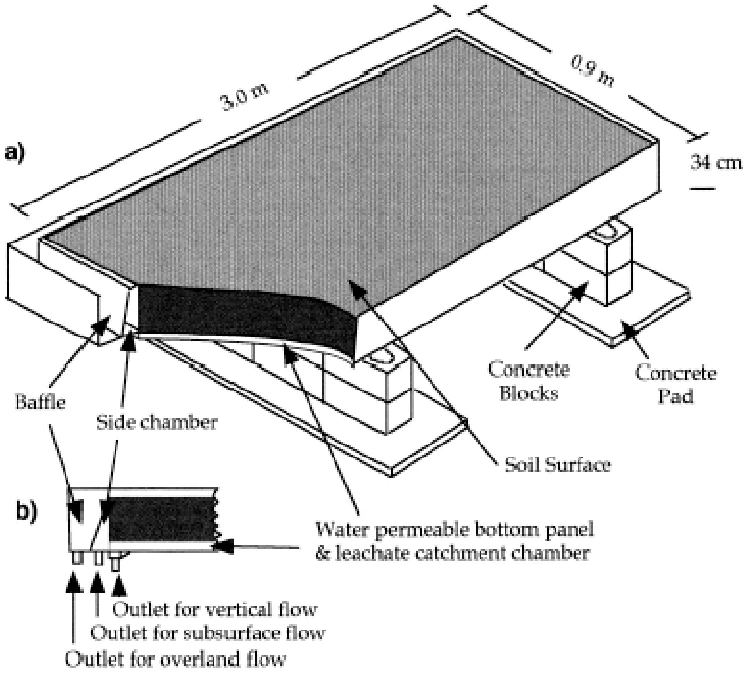


Figure 9-4. Diagram of a) tilted-bed setup and b) side view catchment and drain area of the tilted bed. Source: Seybold *et al*. (2001).

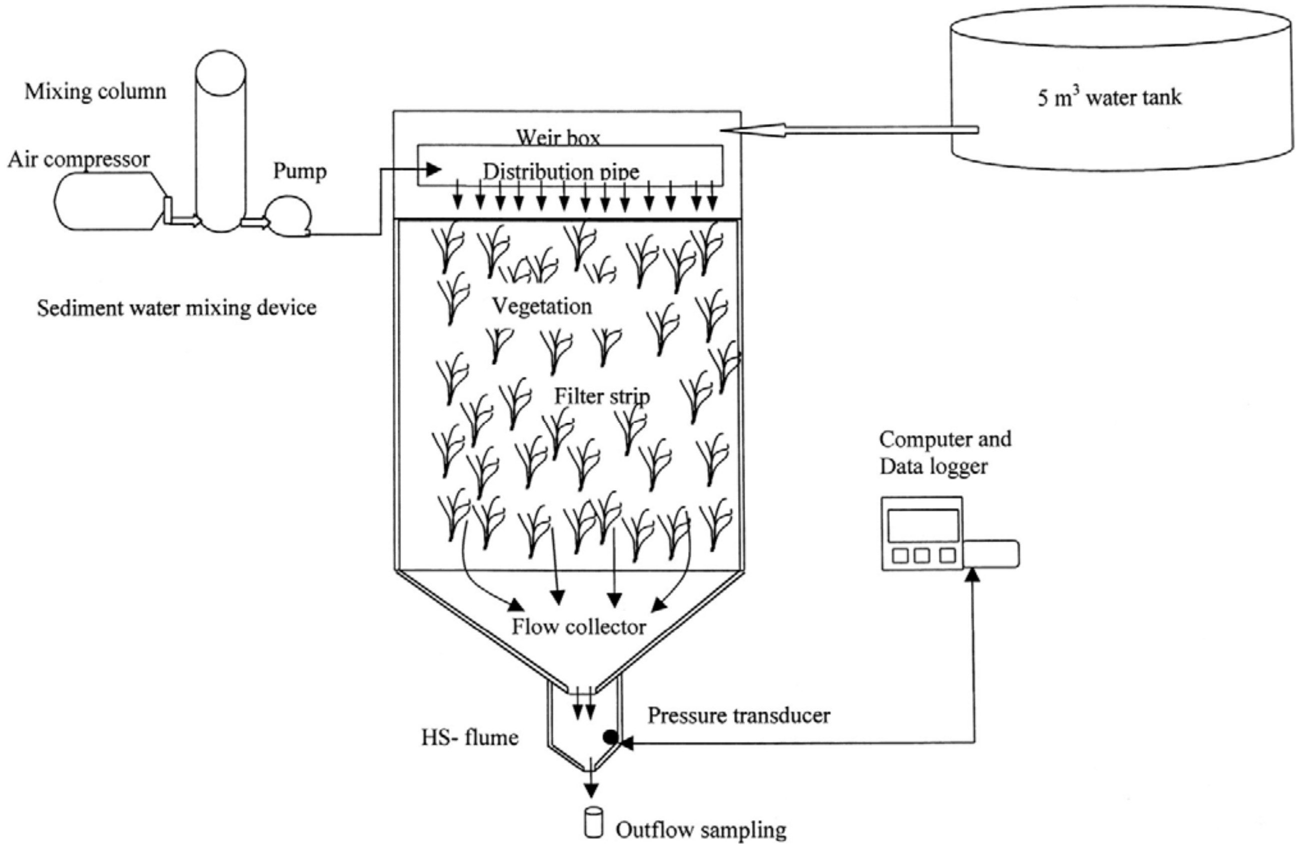


Figure 9-5. Schematic diagram of the experimental supply and monitoring system for artificial runoff. Source: Abu-Zreig *et al.* (2003).

10. References

Abu-Zreig, M. R., R. P. Whiteley, et al. (2003). "Phosphorus removal in vegetated filter strips." Journal of Environmental Quality **32**(2): 613.

Arora, K., S. Mickelson, et al. (1996). "Herbicide retention by vegetative buffer strips from runoff under natural rainfall." American Society of Agricultural Engineers **39**(6): 2155-2162.

Arora, K., S. K. Mickelson, et al. (2010). "Review of pesticide retention processes occurring in buffer strips recieving agricultural runoff." Journal of the American Water Resources Association **FIX**.

Asmussen, L., A. Hauser, et al. (1977). "Reduction of 2, 4-D Load in Surface Runoff Down a Grassed Waterway." Journal of Environmental Quality **6**(2): 159.

Barling, R. D. and I. D. Moore (1994). "Role of buffer strips in management of waterway pollution: a review." Environmental Management **18**(4): 543-558.

Bartkow, M., A. Dunn, et al. (2008). Monitoring of organic chemicals in the Great Barrier Reef Marine Park and selected tributaries using time integrated monitoring tools, National Research Centre for Environmental Toxicology (EnTox). **Report to the Great Barrier Marine Park Authority (Marine Monitoring Programme)**.

Beare, M. (1997). "Fungal and bacterial pathways of organic matter decomposition and nitrogen mineralization in arable soils."

Bell, A. M. and N. C. Duke (2005). "Effects of Photosystem II inhibiting herbicides on mangroves--preliminary toxicology trials." Marine Pollution Bulletin **51**(1-4): 297-307.

Bengtson Nash, S., P. Quayle, et al. (2005). "The selection of a model microalgal species as biomaterial for a novel aquatic phytotoxicity assay." Aquatic Toxicology **72**(4): 315-326.

Blankenberg, A. G. B., B. Braskerud, et al. (2006). "Pesticide retention in two small constructed wetlands: treating non-point source pollution from agriculture runoff." International Journal of Environmental Analytical Chemistry **86**(3-4): 225-231.

Bouldin, J. L., J. L. Farris, et al. (2005). "Evaluated fate and effects of atrazine and lambda cyhalothrin in vegetated and unvegetated microcosms." Environmental Toxicology **20**(5): 487-498.

Boutron, O., C. Margoum, et al. (2011). "Effect of the submergence, the bed form geometry and the speed of the surface water flow on the mitigation of the pesticides in agricultural ditches." Water Resources Research **47**, W08505, doi:10.1029/2011WR010378.

Boyd, P. M., J. L. Baker, et al. (2003). "Pesticide transport with surface runoff and subsurface drainage through a vegetative filter strip." American Society of Agricultural Engineers **46**(3): 675-684.

Brinson, M. M., H. D. Bradshaw, et al. (1984). "Nutrient assimilative capacity of an alluvial floodplain swamp." Journal of Applied Ecology **21**(3): 1041-1057.

Brodie, J. (2007). Nutrient management zones. ACTFR Report, ACTFR.

Brønnum, R., S. Jørgensen, et al. (2004). "Occurence and degradation of alachlor and nitrate in a created wetland in central Ohio."

Budd, R., A. O'Geen, et al. (2009). "Efficacy of constructed wetlands in pesticide removal from tailwaters in the central valley, California." Environmental Science & Technology **43**(8): 2925-2930.

Cantin, N. E., A. P. Negri, et al. (2007). "Photoinhibition from chronic herbicide exposure reduces reproductive output of reef building corals." Marine Ecology Progress Series **344**: 81-93.

Caron, E., P. Lafrance, et al. (2010). "Impact of Grass and Grass with Poplar Buffer Strips on Atrazine and Metolachlor Losses in Surface Runoff and Subsurface Infiltration from Agricultural Plots." Journal of Environmental Quality **39**(2): 617-629.

Connolly, N. M., B. Pearson, et al. (2007). Hydrology, Geomorphology and Water Quality of Four Wet Tropics Streams with Contrasting Land-use Management. Biological Indicators of Ecosystem Health in Wet Tropics Streams. Final Report, Task 3 Catchment to Reef Research Program, Cooperative Research Centre for Rainforest Ecology and Management and Cooperative Research Centre for the Great Barrier Reef World Heritage Area. **3:** 14-78.

Cooper, J. and J. Gilliam (1987). "Phosphorus redistribution from cultivated fields into riparian areas." Soil Science Society of America Journal **51**(6): 1600.

Correll, D. (1996). "Buffer zones and water quality protection: general principles." Buffer Zones: 7.

Daniels, R. and J. Gilliam (1996). "Sediment and chemical load reduction by grass and riparian filters." Soil Science Society of America Journal **60**(1): 246-251.

Dawson, H. (1997). Land use practices for wet tropical floodplains. South Johnstone, Department of Natural Resources: Centre for Wet Tropics Agriculture**:** 1-42.

Devlin, M. J. and J. Brodie (2005). "Terrestrial discharge into the Great Barrier Reef Lagoon: nutrient behavior in coastal waters." Marine Pollution Bulletin **51**(1-4): 9-22.

Devlin, M. J., L. McKinna, et al. (2010). Flood plume monitoring 2009/10. Reef Rescue Marine Monitoring Program. James Cook University Townsville, ACTFR.

Dillaha, T., R. Reneau, et al. (1988). "Vegetative filter strips for agricultural nonpoint source pollution control." work.

Dosskey, M. G. (2001). "Toward quantifying water pollution abatement in response to installing buffers on crop land." Environmental Management **28**(5): 577-598.

Dousset, S., M. Thévenot, et al. (2010). "Effect of grass cover on water and pesticide transport through undisturbed soil columns, comparison with field study (Morcille watershed, Beaujolais)." Environmental Pollution **158**(7): 2446-2453.

Duchemin, M. and R. Hogue (2009). "Reduction in agricultural non-point source pollution in the first year following establishment of an integrated grass/tree filter strip system in southern Quebec (Canada)." Agriculture, Ecosystems & Environment **131**(1-2): 85-97.

Elsenbeer, H., A. West, et al. (1994). "Hydrologic pathways and stormflow hydrochemistry at South Creek, northeast Queensland." Journal of Hydrology **162**(1-2): 1-21.

Evanylo, G. (1994). "Mineralization and availability of nitrogen in organic wasteamended mid-Atlantic soils." Perspectives on Chesapeake Bay, 1994: Advances in estuarine sciences: 77–104.

Fares, A. and M. Ryder (2006). Performance of vegetative filters to control loadings of sediment and nutrients in to surface water bodies in a Hawaiin watershed. 14th International soil conservation organization conference, Marrakech.

Fox, G. A., R. Muñoz-Carpena, et al. (2010). "Influence of flow concentration on parameter importance and prediction uncertainty of pesticide trapping by vegetative filter strips." Journal of Hydrology **384**(1-2): 164-173.

Fox, G. A. S. and J. George (2009). "Comment on “Major Factors Influencing the Efficacy of Vegetated Buffers on Sediment Trapping: A Review and Analysis,” by Xingmei Liu, Xuyang Zhang, and Minghua Zhang in the 2008 37: 1667–1674." Journal of Environmental Quality **38**(1): 1.

Furnas, M. J. (2003). Catchments and corals: terrestrial runoff to the Great Barrier Reef, Australian Institute of Marine Science.

Gregoire, C., D. Elsaesser, et al. (2009). "Mitigation of Agricultural Nonpoint-Source Pesticide Pollution in Artificial Wetland Ecosystems–A Review." Climate Change, Intercropping, Pest Control and Beneficial Microorganisms: 293-338.

Grismer, M. E., A. T. O'Geen, et al. (2006). Vegetative filter strips for nonpoint source pollution control in agriculture, ANR Publications.

Haarstad, K., B. Braskerud, et al. (2005). "Pesticide retention in the watershed and in a small constructed wetland treating diffuse pollution." Water Science & Technology **51**(3-4): 143-150.

Hairsine, P. (1997). "Controlling sediment and nutrient movement within catchments." Industry report, Cooperative Research Centre for Hydrology, Melbourne.

Hamilton, D. and G. Haydon (1996). Pesticides and fertilisers in the Queensland sugar industry: estimates of usage and likely environmental fate. Brisbane, Queensland Department of Primary Industries. **1: Pesticides**.

Harrington, L., K. Fabricius, et al. (2005). "Synergistic effects of diuron and sedimentation on photosynthesis and survival of crustose coralline algae." Marine Pollution Bulletin **51**(1-4): 415-427.

Hatfield, J. L., S. K. Mickelson, et al. (1995). Buffer strips: landscape modifications to reduce off-site herbicide movement. Clean water - Clean environment - 21st Century Team Agriculture - Working to Protect Water Resources, Kansas City, Missouri.

Haycock, N. E., G. Pinay, et al. (1993). "Nitrogen retention in river corridors: European perspective." Ambio. Stockholm **22**(6): 340-346.

Haynes, D., P. Ralph, et al. (2000). "The Impact of the Herbicide Diuron on Photosynthesis in Three Species of Tropical Seagrass." Marine Pollution Bulletin **41**(7-12): 288-293.

Helmers, M. J., T. M. Isenhart, et al. (2008). Buffers and vegetative filter strips. Trans. ASAE. **48:** 541-555.

Hoffmann, C. C., C. Kjaergaard, et al. (2009). "Phosphorus retention in riparian buffers: Review of their efficiency." Journal of Environmental Quality **38**(5): 1942-1955.

Hook, P. B. (2003). "Sediment retention in rangeland riparian buffers." Journal of Environmental Quality **32**(3): 1130.

Hunter, H. and G. Lukacs (2000). Nutrient control in irrigation drainage systems using artificial wetlands**:** 1-13.

Isbell, R. F. (1993). A classification system for Australian soils (Third approximation). Technical Report, CSIRO Australian Division of Soils.

Jin, C. X. and M. J. M. Römkens (2001). "Experimental studies of factors in determining sediment trapping in vegetative filter strips." Transactions of the ASAE **44**(2): 277-288.

Johnson, A. K. L. and S. P. Ebert (2000). "Quantifying Inputs of Pesticides to the Great Barrier Reef Marine Park - A Case Study in the Herbert River Catchment of North-East Queensland." Marine Pollution Bulletin **41**(7-12): 302-309.

Jones, R. (2005). "The ecotoxicological effects of Photosystem II herbicides on corals." Marine Pollution Bulletin **51**(5-7): 495-506.

Jones, R. J. and A. P. Kerswell (2003). "Phytotoxicity of Photosystem II(PSII) herbicides to coral." Marine Ecology Progress Series **261**: 149-159.

Jones, R. J., J. Muller, et al. (2003). "Effects of herbicides diuron and atrazine on corals of the Great Barrier Reef, Australia." Marine Ecology Progress Series **251**: 153-167.

Kao, C. M., J. Y. Wang, et al. (2002). "Non-point source pesticide removal by a mountainous wetland." Water Science and Technology: A Journal of the International Association on Water Pollution Research **46**(6-7): 199-206.

Kapernick, A., M. Shaw, et al. (2007). River pesticide loads and GBR lagoon pesticide data (2006-2007), National Research Centre for Environmental Toxicology (EnTox). **Report to the Great Barrier Reef Marine Park Authority (Marine Monitoring Programme):** 66-104.

Karssies, L. E. and I. P. Prosser (1999). Guidelines for riparian filter strips for Queensland irrigators, CSIRO Land and Water Canberra.

Klöppel, H., W. Kördel, et al. (1997). "Herbicide transport by surface runoff and herbicide retention in a filter strip: Rainfall and runoff simulation studies." Chemosphere **35**(1-2): 129-141.

Krutz, L. J., S. A. Senseman, et al. (2003a). "Infiltration and adsorption of dissolved atrazine and atrazine metabolites in buffalograss filter strips." Journal of Environmental Quality **32**(6): 2319.

Krutz, L. J., S. A. Senseman, et al. (2004). "Infiltration and adsorption of dissolved metolachlor, metolachlor oxanilic acid, and metolachlor ethanesulfonic acid by buffalograss (Buchloe dactyloides) filter strips." Weed Science: 166-171.

Krutz, L. J., S. A. Senseman, et al. (2003b). "Adsorption and desorption of atrazine, desethylatrazine, deisopropylatrazine, and hydroxyatrazine in vegetated filter strip and cultivated soil." Journal of Agricultural and Food Chemistry **51**(25): 7379-7384.

Krutz, L. J., S. A. Senseman, et al. (2005). "Reducing herbicide runoff from agricultural fields with vegetative filter strips: a review." Weed Science **53**: 353-367.

Lewis, S. E., J. E. Brodie, et al. (2009). "Herbicides: A new threat to the Great Barrier Reef." Environmental Pollution **157**(8-9): 2470-2484.

Lin, C., R. Lerch, et al. (2004). "Incorporating forage grasses in riparian buffers for bioremediation of atrazine, isoxaflutole and nitrate in Missouri." Agroforestry Systems **63**(1): 91-99.

Liu, X., X. Zhang, et al. (2008). "Major factors influencing the efficacy of vegetated buffers on sediment trapping: A review and analysis." Journal of Environmental Quality **37**(5): 1667-1674.

Lizotte, R. E., F. D. Shields, et al. (2009). "Efficiency of a modified backwater wetland in trapping a pesticide mixture." Ecohydrology **2**(3): 287-293.

Loch, R. J., T. Espigares, et al. (1999). "Vegetative filter strips to control sediment movement in forest plantations: validation of a simple model using field data." Australian Journal of Soil Research **37**(5): 929-946.

Lovett, S. and P. Price (2001). "Managing riparian lands in the sugar industry: a guide to principles and practices. Sugar Research & Development Corporation." Land & Water Australia, Brisbane. ISBN 0-9579313-0-1.

Lowrance, R., R. Leonard, et al. (1985). "Managing riparian ecosystems to control nonpoint pollution." Journal of soil and water conservation **40**(1): 87.

Lowrance, R., R. Todd, et al. (1984). "Riparian forests as nutrient filters in agricultural watersheds." BioScience **34**(6): 374-377.

Lowrance, R., G. Vellidis, et al. (1995). "Denitrification in a restored riparian forest wetland." Journal of Environmental Quality **24**(5): 808-815.

Lowrance, R., G. Vellidis, et al. (1997). "Herbicide transport in a managed riparian forest buffer system." American Society of Agricultural Engineers **40**(4): 1047-1057.

Lyons, J., S. W. Trimble, et al. (2000). "Grass versus trees: managing riparian areas to benefit streams of central North America." Journal of the American Water Resources Association **36**(4): 919-930.

Magnussen, M., K. Heimann, et al. (In press). "Additive toxicity of herbicide mixtures and comparative sensitivity of tropical benthic microalgae." Marine Pollution Bulletin.

Magnusson, M., K. Heimann, et al. (2008). "Comparative effects of herbicides on photosynthesis and growth of tropical estuarine microalgae." Marine Pollution Bulletin **56**(9): 1545-1552.

Mander, U., Y. Hayakawa, et al. (2005). "Purification processes, ecological functions, planning and design of riparian buffer zones in agricultural watersheds." Ecological Engineering **24**(5): 421-432.

McJannet, D. (2007). Towards an understanding of the filter function of tropical wetlands, CSIRO.

McKergow, L. A., I. P. Prosser, et al. (2004a). "Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland. 1. Riparian hydrology." Australian Journal of Soil Research **42**(4): 473-484.

McKergow, L. A., I. P. Prosser, et al. (2004b). "Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland. 2. Water quality." Australian Journal of Soil Research **42**(4): 485-498.

McKergow, L. A., I. P. Prosser, et al. (2006). "Performance of grass and eucalyptus riparian buffers in a pasture catchment, Western Australia, part 2: water quality." Hydrological Processes **20**(11): 2327-2346.

Mersie, W., C. A. Seybold, et al. (2003). "Abating endosulfan from runoff using vegetative filter strips: the importance of plant species and flow rate." Agriculture, Ecosystems & Environment **97**(1-3): 215-223.

Mickelson, S., J. Baker, et al. (2003). "Vegetative filter strips for reducing atrazine and sediment runoff transport." Journal of Soil and Water Conservation **58**(6): 359.

Mickelson, S. K., M. Helmers, et al. (2004). "Vegetative Filter Education and Assessment in the State of Iowa." 2004IA63B.

Mitsch, W. I. J. (1992). "Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution." Ecological Engineering **1**(1-2): 27-47.

Moore, M., D. Denton, et al. (2008). "Mitigation assessment of vegetated drainage ditches for collecting irrigation runoff in California." Journal of Environmental Quality **37**(2): 486-493.

Mozaffari, M. and J. Sims (1994). "Phosphorus availability and sorption in an Atlantic Coastal Plain watershed dominated by animal-based agriculture." Soil Science **157**(2): 97.

Murtha, G. G., M. G. Cannon, et al. (1996). Soils of the Babinda-Cairns Area, North Queensland. Divisional Report, CSIRO Division of soils**:** 79.

Negri, A., C. Vollhardt, et al. (2005). "Effects of the herbicide diuron on the early life history stages of coral." Marine Pollution Bulletin **51**(1-4): 370-383.

Newham, L., J. Rutherford, et al. (2005). A conceptual model of particulate trapping in riparian buffers, CSIRO Land and Water Technical Report.

Nguyen, L., M. Downes, et al. (1999). Riparian wetland processing of nitrogen, phosphorus and suspended sediment inputs from hill country sheep-grazed catchment in New Zealand. Second Australian Stream Management Conference, Adelaide.

Norris, V. (1993). "The use of buffer zones to protect water quality: a review." Water Resources Management **7**(4): 257-272.

Northcote, K. H. (1979). A factual key for the recognition of Australian soils. Glenside, SA, Rellim.

Osborne, L. L. and D. A. Kovacic (1993). "Riparian vegetated buffer strips in water-quality restoration and stream management." Freshwater Biology **29**(2): 243-258.

Otto, S., M. Vianello, et al. (2008). "Effect of a full-grown vegetative filter strip on herbicide runoff: Maintaining of filter capacity over time." Chemosphere **71**(1): 74-82.

Page, D., P. Dillon, et al. (2010). "Quantification of herbicide removal in a constructed wetland using passive samplers and composite water quality monitoring." Chemosphere.

Parson, J. E. (1994). Reduction in Sediment and Chemical Load: Agricultural Field Runoff by Vegetative Filter Strips, Water Resources Research Institute of the University of North Carolina.

Patty, L., B. Réal, et al. (1997). "The use of grassed buffer strips to remove pesticides, nitrate and soluble phosphorus compounds from runoff water." Pesticide Science **49**(3): 243-251.

Pätzold, S., C. Klein, et al. (2007). "Run off transport of herbicides during natural and simulated rainfall and its reduction by vegetated filter strips." Soil Use and Management **23**(3): 294-305.

Peterjohn, W. T. and D. L. Correll (1984). "Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest." Ecology **65**(5): 1466-1475.

Pinho, A. P., L. A. Morris, et al. (2008). "Contaminant retention potential of forested filter strips established as SMZs in the Piedmont of Georgia." Journal of the American Water Resources Association **44**(6): 1564-1577.

Poletika, N., P. Coody, et al. (2009). "Chlorpyrifos and atrazine removal from runoff by vegetated filter strips: Experiments and predictive modeling." Journal of Environmental Quality **38**(3): 1042-1052.

Polyakov, V., A. Fares, et al. (2005). "Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: A review." Environmental Reviews **13**(3): 129-144.

Popov, V. H. and P. Cornish (2006). "Atrazine tolerance of grass species with potential for use in vegetated filters in Australia." Plant and Soil **280**(1): 115-126.

Popov, V. H., P. Cornish, et al. (2006). "Vegetated biofilters: the relative importance of infiltration and adsorption in reducing loads of water-soluble herbicides in agricultural runoff." Agriculture, Ecosystems & Environment **114**(2-4): 351-359.

Pot, V., J. Simunek, et al. (2005). "Impact of rainfall intensity on the transport of two herbicides in undisturbed grassed filter strip soil cores." Journal of Contaminant Hydrology **81**(1-4): 63-88.

Prange, J., D. Haynes, et al. (2007). Great Barrier Reef water quality protection plan: Annual marine monitoring report. Townsville, Great Barrier Reef Marine Park Authority.

Qiu, Z. and T. Prato (1998). "Economic evaluation of riparian buffers in an agricultural watershed." Journal of the American Water Resources Association **34**(4): 877-890.

QLUMP (1999). Queensland Land Use Mapping Project, Department of Natural Resources.

QLUMP (2004). Queensland Land Use Mappping Project, Department of Natural Resources.

Ralph, P. J. (2000). "Herbicide toxicity of Halophila ovalis assessed by chlorophyll a fluorescence." Aquatic Botany **66**(2): 141-152.

Rankins Jr, A., D. R. Shaw, et al. (2001). "Perennial grass filter strips for reducing herbicide losses in runoff." Weed Science **49**(5): 647-651.

Rankins Jr, A., D. R. Shaw, et al. (2005). "Response of perennial grasses potentially used as filter strips to selected postemergence herbicides." Weed Technology: 73-77.

Reichenberger, S., M. Bach, et al. (2007). "Mitigation strategies to reduce pesticide inputs into ground-and surface water and their effectiveness; A review." Science of the Total Environment **384**(1-3): 1-35.

Reungsang, A., T. B. Moorman, et al. (2001). "Transport and fate of atrazine in midwestern riparian buffer strips." Journal of the American Water Resources Association **37**(6): 1681-1692.

Robinson, C., M. Ghaffarzadeh, et al. (1996). "Vegetative filter strip effects on sediment concentration in cropland runoff." Journal of Soil and Water Conservation **51**(3): 227-230.

Rose, M. T., A. N. Crossan, et al. (2008). "The effect of vegetation on pesticide dissipation from ponded treatment wetlands: Quantification using a simple model." Chemosphere **72**(7): 999-1005.

Rose, M. T., F. Sanchez-Bayo, et al. (2006). "Pesticide removal from cotton farm tailwater by a pilot-scale ponded wetland." Chemosphere **63**(11): 1849-1858.

Runes, H. B., J. J. Jenkins, et al. (2003). "Treatment of atrazine in nursery irrigation runoff by a constructed wetland." Water Research **37**(3): 539-550.

Sabbagh, G., G. Fox, et al. (2009). "Effectiveness of vegetative filter strips in reducing pesticide loading: Quantifying pesticide trapping efficiency." Journal of Environmental Quality **38**(2): 762-771.

Sabbagh, G. J., G. A. Fox, et al. (2010). "Revised framework for pesticide aquatic environmental exposure assessment that accounts for vegetative filter strips." Environmental Science & Technology **44**(10): 3839-3845.

Schmitt, T., M. Dosskey, et al. (1999). "Filter strip performance and processes for different vegetation, widths, and contaminants." Journal of Environmental Quality **28**(5): 1479-1489.

Schnabel, R. R., J. A. Shaffer, et al. (1997). "Denitrification distributions in four valley and ridge riparian ecosystems." Environmental Management **21**(2): 283-290.

Seery, C. R., L. Gunthorpe, et al. (2006). "Herbicide impact on Hormosira banksii gametes measured by fluorescence and germination bioassays." Environmental Pollution **140**(1): 43-51.

Selim, H., L. Zhou, et al. (2003). "Herbicide retention in soil as affected by sugarcane mulch residue." Journal of Environmental Quality **32**: 1445–1454.

Selim, H. M., B. J. Naquin, et al. (2004). "Herbicide retention and runoff losses as affected by sugarcane mulch residue." Louisiana Agricultural Experiment Station Bulletin **Bulletin Number 883**: 43.

Seybold, C., W. Mersie, et al. (2001). "Removal and degradation of atrazine and metolachlor by vegetative filter strips on clay loam soil." Communications in Soil Science and Plant Analysis **32**(5): 723-737.

Sharpley, A. N., T. Daniel, et al. (1993). "Phosphorus movement in the landscape." Journal of Production Agriculture **6**(4): 492-500.

Shaw, M., M. J. Furnas, et al. (2010). "Monitoring pesticides in the Great Barrier Reef." Marine Polution Bulletin **60**(1): 113-122.

Sheridan, J., R. Lowrance, et al. (1999). "Management effects on runoff and sediment transport in riparian forest buffers." Transactions of the ASAE-American Society of Agricultural Engineers **42**(1): 55-64.

Shutes, R. B. E. (2001). "Artificial wetlands and water quality improvement." Environment International **26**(5-6): 441-447.

Smith, M., S. Melvin, et al. (1992). Vegetative filter strips for improved surface water quality. D. Melchert, Iowa State University: University Extension.

Smith, R. (2008). SmartCane riparian and wetland areas on cane farms: SmartCane best management practice booklet, BSES Limited**:** 1-44.

Snyder, C. S. (1998). Vegetative filter strips reduce runoff losses and help protect water quality. New and views, Potash and Phosphate Institute and the Potash and Phosphate Institute of Canada.

Spatz, R. (1999). Rückhaltevermögen von Puff erstreifen für pfl anzenschutzmittelbelasteten Oberfl ächenabfl uss. Diss. Shaker Verlag, Aachen, Institut für Phytomedizin, Universität Hohenheim**:** 176.

Speiran, G. K. and A.-N. P. D. Commission (1996). Geohydrology and geochemistry near coastal ground-water-discharge areas of the Eastern Shore, Virginia, United States Government printing office.

Srivastava, P., D. Edwards, et al. (1996). "Performance of vegetative filter strips with varying pollutant source and filter strip lengths." Transactions of the ASAE-American Society of Agricultural Engineers **39**(6): 2231-2240.

Stace, H. C. T., G. D. Hubble, et al. (1968). A handbook of Australian soils. Glenside, SA, Rellim.

Stearman, G. K., D. B. George, et al. (2003). "Pesticide removal from container nursery runoff in constructed wetland cells." Journal of Environmental Quality **32**(4): 1548.

Sullivan, A. Atrazine management: Guidelines to achieve effective weed control and minimise off farm movement, Grains Research and Developmental Corporation and Queensland Government.

Syversen, N. and M. Bechmann (2004). "Vegetative buffer zones as pesticide filters for simulated surface runoff." Ecological Engineering **22**(3): 175-184.

Tingle, C. H., D. R. Shaw, et al. (1998). "Metolachlor and metribuzin losses in runoff as affected by width of vegetative filter strips." Weed Science: 475-479.

USDA (2000). Conservation buffers to reduce pesticide losses, USDA.

Walbridge, M. R. and J. P. Struthers (1993). "Phosphorus retention in non-tidal palustrine forested wetlands of the mid-atlantic region." Wetlands **13**(2): 84-94.

Weaver, M., R. Zablotowicz, et al. (2004). "Laboratory assessment of atrazine and fluometuron degradation in soils from a constructed wetland." Chemosphere **57**(8): 853-862.

Wilson, L. G. (1967). "Sediment removal from flood water by grass filtration." Transactions of the American Society of Agricultural Engineers **10**(1): 35-37.

Winchell, M. and T. Estes (2009). "A review of simulation models for evaluating the effectiveness of buffers in reducing pesticide exposure." **1**: 1-51.

Wong, T. H. F., P. F. Breen, et al. (1999). Ponds vs wetlands - Performance considerations in stormwater quality management. Conference on Comprehensive Stormwater and Aquatic Ecosystem Management, Auckland, New Zealand.

Yuan, Y., R. L. Bingner, et al. (2009). "A review of effectiveness of vegetative buffers on sediment trapping in agricultural areas." Ecohydrology **2**(3): 321-336.