Effectiveness of vegetated systems in managing contaminated runoff from sugarcane and banana farms to protect off-farm aquatic ecosystems, particularly the Great Barrier Reef

TropWATER Report No. 14/10

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EXECUTIVE SUMMARY

Vegetated systems (e.g. grassed strips, riparian vegetation, wetlands, sumps) are increasingly being incorporated into farming systems in north Queensland, especially in the catchments draining to the Great Barrier Reef (GBR) lagoon, to improve downstream water quality.

The objective of this review was to investigate the role and effectiveness of vegetated systems in trapping nutrients, pesticides and sediment in GBR catchments and hence preventing loading of downstream environments particularly the GBR. The following questions from DEHPs Reef Water Quality (RWQ) Research and Development program were addressed:

- What are the most effective methods for trapping loss of reef pollutants from sugarcane farms?

- What is the effectiveness of water quality filters like floodplains, riparian areas, grassed buffer strips and wetlands in reducing nutrients, sediments and pesticides?

The review investigated the effectiveness of a variety of vegetated systems at sites within the South Johnstone, Tully, Herbert and Burdekin catchments. As a null hypothesis, we postulate that the residence time of contaminants in vegetated systems, especially for dissolved and fine particulate material, is the most important factor in determining trapping effectiveness. As particulate material is generally easier to trap than dissolved matter, properties of contaminants which predispose them to be present in a particulate form or to adsorb onto particulate matter will strongly regulate trapping effectiveness. Thus large hydraulic volume traps or systems with relatively low input volumes will be the most effective at trapping agricultural pollutants.

The review included an evaluation of the likely performance of the different systems in different parts of GBR catchments (freshwater and estuarine) and between catchments and in different rainfall and hydrological conditions. This included some modelling of residence times as the main explanatory factor in the ability of systems to trap different materials. The systems reviewed include:

a. Grassed drains, buffer strips, headlands, inter-rows, etc.

b. Riparian vegetation

c. Natural wetlands (freshwater and estuarine)

d. Constructed wetlands

e. Reclamation sumps

f. Floodplains

The project had three main components:

a. An extensive literature search of relevant studies from Australia and overseas. This component included an analysis of where the review studies were applicable in the north Queensland context.

b. Field studies of the effectiveness of various sorts of constructed wetlands in trapping pollutants under different flow conditions.

c. Modelling water residence times in overbank flow conditions on the Tully-Murray flood plain and making preliminary conclusions as to the degree of likely trapping/removal of pollutants in such conditions.

A principal finding of the study is that the residence time of water in trapping mediums is an important measure of likely effectiveness of any vegetated area. Long residence times lead to effective trapping while short residence times are unlikely to trap anything. The trapping efficiency is also critically determined by the nature of material (correlated with residence times) – in general the order of potential trapping is:

- Coarse particulate material (sediment – sand and gravel) – high efficiency.
- Medium particulate material (sediment - silt and adsorbed/absorbed contaminants) – moderate efficiency.
- Fine particulate material (sediment – fine silts and clay and the adsorbed/absorbed contaminants) – low efficiency.
- Dissolved material (e.g. nitrate, atrazine) – very low efficiency.

As a result of this relationship only at floodplain scales are residence times long enough to achieve some trapping of dissolved and fine particulate material in the wet season. Trapping in smaller vegetated systems is only effective in the dry season or in low flow conditions. The low flow conditions of irrigation tailwater flows is a special case in the lower Burdekin where higher levels of trapping of fine particulate and dissolved material can occur. These findings give strength to our null hypothesis and show that given sufficient information about wetlands in the GBR catchment we can reasonably well predict likely trapping effectiveness for particular contaminants.

Permanent trapping of contaminants is also dependent on the trapped material not being removed by flushing on the next or subsequent high flows. Sump systems which recycle the trapped material back on to the paddock can achieve high trapping effectiveness. Essential to this working are well designed high flow bypass systems so the trapped material is not flushed downstream.

Long residence times of materials like atrazine and nitrate in the trap are necessary to allow processes like denitrification (and hence removal of nitrogen as N_2) and pesticide chemical degradation to benign chemical forms to occur. Degradation half-lives of pesticides commonly used in the sugar industry are in the order of 50 – hundreds of days. Hence to degrade significant amounts of these chemicals they must be held in the trap for long periods.

While it is clear that only constructed wetlands/sumps/vegetated areas with long residence times are capable of significant levels of trapping of all pollutants (except coarse sediment) further research is needed to better be able to accurately quantify the potential degree of trapping in the varying circumstances across the GBR catchment. Experimental work in the current project only focussed in the Wet Tropics and lower Burdekin areas. While the lessons learnt here and from the literature survey may be applicable in other parts of the GBR catchment, some of the conclusions would need to be validated in the actual region e.g. the Fitzroy catchment. While we can make reasonable predictions of effectiveness, more research is needed to be able to predict accurately the effectiveness of particular designs in a Queensland context. In particular the role of vegetation on river floodplains in slowing up flow in overbank flow events needs to be quantified in order to be able to be able to predict residence times and likely degree of trapping through sedimentation of fine sediments, denitrification and pesticide degradation. This may be very important in assessing the effects stemming from changes to the Vegetation Management Act (1999) where increased clearing of riparian and frontage country vegetation may eventuate.

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BACKGROUND

1.1 Brief overview of water quality issues in the GBR

The Centre for Tropical Water and Aquatic Ecosystem Research (TropWATER), of James Cook University, has been commissioned by the Department of Environment and Heritage Protection (DEHP) to undertake this literature review.

Overall, reefs of the Great Barrier Reef (GBR) have declined in condition greatly since the 1960s [36, 75] due to stress from fishing, water quality impacts and climate change [75, 90, 240, 325]. Similarly seagrass meadows, dugong populations, sharks, inshore dolphins and many other important components of the GBR are also in decline from identical causes. One of the most important factors leading to this decline is terrestrial pollutant runoff from the GBR catchment [38, 40].

Deterioration of water quality reaching the GBR lagoon and subsequent degradation of marine habitats continues to be attributed to land use modifications and land management practices in GBR catchments (Figure 1). Since European settlement water quality of the GBR lagoon has been declining [40, 157]. It is now estimated that total suspended sediment load (TSS) to the GBR has increased by 5.5 times (14,000 kilotonnes per year (kt/yr)), total nitrogen load by 5.7 times (66,000 tonnes per year (t/yr)), and total phosphorus load by 8.9 times (14,000 t/yr) [157]. These pollutants are delivered to the GBR in high flow river discharges leading to large plumes of polluted water intruding large distances into the GBR lagoon (Figure 2). Nutrients (especially inorganic forms primarily derived from fertilisers), TSS, and pesticides, derived from current agricultural practices are the pollutants of highest concern to the GBR due to their impact on the planktonic and benthic communities [315]. Dissolved inorganic nitrogen (DIN) is mainly sourced from fertiliser application in sugarcane and the greatest proportion of PSII pesticides (photo-system II inhibitors, e.g., atrazine, diuron) are also sourced from weed and pest management practices in sugarcane, especially in the Wet Tropics [315].

Land management practices in the GBR catchment are continually being refined in all agricultural industries (grazing, sugarcane, horticulture) to increase productivity and profitability of agricultural enterprises as well as reduce their environmental impact and this has also been facilitated by targeted funding programs such as Reef Rescue. One land management practices which is increasingly being incorporated into farming systems is vegetated systems, including grassed drains, constructed wetlands and water reclamation pits (sumps). Such vegetated systems are introduced for a variety of reasons including erosion control, pest control, water use efficiency, improved downstream water quality and to increase on-farm biodiversity [37, 175, 199, 318].

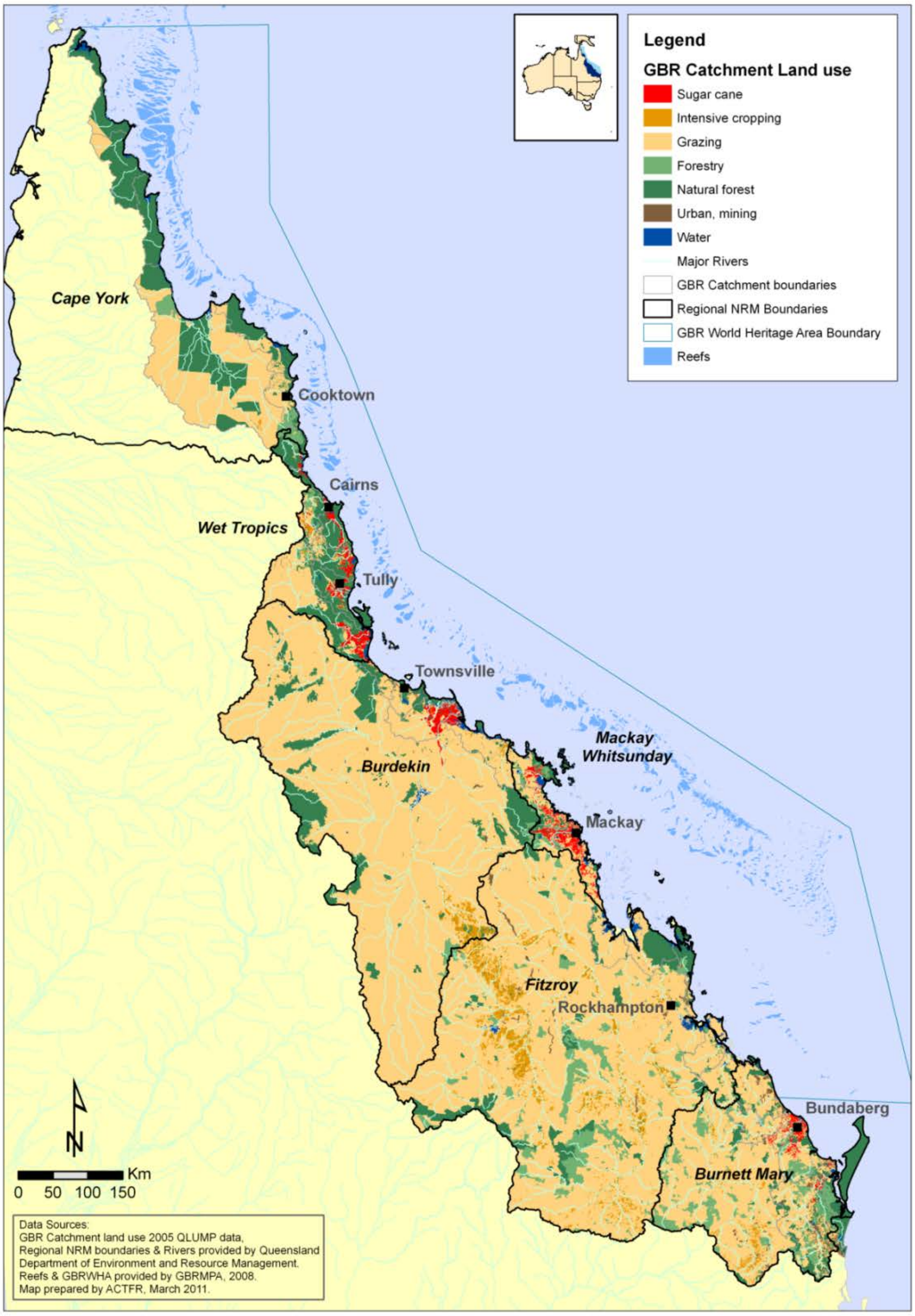


Figure 1: Great Barrier Reef catchment area with land uses.

Intensive agriculture in the GBR catchments is primarily located on floodplains and in hilly coastal areas. Consequently vegetated systems are a natural feature of most intensive agricultural enterprises in the GBR catchment however their origin, extent, type, condition, functionality (agricultural, hydrological and environmental) are highly diverse and include:

- riparian vegetation along rivers and creeks [64, 199, 200];
- on-farm drains which may follow natural drainage lines or be a part of new farm design, and may be vegetated, grassed or clear of vegetation;
- effective vegetated treatment areas (EVTAs, grassed buffer strips);
- natural wetlands [194, 195];

- modified wetlands;
- constructed wetlands; and
- water reclamation pits (sumps).

Constructed wetlands and water reclamation pits are increasingly being incorporated into farming systems for a variety of reasons, including irrigation (particularly in the Burdekin River basin), as a 'sacrifice' area to use sediment in low-lying areas to improve productivity in other areas, to increase on-farm biodiversity and to improve downstream water quality [37].

There is a scarcity of experimental information available on the ability of vegetated systems to trap pollutants in the hydrological and climatic conditions of the GBR catchment. Vegetated systems may prevent pollutants from reaching the GBR and other ecologically sensitive areas, such as mangroves, intertidal and other coastal areas, by retention in the water and sediment, biological utilisation, and/or breakdown. Pollutants may however be remobilised and transported further downstream or across the floodplain, due to overbank flow, in high flow events. Movement of some pollutants (e.g., nitrate, atrazine) through the soil into shallow groundwater systems and streams is also a significant factor in the effectiveness of vegetated systems. Improved understanding of the effectiveness of different vegetated systems in reducing pollutant loads to downstream habitats will better inform prioritisation of investments to reduce pollutants as well as the future design of constructed wetlands and reclamation pits to improve their functionality.

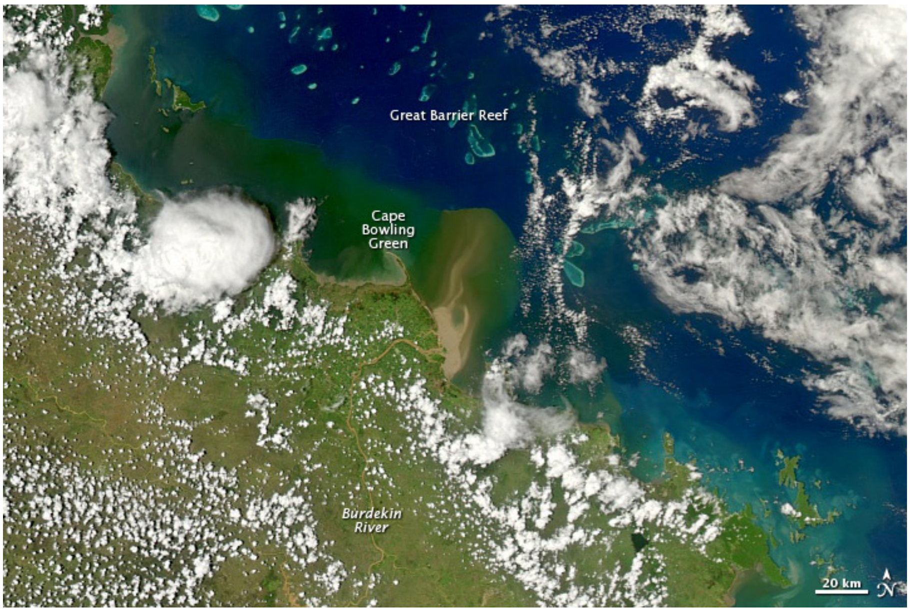


Figure 2: A river plume extending from the Burdekin River into the GBR on 4th January 2011.

Source:[227].

1.2 Management options to reduce pollution

This review includes an analysis of the effectiveness of different vegetated systems at reducing nutrient, pesticide and sediment losses to the GBR. The following questions from DEHPs Reef Water Quality (RWQ) Research and Development program were addressed:

- What are the most effective methods for trapping loss of reef pollutants from sugarcane farms?
- What is the effectiveness of water quality filters like floodplains, riparian areas, grassed buffer strips and wetlands in reducing nutrients, sediments and pesticides?

This review will investigate the effectiveness of a variety of vegetated systems across three regulated catchments (see below) within the South Johnstone, Herbert and Burdekin catchments.

The Queensland Government's Reef Protection Package was introduced in 2010, and included regulatory requirements for the use of fertilisers and pesticides for growers in the Wet Tropics, Burdekin Dry Tropics and Mackay-Whitsunday catchments (regulated GBR catchments), and implementation of Environmental Risk Management Plans (ERMPs) for cane growers with more than 70ha in the Wet Tropics. To achieve the Reef Water Quality Protection Plan (Reef Plan) [260] objectives of reducing nutrients by 50% and pesticides by 60% reaching the reef by 2008, it is important to consider the best options for minimising nutrient and pesticide losses from sugarcane farms. The Reef Plan encourage pre-application management strategies and precision application of pesticides and nutrients as these practices are likely to be more effective in reducing the amount of pollutants lost in farm run-off than from post-application strategies such as vegetated treatment areas. However it is still important that the potential for end-of-paddock treatment is optimised [100]. Currently DEHP is funding CANEGROWERS for the development and delivery of Sugarcane Best Management Practice (BMP), by June 2014, in partnership with the Queensland Government and on behalf of the Australian sugarcane industry. Once the BMP program takes effect, it is proposed that there will be a transition from a regulatory approach to an industry driven, voluntary approach to ensuring the risk of pollutant losses to the reef is managed[50].

The Reef Plan [260] requires growers to take reasonable and practical measures to maximise efficiency of pesticide application through optimising rates, targeting application and avoiding application during high risk periods [17]. However, where photosystem II-active (PSII) pesticide use is necessary, some losses are inevitable. It is therefore important to understand the conditions under which on-farm, end-of-paddock vegetated systems can effectively treat pesticides and nutrients, and optimise their water treatment efficiencies. Current understanding of the various options for treating and trapping pollutants is limited, as are the impacts of environmental conditions on the effectiveness of these options. A literature review by Brodie *et al.* [37] was commissioned by DEHP to assess the likely effectiveness of EVTAs and community drainage schemes in the Babinda area of the Wet Tropics in trapping and treating PSII pesticide pollution. The literature review identified that, under low rainfall conditions, vegetated buffer strips can trap more than 50 percent of pesticides in run-off, but that the effectiveness of such systems may be seriously compromised under the high rainfall and rapid discharge conditions found in parts of the Wet Tropics. Under these adverse conditions it appears likely that neither naturally vegetated areas nor broad grass strips will increase residence times sufficiently to reduce pollutant loads. The review identified that trapping effectiveness is influenced by a number of factors including the vegetation type, height and density, soil conditions, rainfall characteristics and the solubility of the pesticides. It identified residence time as the key factor affecting efficacy of treatment.

The Queensland Government reef protection regulations prohibit application of fertilisers and pesticides under adverse conditions. It is therefore important to know the effectiveness of end-of-paddock treatment and trapping options where applications are made under optimum conditions, e.g. any soil conditions and with no significant rain forecast within 48 hours. Current regulations require EVTAs to be 5 metres wide, however Brodie *et al.* [37] indicated that grassed barriers of 5 metres may have limited efficacy. None of the published literature assessed the configuration of vegetation treatment areas found in actual cane fields and the low flow path of water across vegetated treatment areas. It is therefore important for future studies to characterise the topographic and flow characteristics/paths of typical EVTAs in the regulated catchments to determine residence time of pollutants when PSII pesticides and fertilisers are applied under optimal conditions. This includes consideration of water table, soil characteristics, climatic conditions, landscape features, drainage patterns and EVTA features such as vegetation type and cover, and the presence and nature of berms.

1.3 Objectives

The objective of this report is to review the role of vegetated systems in trapping nutrients, pesticides and sediment in GBR catchments. It will include an evaluation of the likely performance of the different systems in different parts of GBR catchments (freshwater and estuarine) and between catchments and in different rainfall and hydrological conditions. This will include some modelling of residence times as the main explanatory factor in the ability of systems to trap different materials. The systems to be reviewed include:

a. Grassed drains, buffer strips, headlands, inter-rows, etc.
b. Riparian vegetation
c. Natural wetlands (freshwater and estuarine)
d. Constructed wetlands
e. Reclamation sumps
f. Floodplains

As part of this review a theoretical framework will be developed, applicable to the regulated GBR catchments, to assess the pollutant (nutrient, pesticide and sediment as a vector for nutrient and pesticide) treatment and trapping effectiveness of end-of-paddock management options for sugarcane and banana farms in these catchments. This literature review highlights those aspects of vegetated systems that are likely to contribute to trapping effectiveness, which can be translated into practical management advice that can be provided to landholders and Natural Resource Management (NRM) managers concerned with the GBR catchment. Identification of appropriate physical characteristics for end-of-paddock pollutant trapping systems in each catchment will provide water quality and economic benefits. The provision of location appropriate advice will assist land managers to allocate resources (i.e., time, money and labour) to implement and/or manage those treatment systems that will provide the best water quality outcomes.

1.4 Definitions for terms used in this report

<u>Berm</u>: A strip of land or elevated bank bordering a river or canal.

<u>Biodegradation</u>: The breakdown or dissolution of materials or chemicals by microorganisms or other biological means.

<u>Colloid-bound contaminants</u>: Contaminants which are adsorbed onto colloid particles (sized between 1,000 Daltons (atomic weight units) and 0.2 µm, that may be inorganic (e.g., clay minerals) or organic (e.g., humic and fulvic substances).

Constructed wetland: A wetland designed and constructed to treat wastewater. These can include inlet and outlet pipes, bypass flows, sediment traps, varying water depths (deep pools, shallow vegetated zones), vegetation, and other possible mechanisms for treating wastewater.

Dissolved substances: Substances (e.g., nutrients and pesticides) that have been incorporated into solution as compared to particulate matter.

Effective Vegetated Treatment Area (EVTA): Also known as grass buffer strips, EVTAs are grassed areas with widths of either 3 metres, 5 metres, 10 metres or 20 metres, 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 80 percent grass cover, between 10 - 15 centimetres high[102].

End-of-paddock: At the downslope edge of one or more paddocks, catching runoff coming directly from that paddock.

Infiltration: The process of surface water entering the soil.

Irrigation tail-water: That part of the applied irrigation which flows off the end of the irrigated field.

Off-farm: Practices or systems that are based off the farm.

On-farm: Practices or systems which are based on the farm itself.

Low flow runoff 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).

Particulate matter: Small, distinct particles which can be suspended in a liquid, or settled onto a surface. Nutrients and pesticides can adsorb onto such particles and are then measured as particulate fractions, such as NP (particulate nitrogen), PP (particulate phosphorus), etc.

PSII herbicides/pesticides: Class of herbicides which disrupt photosynthesis by blocking electron transfer in Photosystem II (PSII), e.g., atrazine, ametryn, diuron, etc.

Reclamation pit: Also known as a sump, is an excavated pit, usually on-farm, and end-of-paddock, that is designed to collect irrigation tail-water for storage and/or reuse.

Sub-surface drains: Mole drains and agricultural pipes.

INTRODUCTION

Minimal experimental information is available on the ability of vegetated systems to trap pollutants (i.e., sediment, nutrients or pesticides) in the hydrological and climatic conditions of the GBR catchment. Only limited research has occurred into the efficiency of systems such as riparian vegetation [64, 199, 200] and wetlands [194, 195] in the GBR catchment. A desktop study has recently been completed on the potential effectiveness of grass buffer filter strips (EVTAs) and vegetated drains in trapping herbicides in Babinda [37]. This study found that minimal trapping of dissolved phase herbicides could occur in the Babinda drainage scheme, due to low infiltration rates and short residence times. The ability of EVTAs and other vegetated systems to trap pollutants is directly impacted by several factors, including rainfall, groundwater and aquifer characteristics, solubility of pesticides, slope, and the type and condition of buffer vegetation.

1 Whether or not pollutants that have originated on sugarcane and banana farms reach the GBR, is affected by on-farm and off-farm environmental features. Pollutants may be prevented from reaching the GBR or other ecologically sensitive areas by filtering through EVTAs, sediment traps, wetlands or off-farm vegetated areas, or be distributed, settled and/or re-mobilised through overbank flow across floodplains. Here, the following vegetated systems are reviewed:
   a. EVTAs (buffer strips, including grassed drains, inter-rows and headlands);
   b. riparian areas (including grass and/or trees);
   c. natural wetlands (freshwater and estuarine);
   d. constructed wetlands;
   e. reclamation sumps (pits); and
   f. floodplains.

2.1 Agriculture in Queensland

Sugarcane (Figure 3) is grown along the Queensland coast from the Gold Coast to Mossman and specifically in the GBR catchment area from Maryborough to Mossman (Figure 1). Table 1 shows the sugarcane production figures in Queensland, for 2010 and 2011.

Figure 3: Sugarcane paddocks in the Herbert River district, North Queensland.

Photo credit: C. Coppo

Table 1: Queensland sugarcane production figures for 2010 and 2011.

	Tonnes of cane		Tonnes of sugar IPS		CCS		Hectares	
Mill Area	**2010**	**2011**	**2010**	**2011**	**2010**	**2011**	**2010**	**2011**
Mossman	539,569	411,012	61,300	57,269	11.65	13.81	7,150	7,320
Tableland	651,922	644,879	90,225	-	13.59	14.5	6,925	7,066
Mulgrave	1,116,341	733,790	137,015	104,629	11.79	13.78	12,621	12,150
Babinda	630,072	NA	72,273	NA	10.33	NA	7,094	NA
Innisfail	1,209,040	671,368	211,332	155,522	10.94	11.33	13,642	17,450
Tully	1,823,079	1,158,078	-	135,947	11.14	11.98	21,000	24,610
Herbert River	3,274,402	2,920,401	419,090	362,347	12.85	12.89	39,568	52,365
North QLD	10,695,749	6,539,528	854,220	815,714	11.95	12.89	74,379	120,961
Burdekin	6,460,730	9,547,612	901,698	1,308,890	13.7	13.6	49,830	79,669
Proserpine	1,165,086	1,467,079	152,098	191,958	13.07	13.32	15,225	21,206
Mackay	4,555,765	4,162,358	611,017	563,718	13.04	13.25	62,170	69,070
Plane Creek	814,950	1,067,474	111,740	138,338	14.04	13.46	12,274	15,390
Central QLD	6,535,801	6,696,911	874,855	894,014	13.17	13.3	89,669	105,665
Bundaberg	1,544,962	1,394,984	211,249	192,185	13.66	13.83	18,887	18,884
Isis	1,154,751	1,223,135	160,984	171,762	13.31	13.4	13,738	14,686
Maryborough	575,159	669,605	74,929	88,416	13.08	13.04	8,471	9,621
Rocky Point	245,757	267,669	31,629	34,877	13.11	13.03	3,211	3,586
South QLD	3,520,629	3,555,393	478,791	487,240	13.41	13.47	44,306	46,777
Total QLD	22,822,166	26,339,444	3,109,564	3,505,858	12.85	13.33	258,184	353,072

Notes: For NSW tonnes of sugar in 94NT and sugar content is POL not CCS.
CCS = Commercial Cane Sugar, a measure of recoverable sugar in the cane. Numbers represent mean percentage per mill area and region.
IPS - International Pol Scale. A price adjustment scale described in the rules of the Sugar Association of London. It defines incremental price premiums and penalties applied to sugar above 96 degrees polarisation.
POL - A measure of the sucrose content of sugar.

Source: [51, 52].

Within the GBR catchment, sugarcane is grown under various levels of irrigation depending on local weather conditions. Table 2 shows the area of sugarcane under irrigation in each mill area in 1999.

North Queensland is also the leading Australian banana producer, with more than 90 percent of Australia's bananas grown in the Cardwell, Babinda, Tully and Johnstone regions. Approximately 11,000 hectares of bananas are grown in the Wet Tropics north of the Herbert River catchment [192]. Between 2007 and 2010, Queensland banana production increased from 187,636 tonnes to 279,805 tonnes. Phosphorus and nitrogen use are both higher in bananas than in sugarcane but application rates of N have decreased greatly in bananas and a little in sugarcane since the mid-1990's due to improved management practices [6].

Table 2: Area of sugarcane land use in Queensland in 1999.

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	68,004	15	10,201
Burdekin	84,004	100	84,004
Proserpine	24,716	89	22,000
Mackay	98,324	70	68,827
Sarina	22,398	36	8,063
Bundaberg	53,003	100	53,003
Isis	19,102	88	16,810
Maryborough	15,493	47	7,282
Moreton	9,828	0	0
Rocky Point	6,043	2.3	139
TOTALS	520,563	43.3	284,792

Source: [32], Dwyer, Incitec Pivot, unpublished.

2.2 The GBR and pollutant concerns

Many tropical marine ecosystems around the world are at risk from the effects of land runoff and terrestrially derived pollution, over-harvesting of marine species, increasing temperatures and ocean acidification [48] and many show signs of degradation [4, 36, 54, 75, 90, 137, 235, 240]. The GBR lagoon, situated on the north-east coast of Australia, has the status of a Marine Park under joint Australian (Federal) and Queensland State Government arrangements and has been a declared World Heritage Area since 1981 [40]. Despite this protected status and the management effort applied to manage its natural resources, anthropogenic stresses over the last hundred years have changed GBR water quality and threatened the health of this iconic ecosystems [36, 40, 136].

The GBR receives freshwater inputs from thirty-four GBR catchments situated along the adjacent coast that vary in size, land use, water quality, biophysical and socio-economic characteristics, and management regimes [34, 39, 97, 157, 315]. The priority pollutants for water quality management in the GBR are suspended sediment (SS), nutrients (nitrogen and phosphorus), and pesticides, particularly the Photosystem II (PS-II) inhibiting herbicides. Discharge of these pollutants into the GBR lagoon has increased greatly over the last 200 years, mainly due to wide scale agricultural, urban and mining development [97, 315].

2.2.1 Sediments

Recent estimates indicate that since European settlement in the GBR Catchment Area (GBRCA), the mean SS load has increased by 5.5 times to 17,000 kt/yr, and the total nitrogen and total phosphorus loads around 6 and 7 times, respectively [157]. Runoff and sediment loss are mainly related to the condition and spatial patterns of ground cover upstream [e.g. 15, 278] and a dominant source of SS is hillslope erosion in areas with low pasture cover [202]. Flood events with excess sediment and nutrient loads have caused local declines of GBR seagrasses [196, 279, 316] and this loss of seagrass habitat has been linked to increased mortality of dugongs and sea turtles [196, 197].

2.2.2 Nutrients

Bioavailable nutrient discharge to the GBR causes a variety of damage to the ecosystems of the GBRWHA [39]. These include in particular:

1. Promotion of crown of thorns starfish (COTS) outbreaks via providing COTS larvae with a high quality food source [33, 92]
2. Enhanced growth of macroalgae at the expense of corals [74].
3. Enhanced bleaching response in corals [329].
4. Increased incidence of coral diseases [112].
5. Increased bioerosion of coral reef structures [91].

2.2.3 Pesticides

Prior to the original work by Haynes *et al.* [122] on the effects of diuron exposure to seagrass species, there was virtually no information on the impact of this herbicide on 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 [122, 262], mangroves [19], corals [142, 228] and algae [185, 283]. All of these studies use the pulse amplitude modulation (PAM) chlorophyll fluorescence 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 [168]. However, the laboratory experiments showed that the ability of plant species to fully recover, once removed from herbicide exposure, was species specific and 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. Harrington *et al*. [117] showed that diuron attached to sediment particles can produce an enhanced effect on the sedimentation stress on crustose coralline algae [117]. Another study found longer term impacts on corals that have been exposed to diuron, such as reduced reproductive output [53].

2.3 Management practices addressing on-farm inputs

A variety of improved management practices are continually being implemented and refined by the sugarcane industry to address concerns of potential losses of nutrients and herbicides to the GBR [50]. For nutrient application these improved practices include timing of fertiliser application to coincide with prime growing periods, precision application of fertiliser to the plant stool, different forms of fertiliser (control release granules, liquid fertiliser) and precision application of nutrients according to site specific characteristics. For pesticide application these improved practices include variable chemical application rates, targeted spraying of different chemicals using shielded, hooded sprayers linked with GIS data, 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, GPS guided control traffic and planting operations, and the use of legume fallow crops, weed seed germination is greatly reduced when compared to conventional tillage-based systems (e.g., [88, 98]). This means there is less of a need to rely on residual herbicides due to decreased weed pressure.

Figure 4: Shielded sprayer on a sugarcane farm in Queensland.

Photo credit: A. Davis.

One of the methods used to control weeds using less residual herbicide than traditional boom sprayers is a shielded sprayer (Figure 4). Shielded sprayers utilise a shroud to cover spray nozzles in the furrow so 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 percent compared to boom sprayers [189]. The use of such tools allows farmers to achieve both cost savings and environmental improvements while maintaining appropriate levels of weed control. Recent funding has assisted studies into the effectiveness of new agricultural technologies into reducing on-farm inputs. These technologies include: paddock design, lasering of paddocks and drains, spoon drains, GPS guided control traffic, split stool application of fertiliser, and hooded sprayers for application of herbicide.

Improved paddock design, involving lasering of paddocks and drains, has improved on-farm drainage and reduced the detrimental effects of waterlogging of sugarcane. However, it has significantly reduced the retention time of water, at a catchment scale, in GBR catchments. Flow velocities and volume of runoff have increased, resulting in increased erosion in areas where erosion was previously uncommon, reduced retention time in downstream wetland areas and increased likelihood of pollutants being transported further from the source area.

2.4 Management practices concerning downstream water movement

Post application management strategies, as promoted by the Reef Protection Package, include EVTAs (regulatory requirement), on-farm water retention pits, constructed wetlands (implemented as part of an ERMP) and off-farm riparian areas, constructed wetlands and floodplain modification. These strategies increase retention time of water within catchment and therefore provide the opportunity to reduce the amount of pollutants in runoff depending on the characteristics of such areas.

PROPERTIES OF DIFFERENT POLLUTANTS

3.1 Total suspended solids

Total suspended solids (TSS) range from ‘coarse’ solids, such as sand, to ‘fine’ solids, such as clay particles. Suspended solids settle out of suspension once the water they are transported in slows down; coarse materials settle first, fine solids last. Some very fine solids never settle out and water containing such particles remains cloudy. Specific management techniques for capturing particulates include the use of settlement ponds.

Increased sedimentation can have major impacts on aquatic environments, from smothering seagrasses, benthic algae and invertebrates, decreasing light penetration (see review by [90]), and impacting fish behaviour [321, 322], dependent on the concentration and duration of exposure [231]. One method to gauge potential impacts of sedimentation in aquatic systems is to quantify total suspended solids.

3.2 Nutrients

In tropical Queensland, where agricultural and industrial land uses are predominately situated along the east coast and the addition of nutrients to paddocks is usually required to increase the yield of agricultural crops, water making its way to the reef can be rich in nutrients. The two most studied elemental nutrients in the GBR, nitrogen and phosphorus can be applied through overhead spraying, furrow irrigation, granular application and time-release pellets. Once applied, these nutrients can be highly soluble as well as particle adsorbing and so can be found in both dissolved and particulate forms. Nutrients in the particulate phase can undergo settlement, volatilisation or infiltration, or get caught up in mass flood events as suspended sediment. Nutrients in the dissolved phase can move off the application site in run-off (e.g., irrigation tail-water or rain events) and through seepage into groundwater and subsurface flow.

Nutrients often tested for in agricultural run-off include: dissolved organic/inorganic nitrogen, oxidized nitrogen species or NOx (nitrate, nitrite, ammonia), total filterable and reactive nitrogen and phosphorus, dissolved organic phosphorus, and particulate nitrogen and phosphorus. The varied forms of nutrients behave differently post-application and may require different methods for mitigation and uptake by vegetated features. For example, nitrate is water-soluble and not readily adsorbed by soil particles. Usually nitrate is not in runoff because it 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 fertiliser. To trap nitrate effectively, roots of conservation buffer plants need to intercept this subsurface flow. The conditions for denitrification which are present in this biologically active zone, can also reduce nitrate reaching streams.

Degradation time frames for nutrients are dependent on biochemical processes such as phytoaccumulation, or uptake by plants [220], and degradation or transformation (e.g., denitrification). Denitrification depends on heterotrophic, mostly facultatively anaerobic bacteria as they utilise nitrite and nitrate to help break down organic matter, releasing nitrogen gas from the system. This somewhat slow process (0-345 µmol N m^{-2} h^{-1}) requires an environment with low oxygen and long residence times, and is predominant in sediments as opposed to the water column [284].

3.3 Pesticides

A wide range of pesticides are used in agriculture and each has its own specific metabolic pathway, degradation by-products and half-lives. The ability of different soil types to trap different pesticides is very variable (Table 3). Half-lives, which can range between days and months, often vary between soil and water (Table 4). Since many pesticides take months for degradation to occur, it is important to consider how to trap agricultural tail-water long enough for these degradation processes to occur, prior to these pesticides and some of their by-products, finding their way into natural aquatic systems.

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 [180, 255, 259, 267, 296]. 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 (Figure 5). 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 (see Table 3 for K_{oc} values). The relationship between K_{oc} and the percent pesticide trapped is shown in Figure 6 . While pesticides are also bound to clay particles, binding to organic matter is a useful predictor of pesticide behaviour 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. 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.

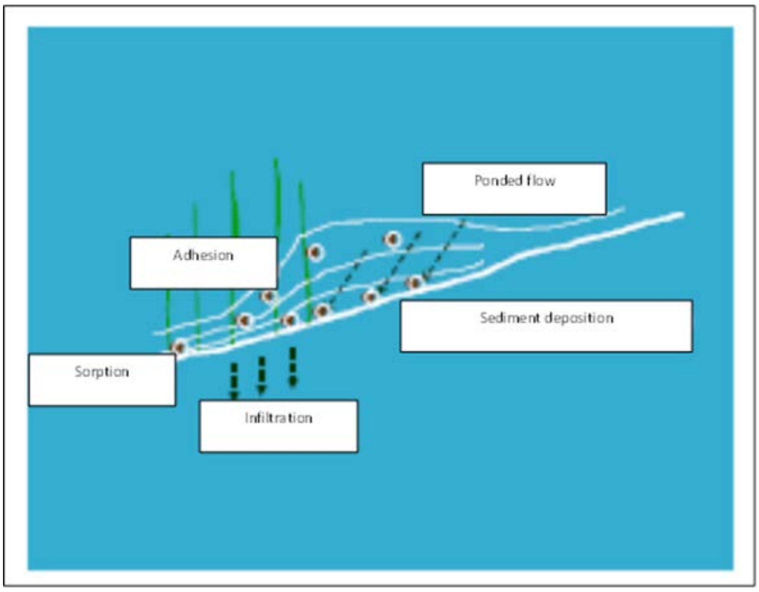


Figure 5. The primary pathways of loss of pesticides from agricultural land.

Source: USDA [309]

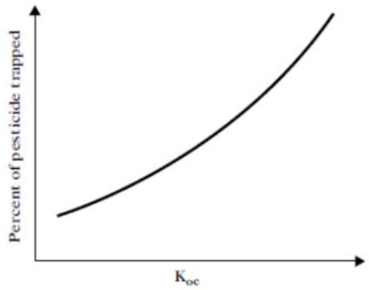


Figure 6: Relationship between percent of pesticide trapped and Koc.

Source: USDA [309].

Example K_{oc} values for specific pesticides are shown in Table 3 and Table 4 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 pesticides typically used in sugarcane in the GBR catchments include paraquat, chlorpyrifos, glyphosate, diuron and probably methoxy-ethyl-mercury chloride (MEMC) [35, 187]. 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 will be considered in detail in this project. Pesticides with lower K_{oc} values (generally less than 500) are transported more by water than sediment and examples typically used in sugarcane 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 [298] 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 [8, 267]. For low to moderately sorbed pesticides, runoff must infiltrate while in the filter strip or pesticides can be removed from solution through contact with the soil or vegetation that may adsorb pesticides in the filter strip [8, 83, 108, 246, 273, 309]. Concentrations of pesticides transported by sediment are higher than that transported by water, but because water quantities running off paddocks are so much greater than the eroded soil quantities, water accounts for the majority of chemicals leaving paddocks.

Table 3: Summary of buffer studies measuring trapping efficiencies for specific pesticides.

Pesticide	K_{oc}	Study Reference	Percent pesticide trapped (%)
High adsorbtion			
Chlorpyrifos	6,070[1]	[29]	57-79
Glyphosate	21,699[2]		
Paraquat	1,000,000[2]		
Trifluralin	8,000[1]	[269]	86-96
		[63]	62-99
Moderate adsorbtion			
Diflufenican	1,990[1]	[245]	97
*Diuron	1,067[2]		
Lindane	1,100[1]	[245]	72-100
MEMC (methoxyethylmecuric chloride),			
Low adsorbtion			
Acetochlor	150[1]	[29]	56-67
Alachlor	170[1]	[176]	91
Ametryn	316[2]		
Atrazine	100[1]	[7]	11-100
		[12]	90
		[29]	52-69
		[115]	91
		[132]	30-57
		[176]	97
		[208]	35-60
		[211]	26-50
		[245]	44-100
		[255]	40-85
Cyanazine	190[1]	[7]	80-100
		[211]	30-47
2,4-D	20[2]	[9]	70
		[63]	89-98
Dicamba	2[1]	[63]	90-100
Fluormeturon	100[1]	[263]	60
		[264]	59
Hexazinone	54[2]		
Imazapic	137[2]		
Imidochloprid	225[2]		
Isoproturon	120[1]	[245]	99
Mecoprop	20[1]	[63]	89-95
Metolachlor	200[1]	[7]	16-100
		[211]	32-47
		[255]	44-85
		[319]	55-74
		[306]	67-97
Metribuzin	60[1]	[319]	50-76
		[306]	73-97
Norflurazon	600[1]	[263]	65
		[264]	63-86

Note: Shaded cells indicate pesticides used in sugar cane application in the GBR catchments.
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).

Source: Derived from USDA, (2000).

Table 4: Soil adsorption coefficient and half-lives of common pesticides in soil.

Common Name/Trade Name	Soil Adsorption Coefficient K_{oc} (μg/g)	Half-life T½ (days)
acephate/Orthene	2	3
dicamba/Banvel	2	14
methamidophos/Monitor	5	6
picloram/Tordon	16	90
2,4-D/Weedone	20	10
dimethoate/Cygon, Dimate	20	7
carbofuran/Furadan	22	50
oxamyl/Vydate	25	4
aldicarb/Temik	30	30
bromacil/Hyvar	32	60
hexazinone/Velpar	54	90
terbacil/Sinbar	55	120
ethoprop/Mocap	70	25
methomyl/Lannate	72	30
tebuthiuron/Spike	80	360
atrazine/Aatrex	100	60
acifluorfen/Tackle	113	14
simazine/Princep	130	60
prometon/Pramitol	150	500
alachlor/Lasso	170	15
captan/Orthocide	200	3
EPTC/Eradicane	200	6
metolachlor/Dual	200	90
carbaryl/Sevin	300	10
linuron/Lorox	400	60
diuron/Karmex	480	90
diazinon/Knox-Out, D.Z.N.	1,000	40
phorate/Thimet	1,000	60
chlorothalonil/Bravo, Daconil	1,380	30
malathion/Cythion, Fyfanon	1,800	1
ethalfluralin/Sonalan, Curbit	4,000	60
fenvalerate/Ectrin	5,300	35
fluazifop-p-butyl/Fusilade	5,700	15
chlorpyrifos/Lorsban	6,070	30
trifluralin/Treflan, Tri-4	8,000	60
diclofop-methyl/Hoelon	16,000	37
glyphosate/Roundup	24,000	47
paraquat/Gramoxone	1,000,000	1,000

Note: These numbers should not be taken as absolute values, but as relative comparisons among the different pesticides.
Shaded pesticides are those in common use in the sugarcane industry.

Source: [308]

3.4 Trapping Mechanisms

The main 'trapping' mechanisms of pollutants (suspended sediments, pesticides and nutrients) are:

a) Infiltration into and retention by soil [7, 9, 28, 95, 154, 158-161, 203, 238, 246, 249, 251, 255, 268, 271, 288, 305, 309]

Infiltration is by far the most important mechanism filtering incoming hill-slope surface flows. Popov *et al*. [255] found that infiltration was the only significant factor that reduced herbicide loads in surface waters. However, when subsurface flows are sizeable, seepage and saturation flows can hinder infiltration [200]. In areas of high infiltration no real trapping of dissolved pollutants may be occurring [203]. Highly soluble pesticides or nutrients are lost via infiltration, through which they can also follow a different flow path through the subsoil and groundwater [28, 154, 246, 271, 274, 276, 305, 309]. In fact, dissolved and particulate-bound pollutants can continue to move to streams through subsurface flow, even though they may have initially been retained through infiltration [249].

b) Infiltration and uptake by vegetation

Nutrient uptake by vegetation is an important mechanism for removal of persistent pollutants, such as phosphorus. However, it is only a long-term solution if the vegetation is removed from the system [107]. Whereas organic pollutants can be degraded, inorganic pollutants can be stabilised or sequestered through bioremediation. Examples of bioremediation of pollutants, including nutrients and metals, include phytoremediation, or uptake and transformation by plants. Phytoremediation processes include accumulation (the plant uptakes the pollutant and accumulates it in tissues), volatilization (pollutants are converted to volatile forms within the plants and released as a gas), and transformation or degradation (pollutants are eliminated by enzymes or plant/root associated fungi or bacteria) [220].

c) Sedimentation

Pesticides adsorbed onto particulates and nutrients in particulate form (as well as the fine suspended sediments themselves) are lost via sedimentation [13, 28, 45, 246, 255, 264, 274, 295]. This requires a long holding time, firstly allowing sediments to settle, from suspension in the water to the soil surface [13, 28, 81, 133, 138, 233, 234, 251, 256, 295, 302, 328], and secondly, to allow for pesticide residence time - to ensure that biodegradation, or breakdown by microorganisms, occurs [7, 9, 13, 28, 42, 81, 95, 108, 110, 145, 234, 239, 241, 246, 249, 251, 255, 256, 264, 271, 272, 288, 295, 302, 305, 309].

d) Conversion to non-problematic forms

Another mechanism for removing or reducing pollutants is via biodegradation of pesticides into non-toxic products and conversion of nitrogen into gaseous forms (ammonia, dinitrogen, nitrous oxide) that are lost to the atmosphere [220].

3.5 Predictive models for estimating trapping efficiency

A number of modeling approaches have been developed to examine vegetative trapping [8, 13, 96, 193, 232, 239, 276, 327] and predictive models exist that can be used to determine appropriate buffer widths. For example, Sabbagh *et al*. [276, 277] 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 Environmental Protection Agency (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 [251, 276]:

$$\blacktriangle P = a + b(\blacktriangle Q) + c(\blacktriangle E) + d\ln(F_{ph} + 1) + e(\%C)$$

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.,* [276]. 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.

3.6 Environmental Drivers

There are many environmental factors that affect the ability of vegetated systems to reduce or remove pollutants from streams and thereby prevent nutrients and pesticides from reaching the GBR. These factors include climatic conditions (Figure 7) and hydrology, soil type, residence times, physical characteristics (grade, buffer width, wetland depth, uniformity/diversity of structure), and type of vegetation (form and species) and organic matter within the system. As can be seen in Figure 7 there are major variations in rainfall across the GBR catchment with rainfall in the Wet Tropics ranging from 2000-3200mm but drier Dry Tropics area only receiving 600-1200 mm annually. In addition rainfall can vary greatly between years in the same catchment. This variability greatly affects many of the parameters which govern the effectiveness of vegetated systems to remove pollutants e.g. residence times.

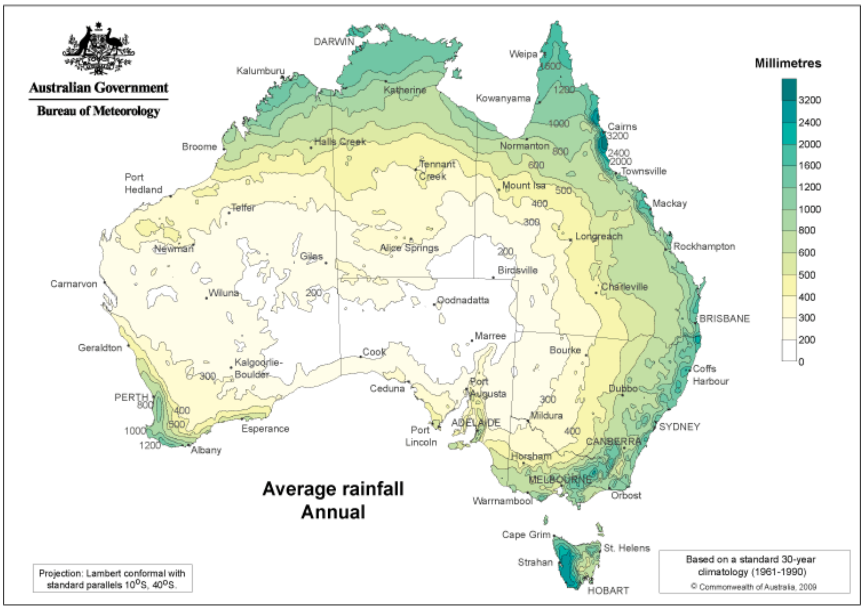


Figure 7: Average annual rainfall in Australia.

Source: [206].

CHARACTERISTICS OF VEGETATED SYSTEMS

4.1 Grassed buffer strips, headlands, and inter-rows

Grassed buffer strips (EVTAs), headlands and inter-rows surround, or are within, paddocks of intensive agricultural areas (Figure 8) and are upstream of wetlands, sumps, streams and other waterways. The efficiency of such grassed vegetative systems, to act as a water filter system, varies depending on their species composition, width, slope and condition such as dried out, mowed, not mown, invaded with weeds or high vehicle use areas. Increased effectiveness has been associated with homogeneous, densely growing plants which prevents the formation of erosion rills, the width of such vegetative systems selected according to particular features of the upstream farming area and medium height (native) grass species which are maintained at a height of at least 10-15 cm [296].

Figure 8: Grassed inter-rows on a banana farm.

Photo credit: J. DeBose

4.2 Riparian Vegetation

Riparian vegetation may be defined as that vegetation (whether herbaceous or woody) adjoining a river or stream. Riparian vegetation has been widely recognised for its capacity to remove agricultural contaminants from groundwater and surface water and protect aquatic ecosystems [11, 82, 124, 135, 177]. The retention or restoration of riparian vegetation has been identified as an effective means of improving water quality caused by contaminated runoff from agricultural areas [11, 82, 124, 130]. While typically occupying only a small fraction of the landscape area, due to their unique, low-lying position in the landscape, located between terrestrial and aquatic environments, riparian vegetation plays a disproportionately important role in controlling and processing contaminant flow to aquatic environments [111, 130, 178, 237].

Some of the key structural components of riparian vegetation that influence stream water chemistry are shown in Figure 9.

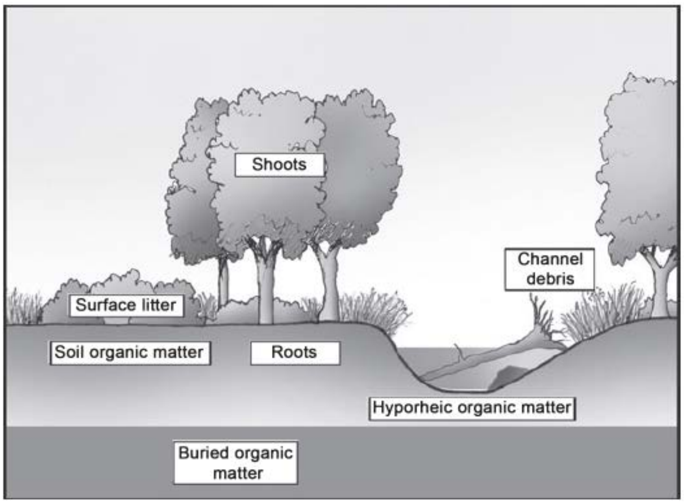


Figure 9: Major components of riparian vegetation buffers that influence stream water chemistry.

Source: *[82].*

4.3 Natural wetlands (freshwater and estuarine)

The Queensland Wetland Program (QWP) has mapped and classified Queensland's wetlands [2] and provided a comprehensive 'Program Wetland Definition' (Figure 10) to facilitate the long term management, conservation and protection of Queensland wetlands [103]. Wetland mapping has traditionally been conducted by examining biotic indicators and surface hydrology however, in environments where these characteristics are dynamic however soil components of the definition need to be considered to provide a more robust wetland identification tool [44, 76].

Figure 10. Queensland Wetland Program - Wetland Definition.

Wetlands are areas of permanent or periodic/intermittent inundation, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed 6m. To be a wetland, the area must have one or more of the following attributes:

1. The land supports, at least periodically, plants or animals that are adapted to and dependent on living in wet conditions for at least part of their life cycle;
2. The substratum is predominantly undrained soils that are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers; and
3. The substratum is not soil and is saturated with water, or covered by water, at some time.

Examples under this definition include:

- those areas shown as a river, stream, creek, swamp, lake, marsh, waterhole, wetland, billabong, pool or spring on the latest 'Sunmap' 1:25,000, 1:50,000, 1:100,000 or 1:250,000 topographic map.
- areas defined as wetlands on local or regional maps prepared with the aim of mapping wetlands
- wetlands regional ecosystems (REs) as defined by the Queensland Herbarium
- areas containing recognised wetland plants
- saturated parts of the riparian zone
- artificial wetlands such as farm dams
- water bodies not connected to rivers or flowing water, such as billabongs and rock pools.

Source [103].

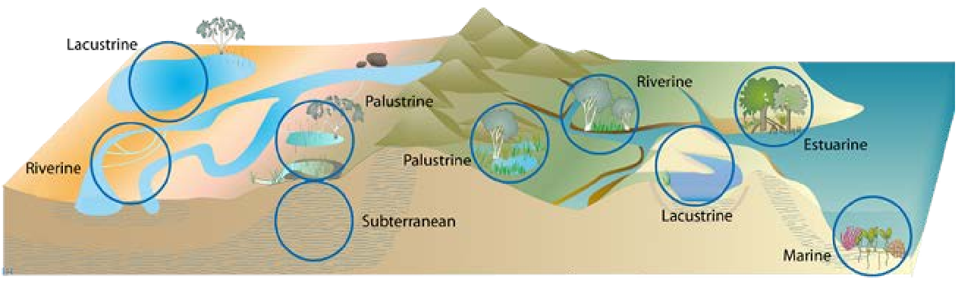


Figure 11. Classification of wetland types in the landscape.

Source: [323].

Queensland wetlands have been classified into six wetland system types (Figure 11); lacustrine, palustrine, riverine, estuarine, marine and subterranean and discrete wetland habitat types on a state scale [323]. In the GBR catchment area the natural wetland systems most likely to be directly affected by agricultural activities are lacustrine (>8ha), palustrine (<8ha) and estuarine systems and these will be considered here. Coastal marine systems and subterranean wetland systems are also likely to be effected by agricultural and grazing activities in adjacent catchments however these systems are not considered within this report.

A number of natural wetlands in the GBR catchment are of national and international significance, with five wetlands listed under the Ramsar Convention and 210 recognised as nationally important wetlands [324]. Natural wetlands are vital for ecosystem function and are the ecotone between all aquatic and terrestrial environments [213] however historically they have usually been regarded, and treated, as waste lands in Great Barrier Reef catchments [123]. Conservation of natural wetlands is likely to maximise nutrient retention within a river catchment [60]

4.4 Constructed wetlands

Figure 12: Constructed wetland on a banana farm in the Johnstone River basin.

Photo credit: J. DeBose.

Numerous studies have been conducted on the effectiveness of constructed wetland (Figure 12) for the mitigation of high nutrient wastewater in temperate areas (e.g., [94, 116, 143, 144, 300]), sub-tropical areas, such as the Florida Everglades (see review by [61]), and in urban catchments [328] however very little work has been conducted in tropical areas which are prone to seasonal flooding and drought (but see [105, 107]). One obvious advantage of constructed wetlands, regardless of how effective they are in mitigating downstream impacts, is that the construction of wetlands on farms brings a sense of ownership to the farmer, as well as a macrocosm to monitor, in terms of directly observing how on-farm practices impact downstream wetland health.

Constructed wetlands for agricultural run-off are usually situated 'on-farm', in areas that had previously yielded low to marginal crop production and were often former wetlands [333]. For water quality improvement services, they are located downstream of tail-water or irrigation discharge areas, or down-land of overland flows of run-off. The ideal size is dependent on the size of the catchment area, or the number of hectares which drain into the wetland and how much water the wetland will generally be treating, while maintaining a steady and moderate inflow. Constructed wetlands also require an impermeable bottom layer, either clay or man-made material, to protect the groundwater from infiltration of pollutants.

A catchment approach to runoff, where upstream wetlands are utilised to increase the effectiveness of downstream wetlands in reducing nutrient loadings [261], might be best to successively treat large loads of polluted water prior to reaching coastal estuaries. Moore *et al*. [222] promotes the use of constructed wetlands, but instead of as the sole run-off mitigation strategy on small farms, they should be combined with other best management practices and vegetative systems (e.g., grassed drains, etc.). Such 'treatment trains' are often utilised for treating runoff, and consist of a series of nutrient and sediment trapping mechanisms, such as grassed/vegetated drains and sediment basins prior to entering the wetland. Treatment trains are often effective, especially when they include different types of flow regimes (e.g., deep pools with slower flow, subsurface flow, turbulent flow through shallow marsh areas, etc.). Constructed wetlands usually have a built-in high flow bypass design, driven by hydraulic/backward flow into an attached sediment basin. Bypass designs redirect large flows around the wetland to avoid flushing of the wetland downstream. Design options for adaptive management, such as water level gauges, and pumps, etc., can also be included in constructed wetlands. Important issues that are considered when designing wetlands, other than upstream catchment area, include grade changes in the land (i.e., direction of water flow) and water table depth.

Many factors influence the ability of constructed wetlands to reduce pollutants and improve water quality exiting the system. These factors include climatic conditions (e.g., amount of rainfall, surface temperature), background levels of organic matter, age, type and distribution of vegetation, nutrients and solids generated within the wetland (such as dissolved organic nitrogen and phosphorus, and detritus) [107], and overall residence time of the water within the wetland. There are also factors associated with the specific quality of the runoff, for example, wetlands with increased nitrate addition usually have increased denitrification rates due to general nitrogen limitation [285].

There are three main types of constructed wetlands: free water surface (FWS) or surface flow, subsurface flow (SF), and vertical flow (VF) wetlands. Here we focus on the characteristics and effectiveness of FWS wetlands, which appear to be the dominant type of wetland maintained on agricultural lands in Queensland.

4.5 Reclamation sumps

Reclamation sumps (pits) are usually constructed wetlands that have the added function of supplying irrigation water back to the surrounding agricultural land (Figure 13). Sumps are found in irrigation areas where the tail water from irrigated land is collected and stored. This water is often high in nutrients and pesticides which is then used to irrigate agricultural land. Reclamation sumps (pits) are excavated to provide an on-farm water resource point; to receive irrigation tailwater and runoff; and be used to irrigate out of.

Figure 13: On-farm reclamation sumps (pits) in the lower Burdekin River catchment.

Photo credits: J. DeBose and D. O'Brien.

4.6 Floodplains

A floodplain is an area of land adjacent to a stream or river that stretches from the banks of its channel to the base of the enclosing valley walls and experiences flooding during periods of high discharge. Floodplains are formed by a complex interaction of fluvial processes but their character and evolution is essentially the product of stream power and sediment character [226]. Topographically it is usually very flat, geomorphologically it is a landform composed primarily of unconsolidated depositional material derived from sediments being transported by the related stream and hydrologically it is subject to periodic inundation [281].

Floodplains are often covered by various land uses and consist of water bodies such as wetlands, swamps, streams, lakes weirs and dams. The floodplains of GBR catchments are characterised by agricultural land uses (e.g. sugarcane, banana, cotton) and grazing. The soils of the catchments are typically of low fertility and exhibit poor soil structure, which has resulted in extensive application of fertiliser and conditioner to maintain agricultural productivity. Since the 1960's, there has been a dramatic increase in the application of nitrogenous and phosphate fertiliser use in the Great Barrier Reef Catchment Area. Over half of the total volume of nitrogen and phosphorus fertiliser used in agriculture within the Great Barrier Reef Catchment Area is applied to sugarcane [99].

Floodplains are increasingly recognized as important component of the sediment budget in many fluvial systems [86, 151, 204]. A number of studies have demonstrated that floodplains can be dominant sources or sinks of sediment-bound contaminants in rivers [67, 104, 188, 210]. Today a very small proportion of sediment brought down the river in floods accumulates on the modern floodplain as it sits significantly higher than the river and therefore there is no accommodation space [26]. Geological processes such as erosion and sedimentation redistribute toxic pollutants introduced to the landscape by mining, agriculture, weapons development, and other human activities. A significant portion of these contaminants is insoluble, adsorbing to soils and sediments after being released. Much of this sediment is stored in river floodplains which can transport to downstream river during floods [186]. Sediment-bound contaminants preferentially adhere to fine-grained particles such as silt and clay, because of their large surface area-to-volume ratios and the high chemical activity of clay minerals. Fine-grained particles are generally transported through rivers in suspension and can be deposited on river bars and floodplain surfaces during overbank flooding. Thus, contaminated sediments tend to accumulate in floodplains adjacent to river channels, and these deposits become important nonpoint sources of downstream pollution as well as local sources for assimilation into plants and animals.

FACTORS INFLUENCING THE FILTERING CAPACITY OF VEGETATED SYSTEMS

Many factors influence the ability of vegetated areas 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 [237]. Riparian buffers must be properly constructed and regularly monitored in order to maintain their effectiveness [110, 170, 212, 234, 238, 252, 259, 265, 296, 297, 309, 328]. 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 [13, 95, 119, 140, 160, 234, 252, 331]. Maintaining shallow sheet flow into the buffer can be especially troublesome in areas where slopes are steep and surface flows tend to be funnelled.

Vegetated areas including vegetated or grassed drains, riparian areas, constructed and natural 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 runoff quality by changing the flow hydraulics, increasing 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 (Figure 14) [7, 28, 45, 73, 81, 95, 108, 110, 113, 114, 119, 127, 133, 140, 161, 172, 207, 209, 234, 241, 246, 251, 273, 280, 287, 288, 295, 328].

The effectiveness of buffers is particularly determined by:

1. Structure/species of vegetation.
   [27, 45, 93, 114, 161, 162, 169, 170, 172, 180, 203, 205, 209, 215, 234, 238, 241, 245, 246, 252, 255, 264, 265, 267, 268, 272, 273, 288, 295, 297, 303, 309, 331].

2. Depth of surface water versus vegetation height and density.
   [13, 140, 172, 255].

3. Hydraulic conductivity and holding capacity of buffer zone soils.
   [9, 110, 234, 295].

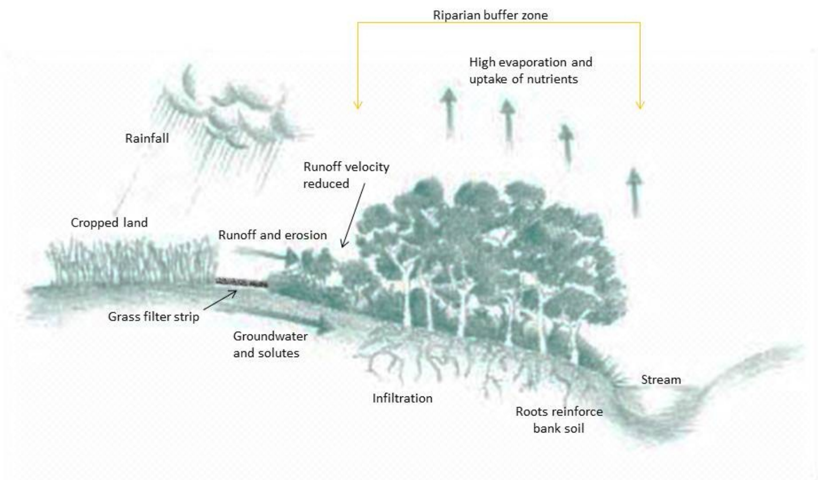


Figure 14: Water quality functions of a grass filter strip and riparian buffer zone.

Source: [175]

Vegetated buffer zones are effective at removing pesticides and some nutrients at the property scale, but not at the broad catchment scale without being part of a strategic 'maze' of filter strips installed across the catchment [234, 252, 267]. However, this still does not account for pollutants lost via infiltration to groundwater.

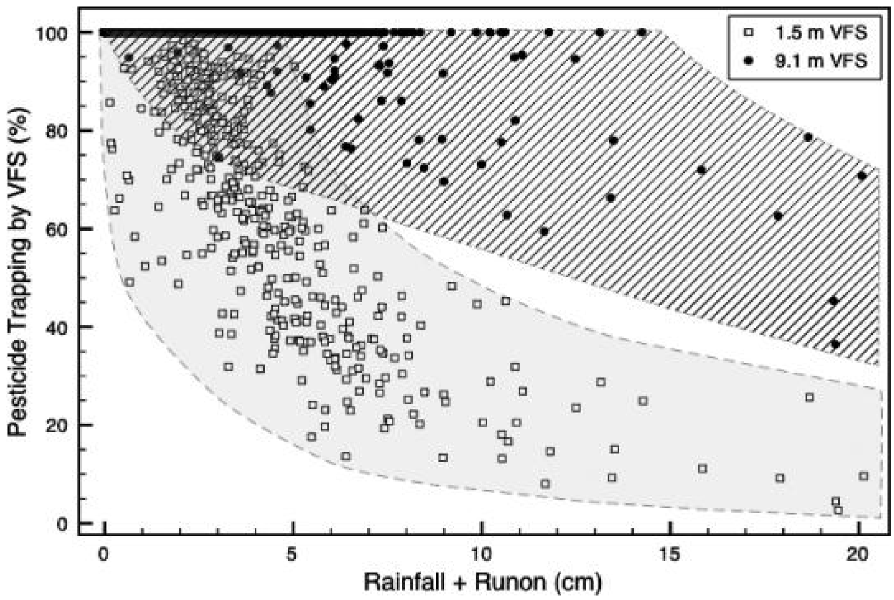
As one of the main forms of herbicide trapping is via sediment trapping, the long-term performance of buffers in sediment trapping is very relevant. Researchers have observed that the effectiveness of grass filter strips may decrease over time as the strip becomes inundated with sediment or as the ground 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 [80]. 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 of the field edge (as little as 3 m), while the removal of silt particles required a buffer of at least 15 metres [326]. Filter strips 100 to 130 metres wide were required to remove clay particles.

5.1 Influence of climatic conditions

The effectiveness of buffers is determined by rainfall characteristics (total amount and intensity) [9, 13, 27, 28, 56, 87, 95, 96, 110, 133, 170, 171, 234, 241, 246, 251, 252, 255, 277, 295, 297, 317, 331]. Heavy rainfall events causing storm runoff are always associated with the production of extremely large volumes of on-paddock water to manage in a short period of time. In many circumstances, these large water volumes may not be retained by the buffer strip (regardless of its characteristics), and erosion channels or rills formed during these conditions may further jeopardize the long-term positive effect of buffer zones. This 'hydrological dilemma' may result in unavoidable pollutant pulses, including pesticide contaminations of surface waters, under conditions where other measures, like pre-application and precise application strategies, are not applicable or do not provide the necessary benefit [276].

In an experimental study, Popov *et al*. [255] showed that biofilters were still effective in conditions of intense runoff (simulated rainfall) from cropland (run-on depths of 160, 320, and 800 mm over 0.7 and 0.8 hr) over grassed strips 4 m long by 1.25 m wide. Pot *et al*. [256] simulated several rainfall intensities (0.070, 0.147, 0.161, 0.308 and 0.326 cm/hr) and found that at differing intensities, contrasting flow patterns within the soil emerged. At the highest rainfall intensity, there was rapid flow through larger (macro-) pore pathways, with slower flow through regions of medium (meso-)porosity and no-flow in remaining micropores. At lower rainfall intensities, macropore flow was not active anymore. Therefore, in high rainfall intensities, there is preferential fast transport of pesticides through porous soils and even in low rainfall conditions, macropores allow pesticides to potentially bypass the surface layer of the grass filter strips [256], so care needs to be taken when trying to determine effectiveness of buffers and underlying soils in trapping pesticides.

Using input data from field trials in the United States [276], Sabbagh *et al*. [277] showed a nonlinear relationship between total water input (rainfall plus run-on) during a storm event and percent pesticide reduction using a predictive model. It was also identified that differences in soil moisture affects 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 run-on from the source area. For example, for a 9.1 metre-long buffer, pesticide trapping or reduction was generally greater than 60 percent unless the total water input (rainfall plus run-on) during a storm event exceeded 10 cm (Figure 15). Note that the K_{oc} included in the study of 100 L/kg is the same as for atrazine.



Note: Predicted using the VFSMOD model relative to two different buffer lengths (1.5 and 9.1m). Data shown are for a pesticide with organic carbon sorption coefficient, Koc = 100L/kg OC.

Figure 15: Nonlinear relationship of pesticide trapping (%) by a vegetative filter strip (VFS).

Source: [277].

In the Wet Tropics of 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 *et al*. [199, 200] 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 sugarcane growing area of 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.

The role of vegetated riparian buffer zones in mitigating sediment, nutrients and other contaminated agricultural runoff in humid tropical regions is little examined [247]. Most studies have been conducted in temperate regions. During the wet season, processes within a riparian zone in the Australian tropics would be expected to be affected by large pulses of water flowing through the riparian zone. Internal drainage is an important issue in high rainfall districts, and many soils in the tropics have high infiltration rates, such that rainfall may carry fertilisers or chemicals down through the soil where it may enter the groundwater or emerge in streams at a later date [182]. Flood events may result in very short residence times of water, and therefore reduced riparian buffering capacity [195].

Losses of cane trash, as well as soil and attached nutrients into watercourses in the tropics can be particularly high during periods of intense rainfall. Movement of soil particles, nutrients, and vegetation debris from agricultural lands to waterways in runoff is a particularly important pathway along the tropical coast with its periodic, high-intensity rainfall events. Dissolved nutrients, organic materials and other contaminants can also move through soils in sub-surface flow, and enter streams along bank faces or even beneath the water level in base flows [175]. Modest grass buffer strips suitable for other climates may be much less effective at trapping sediment and nutrients in the agricultural areas of tropical Queensland, where rainfall intensities are much higher than in the south, and may be combined with erodible soils. In high-intensity rainfall, significant surface flow can become concentrated, and can overload simple grass buffer strips, as such riparian vegetation may be more effective [175].

In the wet and dry tropics of Queensland, monsoonal rainfalls drive very seasonal flows in the rivers and across the landscape (see review [314]). Those wetlands not exposed to irrigation tailwater experience most of their rainfall and incoming flow during short time periods (a few months) during the wet season and then drawdown (i.e., dry out) during the dry season. During the high rainfall months (December to March) wetlands in the tropics are at risk of flooding and scouring, which is problematic for retention of contaminants, especially needed during the 'first flush' of the season. Flushing of the wetlands by excessive or overland flow needs to be avoided, which can be difficult in the tropics where monthly rainfall can exceed 1400 mm (Cairns; Bureau of Meteorology, Queensland, Climatological Summaries). However, constructed wetlands are often designed with an overflow bypass to prevent flushing and scouring of the wetland, and consequently, remobilisation of sediments, phosphorus and other sediment-bound pollutants.

Effective wetlands are designed to insure adequate retention time of irrigation tailwater and rainfall, but to also provide aquatic habitat for invertebrates and fish which play a role in wetland functioning and mosquito control [106]. Therefore, in the dry season, it is important for wetlands to not dry out completely, which would mean loss of vegetation and aquatic food webs; aquatic vegetation and associated food webs are important for the conversion of nutrients. Drawdown, or drying out, of constructed wetlands is often allowed periodically to consolidate soils, and conduct weed control and maintenance operations, however, several studies have reported that phosphorus remobilises after soils are allowed to dry out and the wetland is re-flooded (e.g., [3, 236]). However, if soils are kept moist, sequestration of phosphorus is maximised (e.g., floodplains vs. freshwater wetlands [30, 31]). Seasonal drawdown can be useful in preparing the wetland to capture irrigation and first-flush events, though it is still important that large macrophytes are maintained to capture early storm runoff and be prepared for uptaking the first flush of nutrients into the system with the first rains of the wet season.

There is strong seasonality to how effective wetlands are in removing or reducing pollutants, due to both temperature and flow rates. Nitrate retention and biological removal is temperature-dependent and is most efficient in the summer (e.g., northern hemisphere [300]), or around 20-25°C [304]. Nitrogen retention efficiency is considerably lower during floods [300], which is likely due to low residence time during flood events. During high flow periods, agricultural wetlands may only be expected to retain between 10 and 20% nitrogen and phosphorus [261].

The effectiveness of floodplains as a sediment and nutrient sink is greatly influenced by rainfall volume and intensity [41, 170, 331]. Heavy rainfall events produce large volumes of water and cause water to move faster [148]. These large volume of water stay for a short period of time on the floodplain [147] and reduce the chance of infiltration and other biological transformation for nutrients as well as produce more sediment by land erosion [148].

The climate within the GBR catchments ranges from tropical to sub-tropical [99]. Rainfall is distinctly seasonal, particularly in the northern half of the Catchment where the monsoon influence during the summer months (September to March) produces high rainfall, particularly when associated with the development of tropical cyclones. The eastern Queensland basins may experience extended years of drought and decreased cyclonic activity during low-index years (El Niño), followed by periods of exceptionally high rainfall during high-index years (La Nina) [174]. During the El Niño drought conditions, there is concern of soil compaction and structure decline on grazing land, due to trampling by hard-hoofed grazing stock, such as sheep and cattle [270]. The drought is often followed by unusually high rainfall and runoff rates associated with the La Nina cycle. Sheet-wash, rill and gully erosion over large areas of the tropical rangelands contributes to high sediment yields and reduces the effectiveness of floodplain in removing the pollutants [270]. The studies by McKergow *et al*., [201, 202] have shown that intense rainfall is one of the key factors that reduce the effectiveness of a vegetated system (e.g. riparian buffer strips). The study site is located in the north Johnstone River catchment which receives average rainfall of 3585 mm during the wet season (December to April). The study by Karim *et al*. [148]shows the Tully-Murray floodplain produces large sediment and nutrient loads and one of the key reasons is the high rainfall that produces frequent floods.

5.2 Influence of soil types

The effectiveness of buffers is determined by soil properties [93, 95, 110, 114, 160, 255, 276, 295], and initial soil water content [110, 238, 249, 277, 295]. 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: a) a high or perched water table; b) alternating periods of aerobic and anaerobic conditions; c) healthy populations of denitrifying bacteria; and d) sufficient amounts of available organic carbon [177, 179], as well as an ample supply of nitrate.

Other mechanisms for nitrate removal include uptake by vegetation and soil microbes and retention in riparian soils [18, 89]. 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 [248]. In North Carolina, researchers estimated that only 3 to 6 percent of the nitrogen passing through an alluvial swamp forest was taken up and stored in woody plant tissues [30]. Nevertheless, Correll [66] 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 mineralised and denitrified by soil microbes.

Riparian vegetation does not align with a particular soil type, though tends to have a greater proportion of fines (clay/silt size particles) than upland environments. In general, clay soils will retard subsurface movement of agricultural contaminants through riparian communities to aquatic environments, with low infiltration rates. Conversely, sandy and gravel soils, promote high infiltration and subsurface flow. Soil that is compacted or saturated can minimise the efficiency of riparian vegetation buffers [55].

Soil characteristics influence hydrologic flowpaths and can impact upon the rate and magnitude of subsurface nutrient removal. For example, riparian forests which are characterised by a shallow impermeable layer, forcing groundwater to move laterally through shallow root zones and organic-rich soil, are considered ideal for nitrate attenuation [130]. Preferential flowpaths result from differing hydraulic conductivities of aquifer materials, which in turn modify surface and groundwater interactions [131].

The effectiveness of riparian buffers also depends on the soil type from which the runoff is produced and rainfall energy is a primary source of aggregate dispersion. Sediment trapping efficiency is generally reduced as sediment size decreases [166].

The soils of natural wetlands have developed under wet conditions and so are influenced by climatic region, wetland system type and inundation frequency and only change slowly over time [46]. The defining characteristics of wetland soils are the accumulation of organic (decomposed plant) material, the presence of sulfidic material and gleyed soil matrix colours [44]. Many wetlands exist because they overlie impermeable soils or rocks and there is little interaction with groundwater [46].

A beneficial aspect of wetland soils is that in the oxygenated part of the wetland system nitrogen compounds are converted to nitrate by soil microbes then nitrate is reduced by microbes to nitrogen gas when it moves into the anaerobic zones of the wetland [244]. The oxidised soil–water interface of wetland sediments can also intercept and hold large influxes of dissolved phosphorus which is slowly released as required by wetland algae and plants [118, 244, 253].

It is estimated that 666,000ha of acid sulphate soils (ASS) occur within lowland coastal areas of the Great Barrier Reef (GBR) catchments (generally below 5 m Australian Height Datum (AHD)) and their close proximity to the GBR makes them a substantial threat to reef water quality [257]. They are found in areas usually targeted for agriculture and development [257]. These soils are problematic for wetland and downstream water quality because, if they are exposed to oxygen rich air or water oxidation occurs and sulphuric acid is produced which can cause fish kills and aquatic disease outbreaks. Acid sulphate soils also release bioavailable, soluble iron, which can stimulate harmful algal blooms, such as *Lyngbya majuscula* (fireweed) along the coast [257]. Further, low pH (< 5), inhibits denitrification processes that would otherwise occur in the wetland [225]; pH is lowered with the production of sulphuric acid.

Soil composition in constructed wetlands plays a significant role in facilitating or limiting denitrification processes. For instance, carbon availability is limiting to denitrification. The availability of oxygen in the soil also directly impacts denitrification and phosphorus mineralisation, as well as biological uptake of nutrients. For example, in a reduced oxygen environment ("reducing" environment), particulate phosphorus can be changed to dissolved phosphorus which can then be taken up by plants (stems and leaves) and algae.

In terms of pesticide uptake, Chlorpyrifos, which has low water solubility, sorbs to sediments quite readily and in one study, more than 50% of the initial concentration was associated with sediments [224]. Sorption to sediments is an important pathway for limiting pesticide transport downstream as long as these sediments are not dislodged or remobilised during flood events or from other disturbances.

Soil and water quality are very closely linked and, to a significant extent, soil properties determine water quality [17]. As water passes through soil it is filtered and purified which helps to generate clean and healthy groundwater. This process also includes the removal of nutrients thereby reducing the risk of water eutrophication (the process by which water bodies become enriched by nutrients. A good quality soil can store plenty of water that helps slow release of water flow thereby reducing the risk of flooding. Soil organic matter is an extremely important component of soil [57]. It improves nearly all soil properties (e.g. moisture retention, soil structure, drainage, nutrient storage) and therefore plays a vital role in many functions of soil [312]. The ability of soil to store carbon is important in reducing the amount of carbon dioxide (CO_2) in the atmosphere. Soil organisms continually breakdown complex organic molecules into simpler organic molecules and when the process is complete they are released as nutrients and gases, including greenhouse gases such as CO_2. However, soil organisms are also involved in a process called humification where new, more complex and stable organic matter is formed. In some soils, notably peats, organic matter breakdown does not occur completely owing to the high acidity and water content, which results in the accumulation of organic matter in the soil.

The effectiveness of any vegetated system is determined by soil physical properties [114] and water content [250]. As discussed earlier, certain soil conditions must be met for denitrification to occur. These include high water table, alternating periods of aerobic and anaerobic conditions, healthy populations of denitrifying bacteria and sufficient amounts of available organic carbon [177, 179]. Other mechanisms for nitrate removal include uptake by vegetation and soil microbes and retention in riparian soils. Plants can take up large quantities of nitrogen as they produce roots, leaves, and stems. The agricultural soils of the GBR catchment have a relatively low nutrient status resulting in high artificial fertiliser applications under both cropping and improved pasture for grazing [99].

5.3 Influence of residence times

The residence time of a system is the approximate time that a parcel of water will remain within the system. A longer residence time is an indication that solutes or suspended material reside for a longer time within the system. Generally, when the residence time is combined with biological and chemical measurements, the quantitative effect on the water body caused by the particles and solutes can be determined. A long residence time could, for example, allow increased concentration and accumulation of pollutants within the system.

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. However, chemical breakdown (i.e., by bacteria, light, etc.) of most pesticides will require weeks to years depending on the half-life statistics of the particular chemical Table 4.

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.* [195] for trapping of nitrate in Kyambul lagoon on the Tully Murray floodplain (essentially no reduction in nitrate loads) and by Connor *et al.* [64] for riparian trapping of nitrate in the Mulgrave catchment.

Increased residence time favours retention of sediment and pesticides, and improved processing and uptake of nutrients by riparian vegetation. Greater residence time allows prolonged contact with vegetation, roots and organic-rich riparian soils, promoting uptake, absorption, infiltration and transformation [130]. Plants are known to particularly increase the residence time of nutrients by reducing their mobility [124]. For example, Kaushal *et al*. [150] showed that mass removal of nitrate increased linearly in riparian forests in a Maryland, US study, with increase of groundwater residence time (Figure 16). Rate of nitrate removal by restored (rehabilitated) streams was also significantly greater than for unrestored streams.

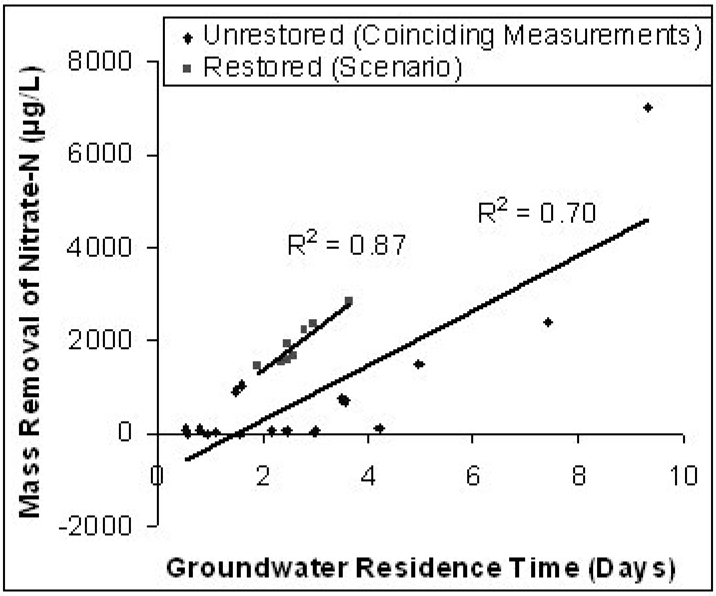


Figure 16: Groundwater residence time in riparian forest versus nitrate removal.

Source: [150].

The tropical climate is subject to great variation, with distinct wet and dry seasons [10] that mean great variation in surface and groundwater flows and therefore residence times for contaminants. For example, studies suggest that, during wet periods, denitrification as a form of nitrogen removal from runoff may not be significant as the residence time of water in the aquifer is greatly reduced [156]. However, in other cases, water flow in shallow aquifers reversed during high flow periods, increasing residence time and denitrification [266].

A comprehensive study conducted by Bullock and Acreman [46] of 196 wetland studies found that floodplain wetlands reduce, or delay floods by retaining water in the landscape and two thirds of the studies concluded that wetlands increase average annual evaporation, evaporating more water than other land types such as forests, grasslands, agricultural land, and reduce average annual river flow and particularly reduce flow of water in downstream rivers during dry periods.

Unlike natural wetlands, constructed wetlands provide the opportunity to manipulate flows into and out of the wetland, thereby increasing residence time and the potential for denitrification and degradation of pesticides (see [195]). In terms of denitrification in sediments, direct limitations are the water column concentration of nitrogen and its rate of diffusion into the sediments [101, 153]. Kjellin *et al*. [153] also found that denitrification decreased with increased residence times, further supporting the idea that nitrate is limiting to the denitrifying community and is controlled by water column exchange and in situ (sediment) nitrate production. With agricultural inputs, water column nitrate is usually not limiting. However, at least 5 days (120 hrs) residence time is required to promote denitrification in treatment wetlands [144] and a wide range of days is required for the breakdown of pesticides, due to the variety of half-lives (for soil half-lives see Table 4; for marine half-lives see [229]. Pesticide half-lives and retention varies depending on the chemical make-up and initial concentration of each pesticide, but also the environmental and treatment conditions. For example, several studies report pesticide half-lives and pesticide retention times (PRT), defined as the amount of time needed for an initial concentration to be reduced to a final target concentration, for various studied wetlands [218, 219, 222-224]. One example is that of atrazine, which had a PRT in constructed wetlands in Mississippi (USA) of 30-39 and 91 days for 73 and 147 µg/L initial concentrations, respectively, to reach the target 'no observed effects' concentration of 20 µg/L [222].

Properly constructed wetlands are designed with residence times in mind; for example, a constructed wetland in the Johnstone region was designed to retain flow for 48 hours. Smaller wetlands might be flushed with the first big rain (i.e., first flush), which also usually transports the highest amount of contaminants (80 % pollutant load within the first 30 % volume of discharge [20]), so considerations as to how to contain this first flush need to be built in to the design of constructed wetlands, especially given the variable flows in tropical North Queensland.

The degree of pollutant removal in the floodplain greatly depends on water residence time [38, 313]. Long residence time allows sediment (with attached pollutants) to deposit, water (with contained dissolved pollutants) to infiltrate, dissolved pollutants to sorption onto soil and vegetation. Residence time required for filtering process differs based on types of pollutants, for example sedimentation of coarse particles is occurred in minutes, for silt in hours and for fine sediment in days. For pesticides, chemical breakdown require weeks to years depending on the half-life statistics of the particular chemical (Table 4). Residence time increases with the length of floodplain and decrease with flow velocity.

The residence times of these flow events varies between catchments in the range of a few days for a small river catchment (e.g. Ross River, Tully River) to a few weeks and up to a few months for the two largest GBR catchments, the Burdekin and Fitzroy [38, 313]. While there have been a number of previous studies to estimate residence times of water in the GBR lagoon (e.g. [62, 163, 181, 313, 320]) using hydrodynamic modelling and remote sensing technologies, studies on estimating residence time on the floodplain environment are still limited for the GBR catchments. The residence time in the floodplain however differs greatly from the mean flood speed due to the complex nature of floodplain flow [128]. A recent study by Karim *et al.* [147] for the Tully-Murray catchments found that residence time varied greatly between locations on the floodplain ranging from 1 day to 12 days for a mean annual flood. The floodplains of wet tropical catchments contribute less in filtering pollutants because of high flow velocity and less width of the trapping medium [64, 194, 195].

Particles stored in floodplains generally have long residence times as compared with channel sediment because they are less accessible to erosion. Because many environmental contaminants break down through processes such as radioactive decay or bioprocessing, their long-term fate is controlled by the relative timescales of contaminant degradation and particle residence time in the valley floor [186].

5.4 Influence of physical characteristics

The effectiveness of buffers is determined by:

- **a.** the length, gradient and shape of the upstream runoff area (width and slope) [24, 28, 45, 95, 110, 114, 140, 161, 170, 172, 207, 215, 234, 246, 249, 252, 255, 277, 295, 301, 331].
- **b.** and the rate of surface water flow [13, 27, 28, 42, 73, 81, 96, 108, 113, 114, 140, 172, 205, 216, 234, 251, 255, 256, 302, 317].

A recent study in the Mulgrave catchment [64] 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 sites on planar slopes were able to withstand peak discharge events as the planar slopes allowed flow to disperse, assisted by the grass vegetation on a low gradient slope which reduced 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.

Selecting an appropriate buffer size often involves consideration of several desired functions, site conditions, and what is economically or practically feasible. 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 affects water infiltration. Amount of runoff is affected by source area size and properties as well as rainfall intensity and quantity. Many studies have investigated sediment trapping efficiency of grass buffers [209, 238, 245, 271, 276, 297, 306, 331]. For example, in a recent review by Yuan *et al*. [331], 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 logarithmic models (Figure 17). According to this relationship, a 5 m buffer can trap about 80 percent of incoming sediment [331]. Table 5 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.

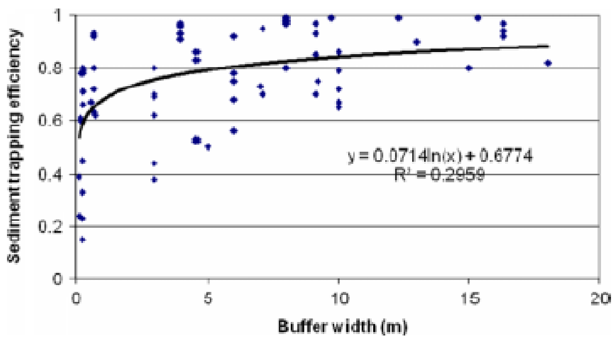


Figure 17: Buffer width and sediment trapping efficiency.

Source: [331].

Table 5: Trapping efficiencies for various buffer widths and rainfall conditions.

Reference	Conclusions	Most efficient buffer width (m)	Rainfall conditions
[280]	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.	15	Simulated - 25.4 mm in 30 mins
[80]	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.	10	Simulated – 50 mm/hr. Virginia
[184]	30 and 15 foot strips of fescue grass 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.	10	Simulated – 48.25 mm/hr.
[58]	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.	>15	Various
[309]	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.	>10	Various – USA conditions.
[309]	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).	15	Various – USA conditions.
[134]	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.	6	Simulated. Montana foothills, USA
[331]	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.	>6	
[246]	12-m wide grass filter strips provided an almost optimal reduction of herbicide output from arable fields via surface run-off.	12	Annual rainfall 1072 mm; 988, 1309 and 1236mm in 1997, 1998 and 1999 respectively.
[249]	Reductions in solution concentrations and mass retention of P and two herbicides (atrazine and picloram) were observed for simulated flow within 10m wide forested filter strips across a range of slopes and organic horizon conditions in coastal Piedmont of Georgia.	10	Simulated wet and dry. – annual ~ 1300 mm, summer ~500 mm, max daily 250 mm.
[245]	Grassed buffer strips effective in restricting pollutant transfer in runoff; those with widths of 6, 12 and 18 m reduced runoff volume by 43 - 99%, suspended solids by 87 - 100%, lindane losses by 72 - 100%; atrazine (and its metabolites losses by 44 - 100%.	6, 12, 18	Various
[241]	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.	8	Kinston, North Carolina – annual average ~1200 mm
[238]	Vegetated Filter Strips 6m wide were very effective in reducing runoff volume and concentration during both wet and dry years, in comparison to 3m wide strips.	6	Annual average ~805 mm (NE Italy)
[234]	Small controlled runoff plots with buffers 5-10 m in width were successful in removing a variety of pollutants from overland flow	5-10	Simulated

Reference	Conclusions	Most efficient buffer width (m)	Rainfall conditions
	(sediment, nutrients, chemicals).		
[297]	Tested the effectiveness of different tillage systems and buffer widths. Most effective was no till, 45 ft vegetated filter – reduced runoff volume (91 %), suspended sediment (99 %), nitrate-N (97 %) and atrazine (98 %).	~ 14	Kentucky - annual average rainfall 1350 mm
[199]	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 15 m 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 trap for sediment and nutrients. Dense grass riparian buffer strips > 15 m 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.	>15	Innisfail Qld, ~ 3585 m average annual rainfall.
[209]	The first 5m of the vegetative filter strip played a significant role in removal of suspended solids (> 40 µm particle size). Length > 10 m did not significantly improve vegetative filter strip performance. Infiltration was the only mechanism that allows for removal of smaller size sediments (< 40 µm). Vegetative cover helped to reduce velocity of runoff and increase residence time for water to infiltrate. High vegetation density led to less erosion and less transport capacity of the runoff and therefore greater settling of sediments. Non-submerged vegetation allowed 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.	5-10	Study sites were within the Rock Creek Watershed (Newton), Iowa.
[271]	The initial 3.0 m of the vegetative filter strip removed more than 70 % of the sediment from runoff while 9.1 m of the vegetative filter strip removed 85 %. There was little decrease in sediment concentration with greater vegetative filter strip widths. Slopes of 12 % grade had greater runoff and soil losses with all vegetative filter strip widths than the 7 % grade. Vegetative filter strips promoted infiltration, reduced runoff volumes, and decreased runoff sediment concentration.	3.0-9.1	Study sites were in northeast Iowa, U.S.
[276]	Filter strip width was not a statistically significant parameter in the empirical model created to predict pesticide trapping efficiency, based on several studies. Hydrological and sediment input parameters, such as, runoff volume reduction (infiltration), sediment reduction, phase distribution factor (i.e., pesticide phase, either sorbed or dissolved), and percent clay content, were all significant.	N/A	Various
[306]	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. Filter strips, regardless of width, reduced cumulative runoff and sediment loss at least 46 % and 83 % respectively. The highest surface runoff was from the unfiltered treatment. Sediment losses were reduced 98-99 % with filter strips.	4	Study sites were in Mississippi, U.S.

Several site characteristics may dictate wider buffers, especially when trying to maximise 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. Studies in Iowa found that water infiltration and trapping of dissolved herbicides by buffers was least effective when previous rains saturated the 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 shown 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 2 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*. [306] compared tall fescue grass buffers measuring 0.5 to 4 metres wide placed downslope of ~20 m-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.

Current research (Table 5) suggests that a buffer width of at least 6 metres and in many cases, 10-15 metres, provides 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 *et al*. [198] and others [199, 200, 202, 203] provide important information regarding the effectiveness of vegetated buffers in trapping materials in a high rainfall area. Karssies and Prosser [149] have also determined indicative soil losses and designed filter widths for 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 6. It is clear from this information that buffer widths in the Wet Tropics (800-5000 mm annual rainfall) in areas where there is poor cover (C = 0.2) must be at least 30 metres to minimise soil loss. In areas with good cover (C = 0.01), buffer widths between 2 and 12 metres are required, depending on the site characteristics (i.e., 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 6: Indicative soil losses and design filter widths for the Wet Tropics and Burdekin regions.

Region (annual rainfall) (mm/y)	**Rainfall erosivity**[1]	**Soil erodibility**[2]	**Slope**[3]	**Poor cover soil loss**[4] (t/ha/y)	**Filter width** (m)	**Good cover soil loss**[4] (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
	Very 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
	Very 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).

Source: [149].

Slope, topography and riparian vegetation width are key physical controls on the effectiveness of riparian buffers. These characteristics often interact to affect retention effectiveness, as discussed below.

A number of researchers have found correlations between riparian buffer strip width and contaminant (pesticides, nutrient, heavy metals) abatement in waterways [1, 5, 191]. A study by Dukes *et al*. [84] on four relatively narrow riparian buffers reported that wider plots (15m) decreased nitrate levels 15% more than narrower plots (8m), with differences attributed to increased residence times through the buffer. Mayer *et al*. [191] also estimated buffer nitrate reduction through a meta-analysis of 89 riparian zones with variable widths. Nitrate reduction was found to significantly increase as widths increased from 0 to 25 m. However, increasing width from 25 to 50 m did not significantly increase nitrate removal, suggesting a plateau in effectiveness for the role of riparian width in controlling nitrate removal. The width of vegetated buffers has also been considered particularly important for reduction efficiency of runoff-related pesticide inputs to streams [258].

Sediment trapping efficiency is also affected by slope, but the overall relationship is not consistent among studies (Figure 18) as other factors such as riparian width have an interrelationship.

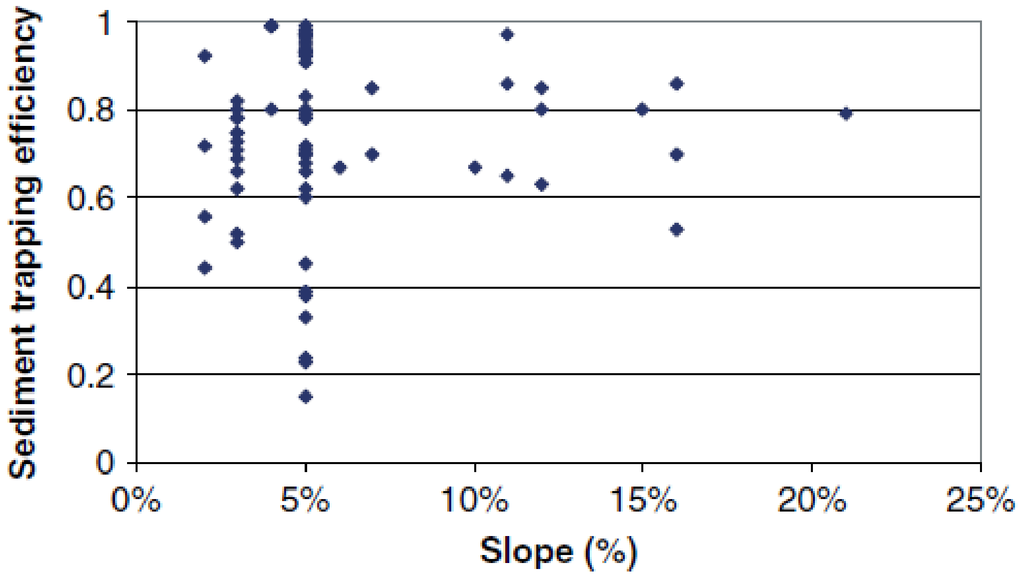


Figure 18: Slope and sediment trapping efficiency in vegetated buffers.

Source:[331] .

Results indicated that under conditions of relatively shallow flow not concentrated in channels, gently sloping, densely vegetated riparian buffers of 3 m width are likely to limit transport of sediment from uplands to streams [23], whereas moderately steep, less densely vegetated buffers of 3m may be vulnerable to much higher rates of sediment delivery [68]. Buffers greater than 6 m are effective and reliable in removing sediment from any situation; for example, Hook [134] reported that more than 97% of sediment was trapped in the rangeland riparian buffer area with a 6m buffer in any of the experimental conditions they studied.

Generally, riparian buffers of 4–6m width can reduce sediment loading by more than 50% ([23, 25, 68, 165, 184]. However, the efficiency is likely reduced on slopes above five degrees due to flattening of understory and loss of surface litter by surface runoff during high rainfall. Narrower buffers are much more effective for less erodible soils types. Buffers greater than 6m are effective and reliable in removing sediment in most settings; for example, Hook [134] reported that 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.

In the sugar-growing areas of northern Queensland where riparian vegetation is intact, it has been recommended for a riparian width of 25 m to be retained for effective trapping of nutrients and sediments on small creeks with slopes approaching or exceeding 5%, with larger streams requiring larger buffer widths for effective trapping [175]. Where significant cane trash or soil may move through a cane paddock, a 6m additional grass buffer strip between the cropped area and the retained riparian vegetation is recommended. To limit pesticide movement into stream waters, retention of a riparian width of at least 25m is recommended, with a mixture of trees, shrubs and other understory. Where lands are already cleared, revegetation of a minimum of 10m of riparian vegetation may be suitable for the region [175].

Where topography is complex, surface runoff is often concentrated in fields and flows through only parts of riparian buffers, reducing their efficiency [68, 82, 126]. Sediment and associated nutrients and agricultural chemicals can be delivered directly to a stream along concentrated flow paths (CFPs) that transect the riparian zone [155]. Studies indicate that under conditions of relatively shallow flow not concentrated in channels, gently sloping, densely vegetated 3 m buffers are likely to limit transport of sediment from uplands to streams [23, 165, 264, 271], whereas moderately steep, less densely vegetated buffers of 3m may be vulnerable to much higher rates of sediment delivery [68].

The presence of the stream at a topographical low point in the landscape complicates connections between surface water and groundwater, as groundwater flow direction generally turns in close proximity to the stream, becoming closer to the direction of stream flow [131]. Streams generally gain water when the watertable gradient is angled downward towards the stream, while loss of water from the stream to an aquifer occurs when the pieziometric surface slopes away from the stream. Flow-through occurs when the hydraulic head on one side of a stream leads to discharge while the head on the other side leads to recharge [131].

The physical position and distance of the riparian buffer zone in relation to cropped fields is also significant. McKergow *et al*. [199] studied the effectiveness of riparian zones in the humid tropics at trapping sediment and nutrients transported in surface waters. They concluded that the position of the riparian buffer zone in relation to cropped fields was important in trapping ability, particularly for sediment [199].

The area of a constructed wetland needs to be sized to contain low-normal-high flows and especially first flush flows, but also needs a bypass for overland flows. Constructed wetlands can be designed with various shapes and depths, including U-shaped, V-shaped and rectangular, and including horizontal, subsurface or vertical flow, and deep pools or shallow marsh regions. Sizing of treatment wetlands to gain 40 % reduction in TN, 50 % reduction in TP, and 60 % reduction in TSS, needs to be between 5-7 % of catchment (including 30 % pollutant reduction from a change in on-farm practices). To make a 50 % reduction in TN, a particular farm would need a wetland sized at 15 % of the catchment [78] (Qld Wetlands Program).

Appropriate sizing is important for retaining pollutants, and is specific to the pollutant of concern, as well as regional characteristics. As an example, in Mississippi wetland trials, for adequate atrazine reduction (initial concentrations of 73 and 147 µg/L reduced down to 20 µg/L), given aqueous and soil-sorbed pesticide half-lives discussed above, wetlands or buffers need to be between 100 and 281 m in length [222], whereas metolachlor requires between 100 – 400 m wetland length for effective mitigation [223].

Prevailing winds or fetch along wetlands can cause re-suspension of particulate phosphorus into the water column, and potentially move it through the wetland and downstream. Thus, placing the outflow of the wetland at the upwind end helps minimise nutrient release from decaying plant matter and resuspended particulates that otherwise would pool up downwind [94].

The soil exerts an important influence on water quality. Its capacity to use, retain, or reduce the undesirable effects of pollutants varies significantly according to the physical, chemical, and biological properties of the soil and the characteristics of the contaminants involved. The effectiveness of any river-floodplain environment in absorbing sediment and nutrients loads greatly depends on physical properties of the system such as gradient, morphology, soil types and width of the trapping medium [164]. For example, the more permeable the soil surface is, the more easily water can infiltrate. However, in the Wet Tropics infiltration can be limited due to the high groundwater level and flow velocity. The movement of metals in the floodplain is mediated by many factors including the solubility of the contaminant, the pH of the floodplain, climate and floodplain permeability and hyporheic exchange [67, 85, 183, 188].

5.5 Influence of vegetation

The presence of vegetation can exert a strong influence on the downstream movement of agricultural run-off. Ditch half-distances were calculated in a California study, where it was found that a non-vegetated ditch would require three times the distance of a vegetated ditch to remediate the same diazinon load [215]. In a study on methyl parathion transport in vegetated vs. non-vegetated constructed wetlands, no parathion was detected in the outflow of the vegetated wetland, whereas parathion was detected in the outflow of the non-vegetated wetland 30 minutes after release [216]. A study from California showed that V-shaped vegetated ditch length needed to be 2.3-2.8 times less than non-vegetated V-shaped ditch length to reduce initial loads of pesticide by 50 % (depending on the type of pesticide) [215]. From these and other similar studies, vegetation seems to be fundamental to the effectiveness of uptake of pollutants. However, the type of vegetation is also important to consider since particular types (e.g., species, growth forms, height) of grass provide more or less uniformity in ground cover and overall density and also provide varying results in terms of bioremediation of pesticides [114, 133, 169, 242, 264]. Vegetation also influences settlement of suspended solids by creating a trapping mechanism (Figure 19) [221]. Moore *et al.* [221] found that vegetated ditches trap suspended solids significantly better than non-vegetated ditches, by increasing friction and turbulence in stream channels.

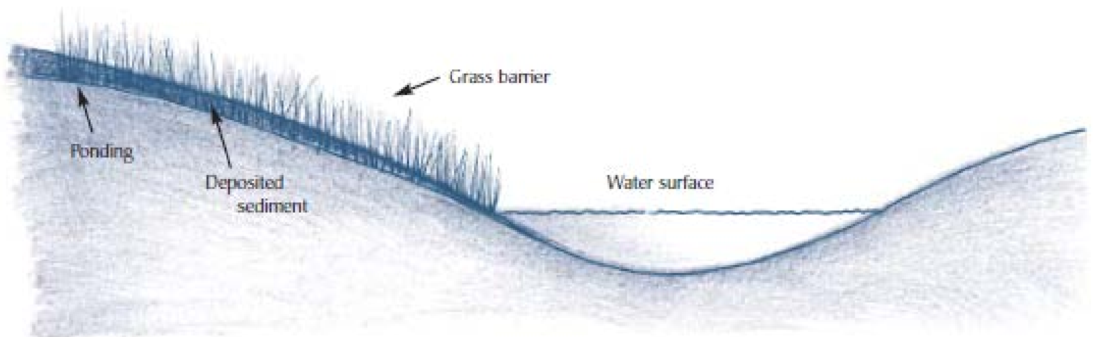


Figure 19: How a grass buffer strip functions to trap sediment.

Source: [175].

Studies have also been conducted to determine the effect of pesticide dosage on grass strips. For example, Popov and Cornish [254] 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 as the atrazine did not kill the grass. These results may alleviate potential concerns regarding the effect of pesticide runoff on the health of the actual buffers.

As plants vary widely in size, form, growth rate, longevity, and litter quality, their influences on stream water chemistry may range widely as well [82]. To date, there have been few comparative studies of vegetation types on the combined effect of vegetation influences on stream water chemistry. Scant research is also available on the response of stream water chemistry to the loss of riparian vegetation or its restoration [82]. The processes by which vegetation influence transfer of contaminants to streams include infiltration, transformation, uptake, volatilization (Figure 20).

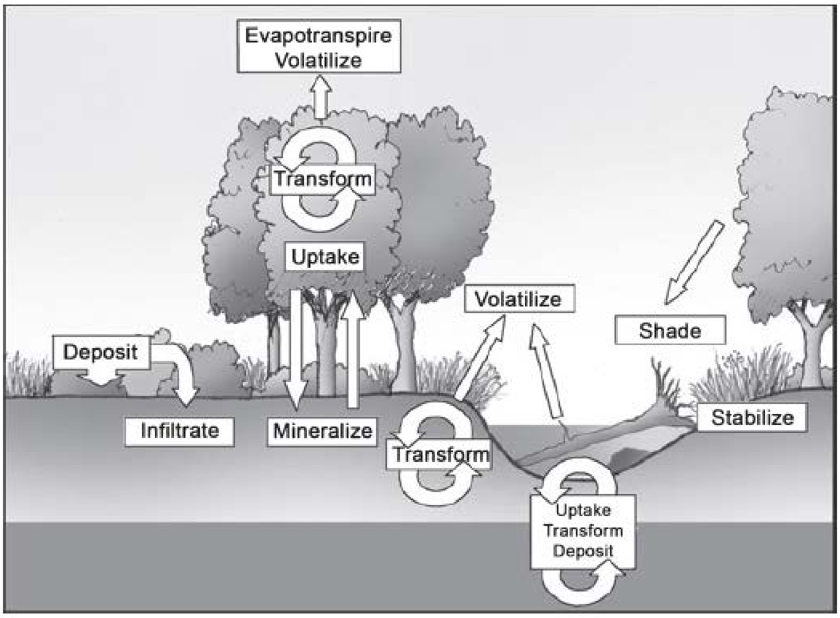


Figure 20: Stream water chemistry processes of riparian and channel systems.

Source: [82].

Mayer *et al*. [191] examined 45 studies of 89 different riparian zones including forest, herbaceous and mixed vegetation. Rates of nitrate removal from groundwater were not found to be related to vegetation type. In general, denser vegetation cover on the ground surface leads to higher buffer strip reduction efficiency [1, 311], provided that the soil water saturation is within certain limits. Yuan *et al*. [331] found that sediment trapping efficiency did not vary by vegetation type, and that grass buffers and forest buffers have roughly the same sediment trapping efficiency (Figure 21).

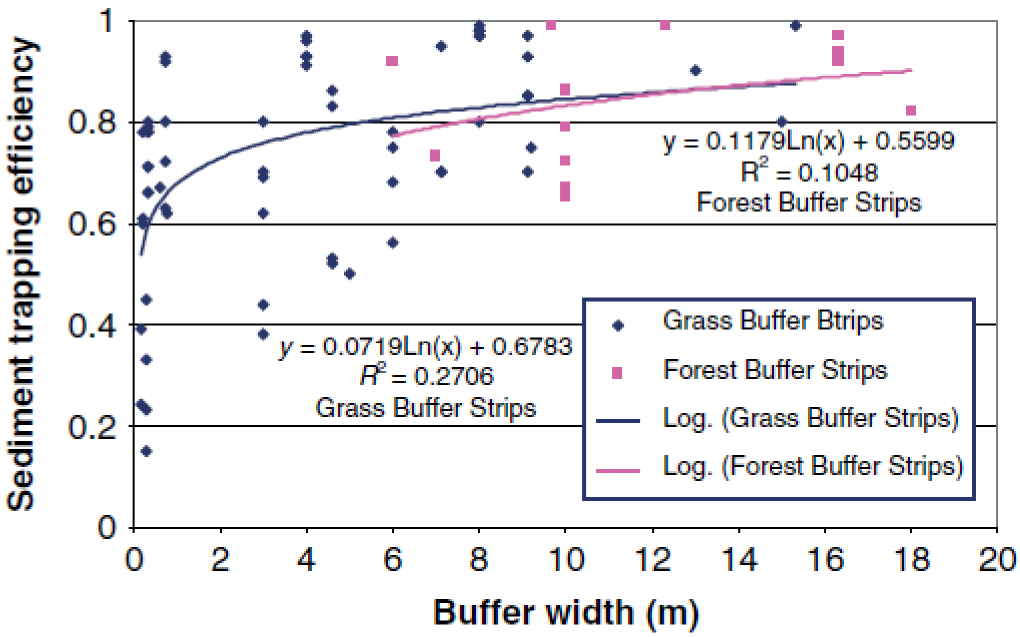


Figure 21: Vegetation type and sediment trapping efficiency of vegetated buffers.

Source: [331].

Some studies have shown that woody plants alone are not sufficient for non-point source pollution retention by buffers [280, 310]. Continuous vegetative ground cover may minimise the development of concentrated flow channels. Remnant forests with grass filters dispersed or buffered 100% of the concentrated flow channels for pollutants observed in a study by Knight *et al*. [155], whereas remnant forests without the adjacent grass filters dispersed or buffered 80%. Grass type in may be particularly critical, with dense, stiff grass the preferred vegetation based on its flow-retarding structure [82]. Taller grasses may also function better than short grasses under high runoff conditions because buffer performance is reduced if grasses become submerged by runoff [155].

The spatial distribution of plant shoots, roots, and plant litter within a riparian forest and adjacent stream channel defines the spatial dimensions of interaction between riparian vegetation and water. Above ground vegetation and surface litter interact directly with precipitation, surface runoff, and flood waters in riparian zones. Root systems interact with soil water and with groundwater that is shallow enough for roots to reach. Roots of many plants have the potential to reach several meters deep into the soil [49], but most roots occur in the upper 1 m of soil [139, 307].

Vegetation also affects the transport of chemicals by mediating water flow and distribution in riparian zones. Infiltration of precipitation and overland runoff transports dissolved and colloid-associated chemicals into the root zone where they can interact with soil minerals, living roots, soil organic matter, and microbes. Infiltration is improved by the presence of vegetation [14, 21]. Plant stems and litter at the ground surface create roughness that retards overland flow and increases concentration time for water to infiltrate the soil. Stems and plant litter at the soil surface also promote infiltration by providing roughness that slows overland flow and disperses it more widely across the riparian soil surface [82]. Infiltration of overland flow strongly promotes the deposition of sediments and sediment-bound agricultural chemicals carried in overland runoff. Infiltration reduces runoff volume and its physical capacity to carry sediment, so excess sediment deposits on the ground surface [121, 165]. The deposited sediments eventually become overgrown by vegetation and the associated chemicals become part of the root zone pool and subject to soil biogeochemical processes [290].

The extent and condition of wetland vegetation will determine its buffering capabilities and functionality which ranges from water temperature moderation, sediment removal, nutrient removal and, for wider buffers, species diversity Figure 22 [58]

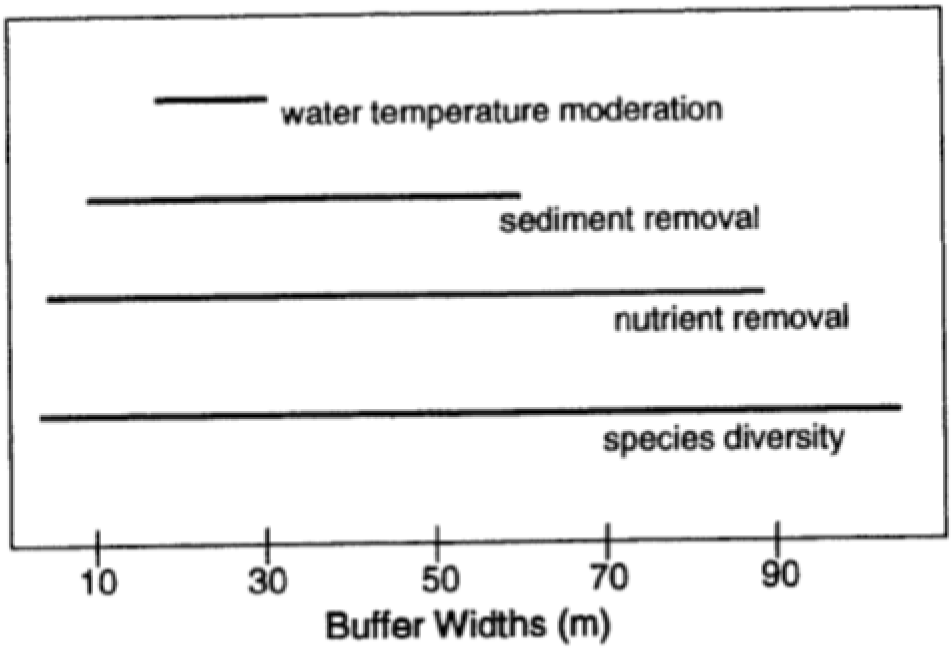


Figure 22: Functionality of vegetated buffers.

Source: [58].

A large buffer in good condition (dense native vegetation and undisturbed soils) is needed if adjacent wetlands and waterways are adjacent to intensive agriculture [58].

Under most conditions it was found that buffers less than 5-10m may retain natural physical and chemical characteristics of their environment however provide little protection of the biological component of aquatic resources; to do that buffers should be at least 15-30m wide [58, 60].

Natural wetlands, with a healthy aquatic plant community have been used to successfully remove phosphorus and nitrogen from runoff water (ref) with important characteristics being long residence times (low flow rates), water levels near or below the sediment surface as they enhance water/sediment interactions, optimise emergent plant growth and nutrient uptake by plants and sediment [60]. Chambers *et al* [60] found that natural (and artificial) wetlands have the potential to reduce phosphorus loads in agricultural runoff which they receive which the proportion depending on P concentration and flow rate of water through the system [60]. In temperate climates, such as Peel-Harvey catchment, Western Australia, natural wetlands were found to absorb P that they receive from agricultural runoff and confine this P within the boundaries of the wetland when outflows did not occur [60].A network of agricultural drains and wetlands have the potential to be effective nutrient removal systems.

Chambers *et al* [60] found that natural wetlands in the Peel-Harvey catchment, Western Australia absorbed phosphorus received from adjacent agricultural land and that this phosphorus remained in the wetland if there was no outflow. A mass balance indicated that the largest store of phosphorus was the surface sediments, not the wetland plants [60] and that 'the importance of the wetland vegetation was primarily its mediation in phosphorus pathways and transformations and, through the formation of litter, its contribution to the sediment' [60], p158.

One major difference between reclamation pits, natural and constructed wetlands is that constructed wetlands have a mix of purposefully planted vegetation for enhancing nutrient uptake, sediment deposition and biofilm surface area. Various types of vegetation have been used in constructed wetlands, ranging from native and exotic species. However, some native species turn weedy in constructed wetlands depending on the flow and drawdown regimes. Native Phragmites sp. is good at growing in fluctuating water levels and on vertical slopes and competes with paragrass and hymenachne; Phragmites is also rhizomous (i.e., its root system spreads underground), so it will regrow after slashing or dredging. Planting trees along the wetland edges help shade out weeds and though most native species will not persist in 1-2 m water depth, if they are planted, they will hold back the colonisation of weeds.

Vegetation plays multiple roles in nutrient mitigation and removal. Large aquatic plants manage the sediment surface oxygen layer, which is needed for both the phosphorus cycle (mineralization of P) and denitrification. Roots and shoots of aquatic plants uptake nutrients and pesticides from the sediment. Roots/rhizomes hold higher levels of phosphorus than stems and leaves, which tend to hold more nitrogen and carbon (Greenway and Woolley 1999); this uptake of nutrients from the wetland could be exploited through harvesting plant tissues. Greenway and Woolley (1999) found that harvesting emergent plant species could remove more nitrogen and phosphorus from wetlands than through other removal processes. Specifically, from analysing nutrient content of new shoot growth after harvesting, they suggested that in a 6 month period, all of the reactive phosphorus and half of nitrogen oxides in the studied wetland could be incorporated in plant biomass with a harvesting regime. Particular species may be better suited for nutrient uptake than others since species differ in their effectiveness of uptaking and assimilating P and N (Table 7; e.g., [86, 130]). For example, aquatic grasses (Gramineae), as compared to sedges, were better at uptaking nitrogen and carbon, and submerged and floating macrophytes (e.g., *Ceratophyllum*, duckweed, water lilies and ferns), as compared to emergent species, were better at uptaking phosphorus and nitrogen [107].

Plants also uptake and retain some herbicides, including atrazine and fluometuron. Using run-off simulations, Locke *et al*. [173] found that atrazine levels increased 5-fold in plant cells within 24 hrs and then remained steady throughout the study period and the highest concentrations of fluometuron were found in plant cells in the first hour. Moore *et al*. [216, 218, 219] have similarly shown that wetland plants (e.g., macrophytes) can be essential in efficient uptake and removal of various pesticides (e.g., pyrethroids (lambda-cyhalothrin, cyfluthrin), diazinon, and methyl parathion).

Vegetation provides surface area for microbes and forms the basis for the invertebrate community [162]; this periphyton community can provide effective mitigation for pesticides (e.g., atrazine [274] and fluometuron [273]) and nutrients. Periphyton, which describes the community of algae and microbes that live on submerged surfaces, has some of the highest denitrification potential as compared with soil, due to the biodegradable organic carbon produced by the [294]. There are also differences in denitrifying capacity between the periphyton found on the detritus in ponds dominated by different species of submerged plants. For example, detritus from ponds dominated by *Typha latifolia* and *Phragmites australis* had lower denitrifying capacity than those dominated by *Elodea canadensis* [16]. Biofilm mediation of pesticides (e.g., fluometuron [272]) was the dominant removal mechanism in vegetated pond experiments, where 40% of the initial load of fluometuron occurred in vegetated ponds, compared to 70% remaining in the open pond at the end of the study. Further, there is evidence of microbial adaptation, where there is enhanced degradation in wetlands with previous exposure to certain pollutants such as [332].

Studies	Nutrient removal efficiencies ($kg\,ha^{-1}\,day^{-1}$)		Types of wastewaters	Plant species used	Nutrient plant uptake in plant biomass
	N	P			
Pilot study	3.44	0.24	Nutrient solution	*Phragmites karka*	42.12% N; 28.92% P
	1.56	0.23	Nutrient solution	*Lepironia articulata*	17.43% N; 26.08% P
Headley (2004)			Nursery runoff	*Phragmites australis*	41–54% N; 36–63% P in above-ground biomass, 24–30% N; 36–39% P in below-ground biomass
Toet (2003)	0.53	0.082	Sewage effluent	*Phragmites australis*	37–42% N; 22–40% P
Browning and Greenway (2003)	0.8–7.3			*Baumea articulata, Carex fascicularis, Philydrum lanuginosum and Schoenoplectus mucronatus*	11% N; 3% P
Greenway and Woolley (2001)	0.72–1.93	0.22–0.68	Secondary effluent	*Typha domingensis, Schoenoplectus validus, Eleochrais equisetina, Eleocharis sphacelata*	14.5–80% N, 24–80% P
Kantawanichkul et al. (2001)	11.2		Livestock effluent	*Cyperus flabelliformis*	7–9% N
					0.414–0.491 $g\,N\,m^{-2}\,day^{-1}$ in above-ground biomass 0.197 $g\,N\,m^{-2}\,day^{-1}$ in below-ground biomass
Lim et al. (2001)	4.5		Septic tank effluent	Cattail *Typha sp.*	2.6 $kg\,ha^{-1}\,day^{-1}$ (50% N)
Tanner (2001)	3.0	1.0	Dairy farm wastewaters	Soft-stem bulrush *Schoenoplectus tabernaemontani*	
Okurut (2000)	7.1	0.24	Septic tank effluent	*Cyperus papyrus*	14.95–21.91 $mg\,g^{-1}$ N; 5.61–5.95 $mg\,g^{-1}$ TP in above-ground tissue,
Okurut (2000)	10.4	0.26	Septic tank effluent	*Phragmites australis*	19.96–22.16 $mg\,g^{-1}$ TN; 7.90–10.05 $mg\,g^{-1}$ TP in below-ground biomass
Koottatep and Polprasert (1997)	3.0		Septic tank effluent	Cattail *Typha angustifolia*	43% TN (31% in leaf, 10% in stem, 2% in root)
	1.08	0.229	Sewage effluent	*Typha latifolia*	
Greenway (1997)	3.68	1.997			64.8% N
Tanner et al. (1995)	1.5–14	1.3–3.2	Dairy farm wastewaters	Soft-stem bulrush *Schoenoplectus tabernaemontani*	

Table 7. Nutrient removal efficiencies of plant species used in constructed wetlands.

Source: [293]

Though there can be vegetation in sumps, the purpose of these pits is to hold water and sediments for re-use on paddocks, however, similar to vegetated systems discussed elsewhere in this document, the presence of vegetation may assist uptake, and provide biofilm surface area for bioremediation of, nutrients and pesticides within the sump water.

The presence of vegetation can exert a strong influence on the downstream movement of agricultural run-off [214]. Vegetation influences settlement of suspended solids by creating a trapping mechanism and increases trapping by reducing flow velocity. Vegetation reduces pesticide movement to streams by reducing runoff volumes through infiltration in soil, through contact between dissolved phase pesticide with soil under the vegetation and/or by reducing flow velocities to the point where eroded sediment particles, with sorbed pesticide, can settle out of the water [276]. The degree of flow retardation depends on the type of vegetation (e.g. sugarcane, banana, grass, riparian vegetation). Karim *et al*. [146] showed that sugarcane offers more resistance to flow comparing with Banana field. The mechanism of pesticide trapping has historically been assumed to be a function of the organic carbon sorption coefficient. Most researchers agree that vegetation trap highly sorbing pesticides in the same manner that they trap sediment. However, pesticide attached to eroded sediment becomes the dominant transport mechanism only for strongly sorbing pesticides [267]. 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 in the filter strip. Leonard [167] suggests that for most pesticides with relatively low sorption capacity, the most important transport mechanism is runoff as opposed to eroded sediment due to the difference in magnitude between runoff volume and sediment yield as found in most floodplain [276]. Corenblit *et al*. [65] showed there are inter-relationships between vegetation and sediment scour or deposition on sandbars that affect the time scale and progression of channel evolution, particularly the later stages of evolution tending toward channel equilibrium. Vegetated systems are also effective in removing methyl parathion from agricultural runoff [217].

5.6 Influence of organic matter

Organic matter (e.g., straw mulch, cane mulch, leaf litter, manure) is another major trapping mechanism for pesticides [83, 145, 239, 246, 255, 272, 276, 286, 287, 291]. Organic matter can actively trap nutrients and pesticides through chemical adsorption (i.e., cation exchange capacities). When pollutants are adsorbed to organic matter, there is less likelihood that the pollutants will run-off with surface water or leach into groundwater, and most pesticides begin to break down once they are adsorbed. As an example, sugarcane mulch, or surface crop residue, can reduce runoff losses of atrazine and metribuzin [286, 287].

The presence of organic matter in riparian forests can improve the interception of contaminants from agricultural runoff. In the superficial layer of the riparian forest, the litter and relatively high concentration of organic matter of the soil may adsorb some pesticides such as atrazine, thus reducing them in the runoff [59, 190]. Thick layers of organic matter litter enhance sorption of pesticides, thus retarding their transfer to aquatic systems, decreased leaching, decreasing concentration peaks in runoff, and likely increased degradation of the pesticides [243]. Adsorption of pesticides is particularly effective in saturated soils [238].

The presence of organic matter also drives many biogeochemical processes in riparian forests and supports microbial communities that can further contribute to the processing and transformation of contaminants [243]. Evidence suggests that rates of denitrification are directly proportional to organic matter content in stream side aquifers [275]. Buried organic matter can also profoundly affect groundwater quality below the root zone [79].

In general, the increasing age and density of riparian forest contributes to increased litter layers and organic matter content of soils [82]. It is expected that increasing maturity and complexity of riparian forest would promote a greater diversity of microbial functional groups, and therefore increased biogeochemical activity for processing and transformation of different contaminants. In comparison, grassy areas accumulate less organic matter and generally have lower rates of biogeochemical activity.

The sediment of most undisturbed wetlands has a high accumulation of organic matter if they experience inundation, even those inundated for short periods (<6 days per year). Bryant *et al*. [44] and Chambers *et al*. [60] considered wetland vegetation to be involved in phosphorus pathways and transformations and importantly the formation of plant litter which was then contributed to the sediment and important for microbial pathways.

Chambers *et al*. [60] found that the sediment in a natural wetland, in an undisturbed area, had a high organic matter component (86-95%) which was a fibrous peat derived from the sedge canopy, whereas the organic matter component was only 4-41% in wetlands within agricultural lands perhaps due to soil disturbance in the adjacent catchment. It was also found that the phosphorus concentration in open water within a wetland was highest where the ratio of vegetation to water was lowest, that is when there was less surrounding vegetation and that in uncultivated land the phosphorus was in sediment rather than plants whereas in disturbed farmland this was reversed, with more phosphorus being in the vegetation than in the sediment.

As well as plant or microbial uptake (assimilation) or respiratory denitrification by bacteria there are alternative, microbially mediated processes of nitrate transformation including dissimilatory (the reduction of nitrogen into other inorganic compounds, coupled to energy producing processes) reduction of nitrate to ammonium (DNRA), chemoautotrophic denitrification via sulphur or iron oxidation, and anaerobic ammonium oxidation (anammox) as well as abiotic nitrate removal processes [47]. These alternative pathways are of particular importance for the management of excess N in the environment, especially in cases where nitrate is transformed to ammonium, a biologically available and less mobile N form, rather than to dinitrogen gas. [47].

Methane is produced as a result of organic matter decomposition in reducing environments such as wetlands and is thought to account for approximately 25 percent of the global methane flux (US Climate Change Programme 2006).

Organic matter accumulates in wetlands as a result of primary production and decomposition. The quantity of organic matter and overall denitrifying potential of wetland soils can vary depending on vegetation types present in the wetlands. For example, [129] found that emergent macrophyte zones were higher in organic matter and denitrifying potential than open water and forest edge zones. Organic matter found in wetlands is effective in adsorbing pollutants and helps prevent leaching of these pollutants into groundwater [69, 141]. In paired trials, the increased organic matter found in wetland soils, as compared to paddock soils, was found to adsorb more pollutants (e.g., fluometuron [273, 289]). Increased adsorption can lead to an increase in degradation of these pollutants through retention (i.e., time for decay) and bacterial decomposition. However, organic soils may also act as slow release agents, depending on the chemical retained, if the higher acidity of the organic matter protects pH-sensitive pollutants from degradation [330].

Organic matter is important for denitrification processes, in terms of providing organic carbon, and there is a positive relationship between the two in the sediment surface layer [129]. As an example, denitrification accounted for up to 30% of organic matter oxidation in one wetland study conducted in the USA [284]. Due to this significant contribution of organic matter to denitrification, it is important to note that constructed wetlands on agricultural properties tend to have less organic matter than constructed wetlands used for urban wastewater treatment [333].

Organic matter can actively trap nutrients and pesticides through chemical adsorption. When pollutants are adsorbed to organic matter, there is less likelihood that the pollutants will run-off with surface water or leach into groundwater, and most pesticides begin to break down once they are adsorbed.

Selim *et al*., [286] evaluated the effectiveness of sugarcane residue (mulch cover) in reducing nonpoint-source contamination of applied herbicides from sugarcane fields. They found significant amounts of applied herbicides were intercepted by the mulch residue. Extractable atrazine concentrations were at least one order of magnitude higher for the mulch residue compared with that retained by the soil.

5.7 Trapping efficiencies

Grass buffers may reduce nitrogen levels from agricultural runoff. For example, scientists in the Piedmont region of North Carolina found that both grass and grass/forest riparian buffers reduced total nitrogen by 50 percent [68]. On experimental plots at Blacksburg, Virginia, orchard grass buffers 30 feet wide reduced total nitrogen by 76 percent [80]. However, scientists in England reported that although both grass and forested buffers can effectively remove nitrogen, forested buffers may be more efficient [120]. They found that a buffer of poplar trees 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, but on an annual basis, grass was less effective than the forest [237]. 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 [282]. 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 [299]. In this study, soil texture, organic matter content, and groundwater flow paths were reported to be the most important factors influencing the fate of nitrogen.

Data from a range of studies included in Appendix 2 leads to a number of broad conclusions regarding the trapping efficiency of sediments, nutrients and pesticides in grass 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 [252]; and
    - 47-100 % reduction in nitrate [245].
- Table 5 includes results from studies showing the pesticide trapping efficiencies for a range of pesticides. Highly adsorbed pesticides were trapped at rates of 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 [251]. Results in Boutron *et al*. [216] 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*. [280] 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*. [267] is an excellent review paper on the efficiencies of different trapping mechanisms (e.g., 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.

Further work assessing trapping efficiency (see [199]) 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 convergent slopes could be effective at trapping nutrients and sediment in the extreme rainfall 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 buffer performance in these conditions including exfiltration, flow channelisation, scour and low vegetation density. 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. The outcomes of this work are also supported by the information reported by Karssies and Prosser [149] in Table 6.

Studies have indicated that 'near-stream' vegetation can retain over 75% of the nitrate, 65% of total nitrogen and 30% of total phosphorus contributed by soil solution draining from surrounding agricultural land [22, 109]. A reforested riparian buffer in the eastern United States, removed 26% of the subsurface nitrate flux and 43% of the suspended sediment concentration delivered from upslope, but not total phosphorus [230]. Mayer *et al*. [191] reviewed the effectiveness of forested riparian zones at removing nitrate from subsurface waters, and found great variation (Table 8).

Riparian zone width (m)	NO_3^- concentration (mg L^{-1})		NO_3^- removal effectiveness (%)	Source
	Influent	Effluent		
50	26	11	58	Hefting *et al.* (2003)
200	11	4	64	Spruill (2004)
10	6.29	1.15	82	Schoonover and Williard (2003)
14	0.02	0.02	0	Sabater *et al.* (2003)
30	0.02	0.01	50	Sabater *et al.* (2003)
50	0.49	0.76	−55	Sabater *et al.* (2003)
15	28.64	35.84	−25	Sabater *et al.* (2003)
20	1.14	0.7	39	Sabater *et al.* (2003)
20	0.12	0.43	−258	Sabater *et al.* (2003)
15	3.23	0.72	78	Sabater *et al.* (2003)
20	6.4	1.44	78	Sabater *et al.* (2003)
55	–	–	83	Lowrance *et al.* (1984)
85	7.08	0.43	94	Peterjohn and Correll (1984)
204	29.4	1.76	94	Vidon and Hill (2004b)
50	13.52	0.81	94	Lowrance (1992)
60	8	0.4	95	Jordan *et al.* (1993)
16	16.5	0.75	95	Osborne and Kovacic (1993)
16	6.6	0.3	95	Haycock and Pinay (1993)
15	–	–	96	Hubbard and Sheridan (1989)
165	30.8	1	97	Hill *et al.* (2000)
50	6.26	0.15	98	Hefting and de Klein (1998)
220	10.8	0.22	98	Vidon and Hill (2004b)
50	7.45	0.1	99	Jacobs and Gilliam (1985)
10	13	0.1	99	Cey *et al.* (1999)
100	5.6	0.02	100	Spruill (2004)
30	1.32	nd	100	Pinay and Decamps (1988)
100	12	nd	100	Spruill (2004)
60	–	–	27	Groffman *et al.* (1996)

Table 8: Effectiveness of forested riparian zones at removing nitrate from subsurface water.

Source: [191].

The sediment trapping efficiency in riparian buffers depends primarily on buffer width, vegetation type, density and spacing, sediment particle size, slope gradient and length, and flow convergence. Other factors also affect sediment trapping efficiency include soil properties, initial soil water content, and rainfall characteristics (total amount and intensity) [331]. For example, Brunet *et al.* [43] estimated that the floodplain and riparian zone of a 25km reach of a seventh-order river retained 10 to 20% of the suspended sediment and particulate N load carried into that reach during two floods. Even though the riparian zone occupied only 6 to 7% of the floodplain, the riparian zone was responsible for the majority of retention.

Performance of constructed wetlands in removing pollutants is influenced by area, length to width ratio, water depth, rate of contaminant loading and retention time. The overall size of the wetland needs to be large enough to insure enough residence time to allow wetland processes to operate (Raisin *et al.* 1997). Suspended sediments are most easily trapped, especially the coarser fractions and especially if particular vegetation types are present. Pollutants attached to particulate matter are not as easily trapped as those bound to suspended sediments since most sediment-bound pollutants are usually attached to finer particles which are more difficult to retain. [94] suggested that a wetland, designed for both nitrogen and phosphorus retention, needs a mud-flat area near the inflow, which can seasonally flood and then dry out, and a more open water area near the outflow of the wetland to recapture any liberated phosphorus derived from seasonal drying of the mud-flat sediments.

Wetlands can be effective in reducing concentrations of pesticides as a result of retention time, sedimentation, adsorption onto organic matter/organic carbon, and plant uptake, with reductions of 33–51% in diuron and 20–60% in simazine [239, 267, 272, 274]. One study found removal/trapping efficiencies of organic material and suspended sediments were 80%, while nutrients were less than 60% [292].

REGIONALLY SPECIFIC CASES FROM PREVIOUS STUDIES

6.1 Mulgrave River banana farm

An Australian Research Council Linkage project held by the (then) Department of Environment and Resources Management (DERM) and James Cook University (JCU) investigated the transport of nitrate, resulting from fertiliser application, into Behana Creek, in the Mulgrave River catchment of the Wet Tropics of Far North Queensland. The project considered how fertilisers leach below the root zone and enter groundwater in sugarcane farming areas, focussing on transportation of nitrate through the aquifer and natural attenuation of the contamination by denitrification in riparian forest buffers.

Behana Creek (Figure 23) in the Mulgrave River catchment is located near Aloomba, about 22 km southeast of Cairns, Queensland, Australia. More than 7% of the Behana Creek catchment is made up of protected rainforest, though this is not equally distributed. Whilst the headwaters are completely covered by rainforest, the lower reaches, used for agriculture (primarily sugarcane cultivation), lack riparian zones, such that agricultural runoff can reach the creeks and rivers without passing through a buffer zone in these areas.

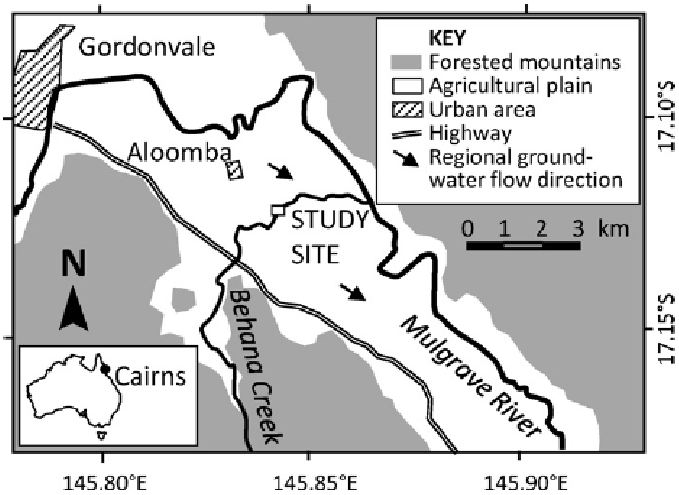


Figure 23: Location of Behana Creek sampling site in the Mulgrave River catchment.

Connor *et al*. [64] considered removal of nitrate in groundwater as it enters a forested riparian zone and was transported towards Behana Creek. The hydrology of the riparian zone was characterised using measurements of soil water content and water table depth (13 piezometers).

Groundwater (presumably sourced from surrounding sugarcane fields) entering the riparian zone of Behana Creek was found to have low concentrations of nitrate (mean <0.03 mg NO3-N L^{-1} over both wet and dry seasons), however, concentrations increased (by up to 50 fold) as groundwater progressed through the riparian zone, suggesting the riparian zone was a potential source of nitrate to the adjacent creek. The addition of nitrate was attributed to nitrification in riparian surface soils, driven by large net primary productivity, including large amounts of litterfall (12.19 Mg ha-1 y^{-1}). Nitrate generated in riparian soil was subsequently leached into groundwater in the wet season during rainfall events.

Nitrate was also derived from nitrification in groundwater and, potentially, from the mixing of deeper groundwater of higher nitrate concentrations. Connor *et al*. (2012) demonstrated that groundwater leaving sugarcane fields does not always have high concentrations of nitrate, and that these concentrations are not necessarily reduced during the passage of groundwater through riparian forest buffers. Nitrate generated within the riparian zone, on the other hand, was leached into groundwater during heavy rain events and became a potential source of nitrate to Behana Creek. Connor *et al*. (2012) concluded the riparian forest buffer was not demonstrated to show water quality benefits with respect to nitrate reduction at their study site.

Similarly, [152] found low concentrations and fluxes of nitrate in groundwater seepage from sugarcane areas adjacent to Behana Creek, and poor correlation with riparian forest width. The results suggested that any possible effects of riparian forest on nitrate in groundwater in this environment were out-weighed by hydrogeological factors, in particular preferential flow paths for nitrate movement.

A study by [125] showed that nitrate distribution and mineralisation activity varied spatially in the riparian zone of Behana Creek and were dependent on topography. The effect of topography was subsequently due mainly to differences in flooding frequency, as lower-lying locations flooded more often and thus received more nutrients and retain water, favouring denitrification.

Soil carbon was a strong determinant of microbial activity for nitrogen mineralisation and denitrification in the riparian buffer. In both of these microbial processes carbon appears to be the main limiting factor for microbial activity with soil depth, hence microbial activity was limited to topsoils of the riparian zone at Behana Creek.

REGIONALLY SPECIFIC CASES WITHIN CURRENT PROJECT

Experimental sites (Figure 24) were selected in the Johnstone River, lower Burdekin/ Haughton Rivers and Herbert River catchments. Field sampling was conducted in the Burdekin on 23 November 2012, 28 March 2014 and 12 August 2013. Sampling was conducted in South Johnstone and Herbert regions on 18 December 2012 and 27 March 2013. The first wetland visited in the Herbert (18 December 2012) turned out to be highly acidic (2.8 pH), due to acid sulphate soils, and so this data was discarded from this study and a new site was found and sampled on 27 March 2013. The Johnstone and Herbert region wetlands have not been revisited due to the absence of a rainfall event that caused overflow of the wetlands to be sampled.

Though a suite of water soluble nutrients were analysed only data for those nutrients which are bioavailable are discussed here (see Appendices 1 & 2 for complete data set). This includes filterable reactive phosphorus (FRP), which is a measure of available phosphorus (e.g., from phosphate fertilisers) and dissolved inorganic nitrogen (DIN), which is a measure of available nitrogen (e.g., from nitrate fertilisers). Total suspended solids (TSS) are also shown as this measure depicts the amount of turbidity in the water column and suspended particles that are vectors for particle-bound pollutants and can also smother downstream vegetation and organisms. Dissolved organic nitrogen (DON), which is not strictly bioavailable, is derived from organic production (e.g., aquatic vegetation, leaf litter decomposition), and is usually found as an export of wetland production. Overall, particulate-bound pollutants (e.g., nitrogen and phosphorus and some pesticides) may be settling in the wetlands, whereas dissolved pollutants, at times, decrease throughout the wetlands, depending on the irrigation and rainfall patterns.

In the Burdekin region, irrigation and fertilising schedules dictate when the wetland will receive an influx of nutrients and pesticides. The sampling sites in the Burdekin (constructed wetland and reclamation sump) are both designed for tailwater re-use, meaning that the levels of pollutants may be heightened by the repeated use (and addition of fertilisers/pesticides) of the water coming from off the paddock. This aspect of increasing pollutant loads in recycled water needs to be addressed in terms of safe nutrient and pesticide levels for both paddock and wetland health, but also in terms of reducing new applications given the amount already in the recycled water supply.

The South Johnstone constructed wetland can be effective in reducing the TSS and nutrient load exiting the wetland during the dry season, however, during higher flow events as experienced during the wet season, there was minimal reduction, if any, to higher levels of nutrients entering into or being remobilised within the system.

In the Herbert constructed wetland, the homogeneous shallow depth profile may be inhibiting mid-wetland settlement of particulate-bound pollutants. However, only one sampling event has been conducted and analysed for this wetland, so it is difficult to draw conclusions as to the effectiveness of this wetland.

Experimental Design

Experimental sites (Figure 24) were selected in the Johnstone (Figure 44), lower Burdekin (Figure 26 and Figure 29), and Herbert (Figure 40) basins. Field sampling was conducted in the Burdekin on 23 November 2012, 28 March and 12 August 2013. Sampling was conducted in South Johnstone and Herbert regions on 18 December 2012 and 27 March 2013. The first wetland visited in the Herbert (18 December) turned out to be highly acidic (2.8 pH) most likely due to acid sulphate soils, and so this data was discarded from this study and a new site was found and sampled on 27 March 2013.

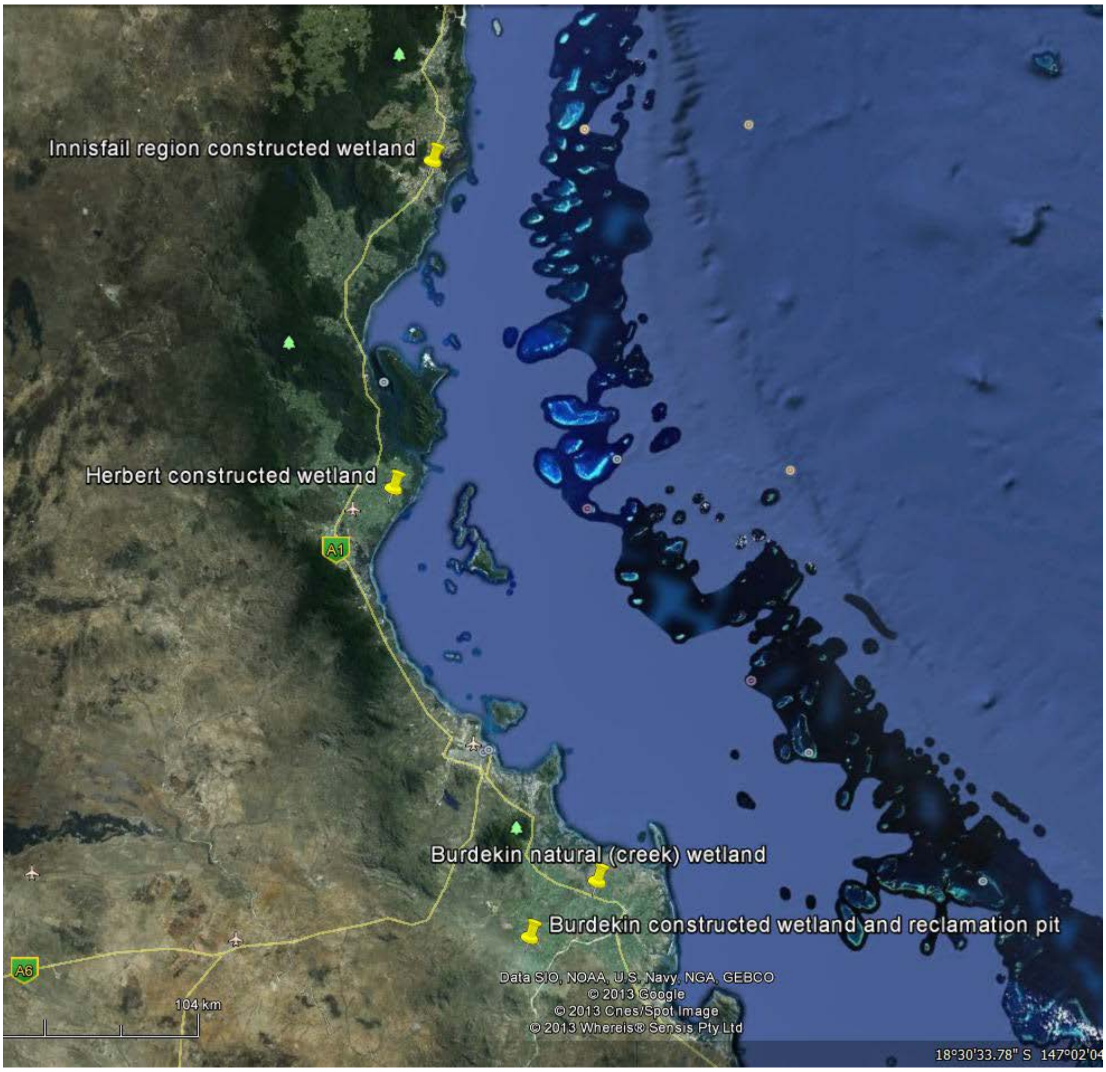


Figure 24: Location of study sites.

7.1 Lower Burdekin River catchment sugarcane farms

In the lower Burdekin River catchment areas, in the dry tropics region of North Queensland three sites were studied and each were sampled three times; November 2012, March 2013, and August 2013. Average annual rainfall for the Burdekin catchment ranges between 600 – 2500 mm/yr. The three sites included one constructed wetland, one natural 'creek' wetland and one reclamation sump.

The natural 'creek' wetland (> 700m long, ~10m wide, 2.1m deep) runs between two different farming enterprises (Figure 25). The adjoining paddocks have sodic soils that require lime application and irrigation is achieved by furrow irrigation. The creek drains the particular farm's property and the paddock furrows of the neighbours adjoining property back up onto the berm of the creek. The creek has been planted with riparian vegetation (NQ Dry Tropics project); *Typha spp.* grow at the edges and other plant species are present. Tarpon are often seen as well as cormorants, various duck species, turtles and crocodiles. This site was sampled at the inlet point, midpoint and outlet point (Figure 26).

Figure 25: Natural 'creek' wetland in the Burdekin region.

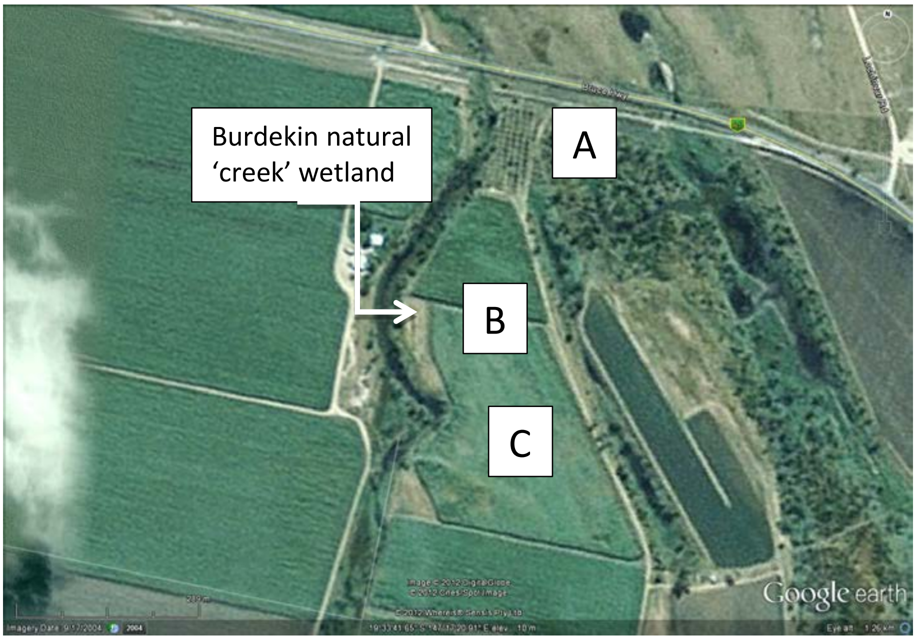


Figure 26: Sampling sites within the Burdekin natural 'creek' wetland.

A - Inlet, B - Midpoint, C - Outlet.

The constructed wetland (2.3 hectares; 340m long, 60m wide, 1.5m deep) (Figure 27) and reclamation sump (600m long, 15m wide, ~2m deep) (Figure 28) are situated on one farming enterprise however drain two different catchments. Water collected in these water bodies is re-used, an outlet pump enables irrigation of different paddocks in the area. This site was sampled at the inlet point, midpoint and outlet point (Figure 29).

Figure 27: Constructed wetland in the Burdekin region.

Figure 28: Reclamation sump in the Burdekin region.

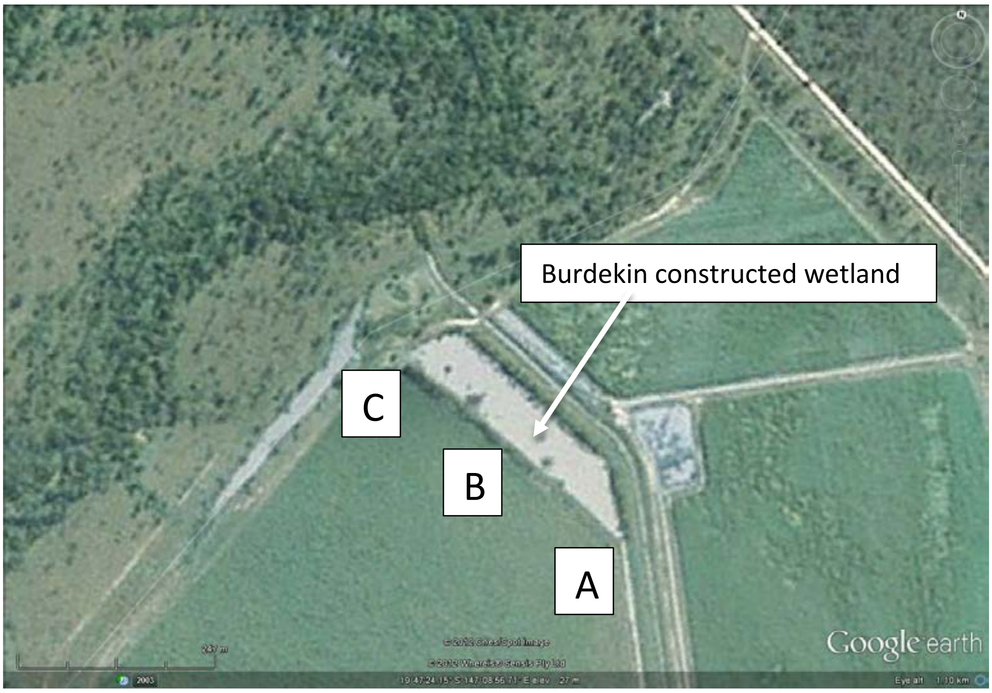


Figure 29: Sampling sites within the Burdekin constructed wetland.

A - Inlet, B - Midpoint, C - Outlet.

The vegetation of these wetlands and reclamation sump consist of a mixture of native aquatic and invasive species. The 'inlet' of the natural wetland is choked by *Typha spp.*, and the midpoint usually has a greater than 50% cover of lilies, duckweed, *Ceratophyllia* species and filamentous algae. The banks of the creek near this midpoint consist of grasses and sugarcane. The constructed wetland contains lilies, *Hydrilla*, *Ceratophyllum*, and filamentous algae, with some *Typha* and planted native tree species located on the edges. The sump has some paragrass and hymenachne (weed species) along the edges, as well as a few submerged macrophytes, and is surrounded by road and sugarcane. Various duck species, cormorants, and pelicans, as well as small fish species (e.g., rainbow fish) are often seen at the wetland.

Findings – Water analysis

The total suspended solids (TSS) tended to decrease from the inlet to the outlet of the three sampling sites; natural 'creek' wetland, constructed wetland and reclamation sump (Figure 30). In August 2013, however, TSS increased toward from the inlet to the outlet of the sump and constructed wetland.

Dissolved inorganic nitrogen (DIN) was at relatively low levels in the natural 'creek' wetland however it did peak at the midpoint sample, in deeper water (Figure 31) which may be due to input from adjacent paddocks in overbank flows. The constructed wetland and reclamation sump had higher concentrations of DIN than the natural wetland, and overall, did show a decrease in concentration in the area of the outlet/midpoint as compared to the inlet sampling point.

Dissolved inorganic phosphorus (=Filterable reactive Phosphous; FRP) concentrations were relatively low (less than 11 µg/L across all sampling periods) in the constructed wetland and sump, but did show a decrease from the inlet to the midpoint of the sump (Figure 32). In the natural wetland phosphorus concentrations were high (peak of 335 µg/L in August 2012) which may be due to phosphorous fertiliser application to adjacent cane paddocks, and concentrations decreased from the inlet to the outlet sampling points. There was one exception to this in March 2013 when phosphorus concentrations showed a reverse relationship; increased from the inlet to the outlet).

Findings - Sediment analysis

Concentations of total nitrogen in sediments indicated a general increase in nitrogen in the sediments from the inlet to the midpoint or outlet (Figure 33). However, in November 2012, a reverse trend was found, where more nitrogen was found in the inlet sediments than the midpoint and outlet sediments.

Total phosphorus also increased in the sediments, across all study sites (Figure 34) from the inlet to the outlet point with the exception of in the reclamation sump in August 2013 when there was a decrease in phosphorus from the inlet to the midpoint. An overall increase in phosphorous in the sediment, over the year, was expected (due to sequestration of phosphorus in the sediments) but was not the case. The trends found could indicate point to plant uptake or remobilisation of phosphorus in the system.

Findings - Pesticide analysis of water

In November 2012, in the natural 'creek' wetland – diuron and atrazine were the prominent pesticides found, both at levels above the Water Quality Guidelines for the GBR Marine Park [100] (0.9 µg/L and 0.6 µg/L, respectively; 0.7 µg/L atrazine ANZECC freshwater guideline)(Figure 35). Metribuzin and atrazine were detected in the constructed wetland, with the inlet concentration of metribuzin to be extreme at 5.27 µg/L. Concentrations of dissolved pesticides in March 2013 were much less than in November 2012 (0.28 µg/L maximum compared to 5.27 µg/L) (Figure 36) and were most conspicuous in the natural wetland. Both diuron and fluometuron decreased across the wetland samples, but atrazine increased toward the outlet. In August 2013, only atrazine was detected in the samples, and as in March 2013 for the natural wetland, atrazine concentrations increased across the wetland (Figure 37). In the constructed wetland and sump, where the adjacent canefarmer had switched from atrazine to metribuzin, only low levels (less than 0.05 µg/L) were detected.

Regarding analysis for only atrazine across all sampling events (Figure 38), there was a peak in detection in November 2012 at all samping sites, although highest in the natural wetland, where it increased across the wetland.

Findings - Pesticide analysis of sediment

Pesticide analysis of the sediments was only collected in March and August 2013, and only the data from March 2013 has been analysed. Ametryn, atrazine, diuron and metribuzin were all detected in these samples (Figure 39). Diuron and it's breakdown product has built up in the sediments across the wetlands, as well as ametryn and atrazine to lesser degrees.

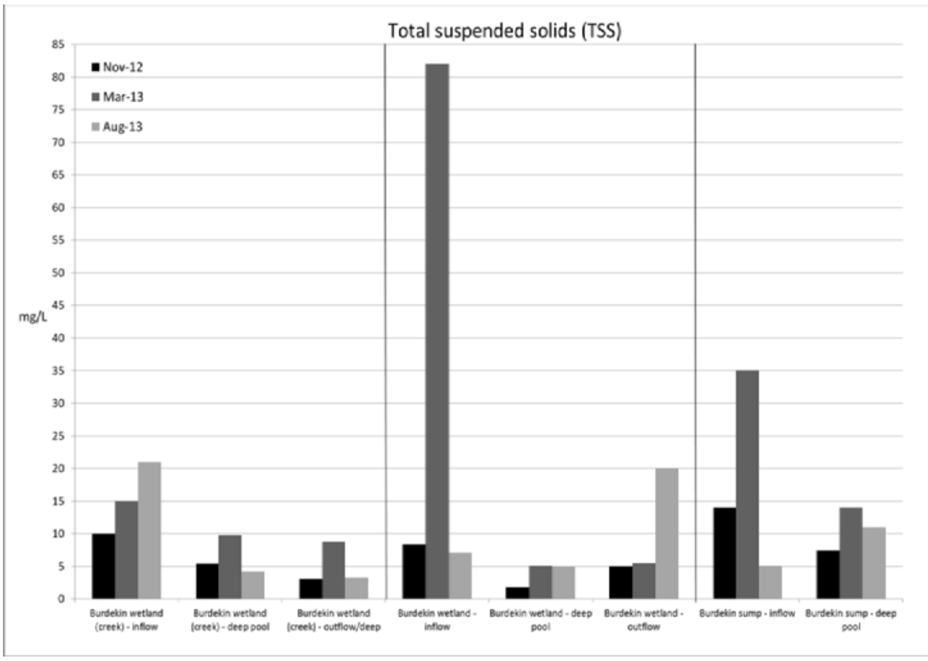


Figure 30: Total suspended solids of water samples collected from Burdekin sites.

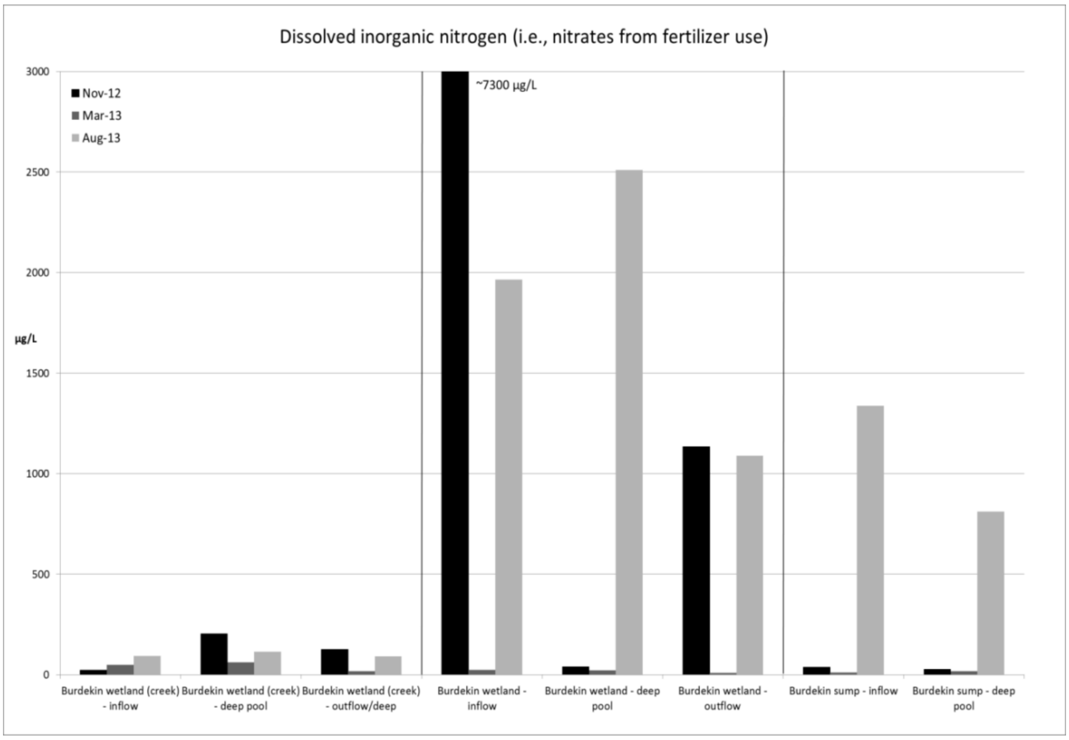


Figure 31: Dissolved inorganic nitrogen of water samples collected from Burdekin sites.

Note the off-scale measurements labeled on the graph.

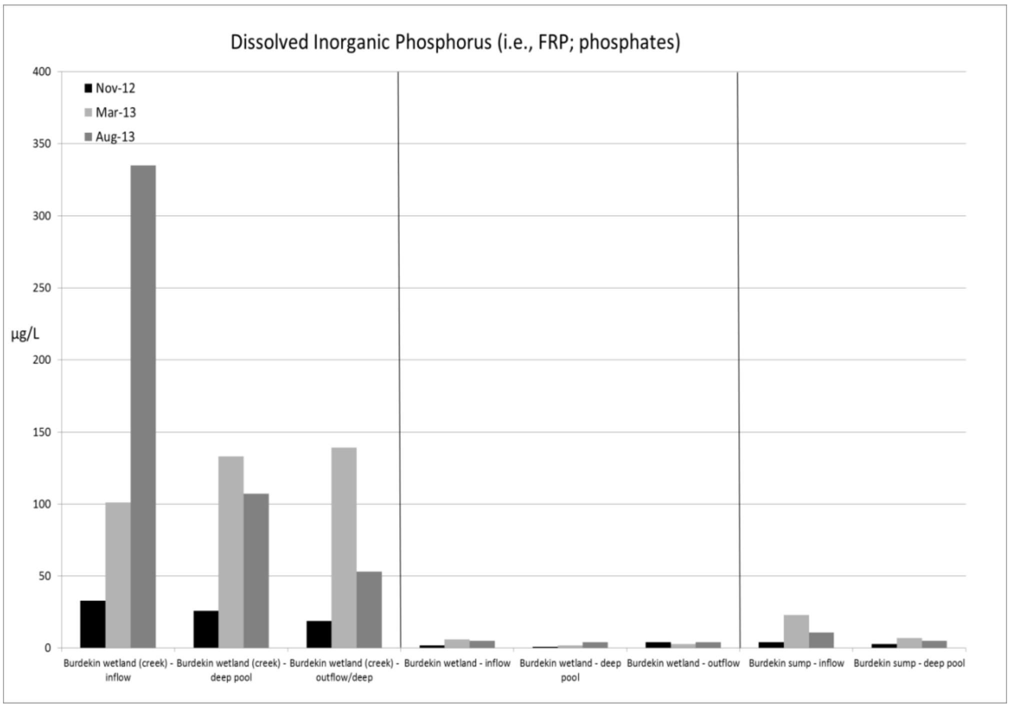


Figure 32: Dissolved inorganic phosphorus of water samples collected from Burdekin sites.

(DIP=FRP (filterable reactive P)=~phosphates).

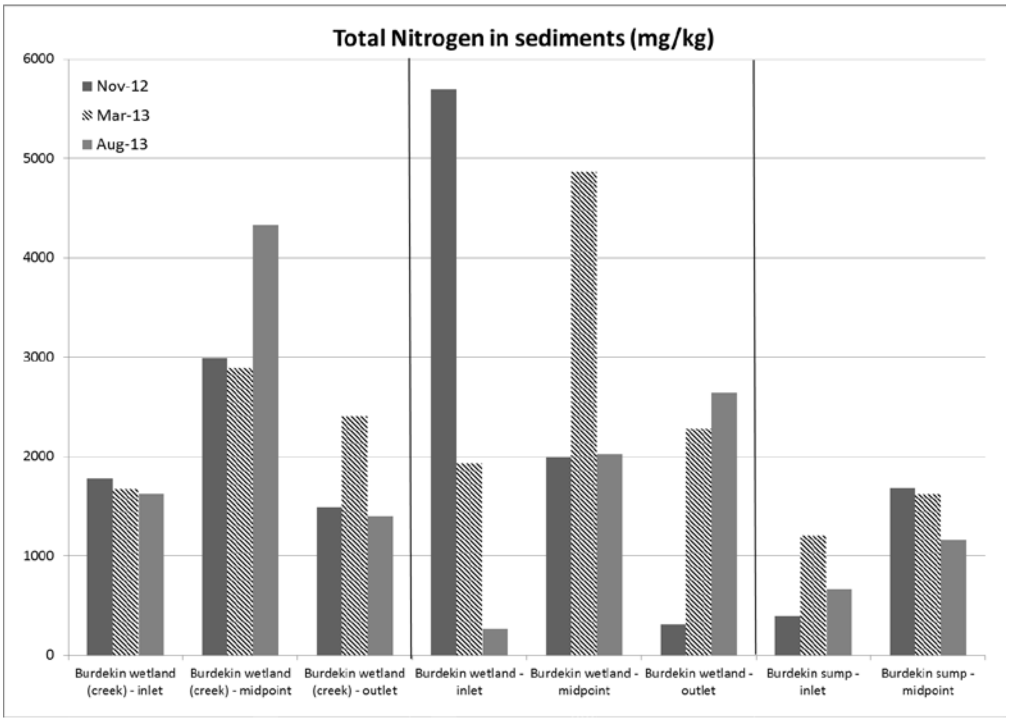


Figure 33: Total nitrogen in sediments of water samples collected from Burdekin sites.

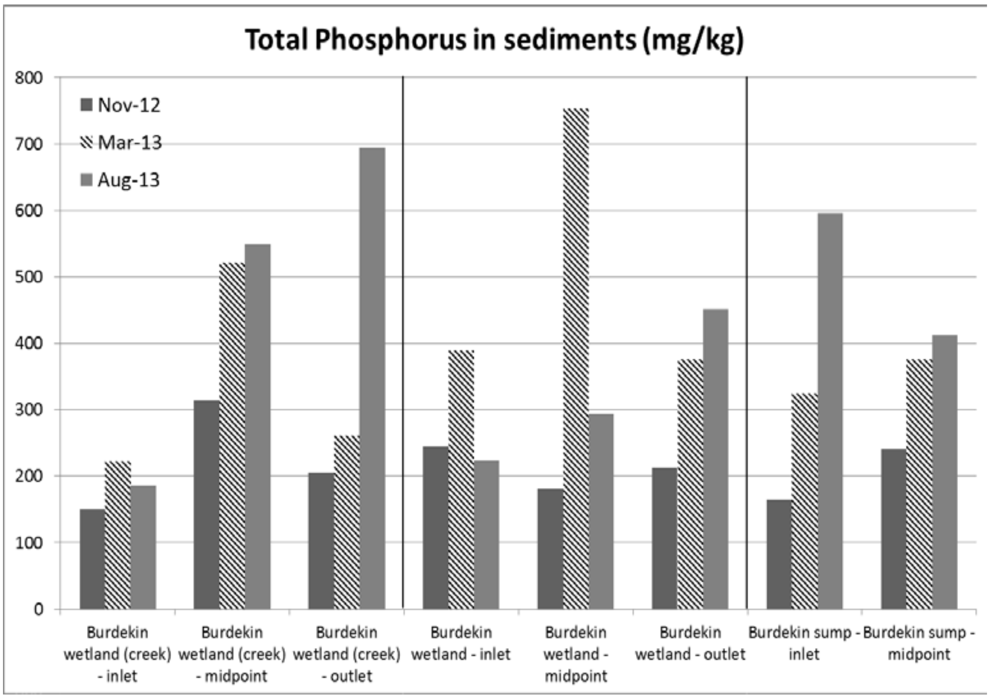


Figure 34: Total phosphorus in sediments of water samples collected from Burdekin sites.

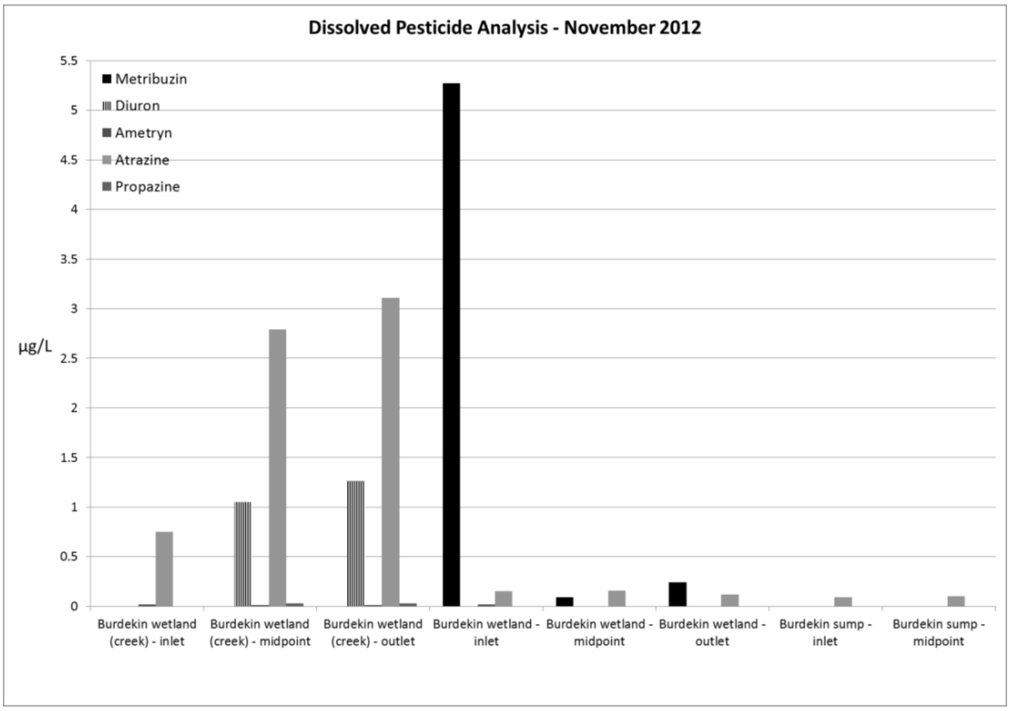


Figure 35: Dissolved pesticide analysis of water samples collected - Burdekin – Nov 2012.

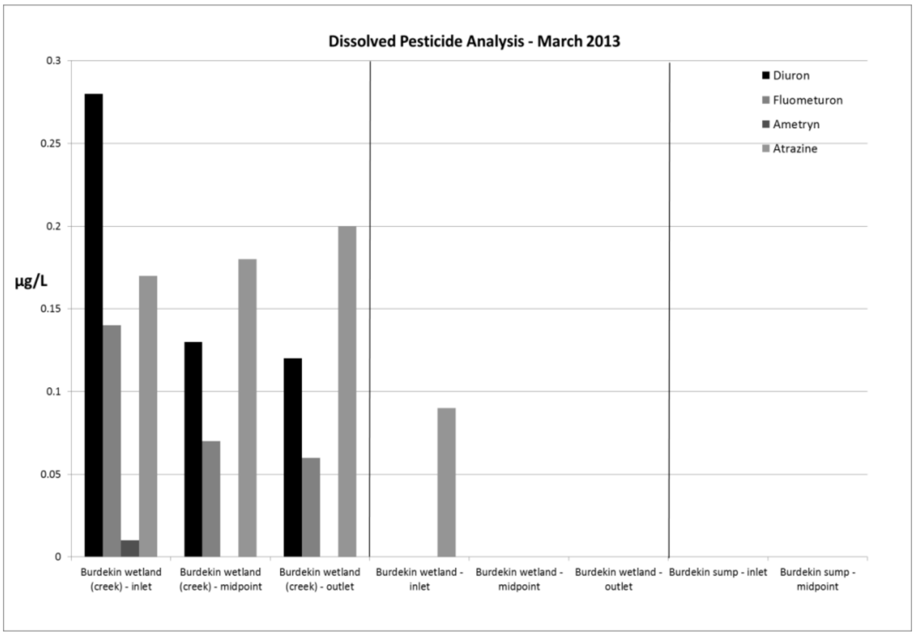


Figure 36: Dissolved pesticide analysis of water samples collected - Burdekin – Mar 2013.

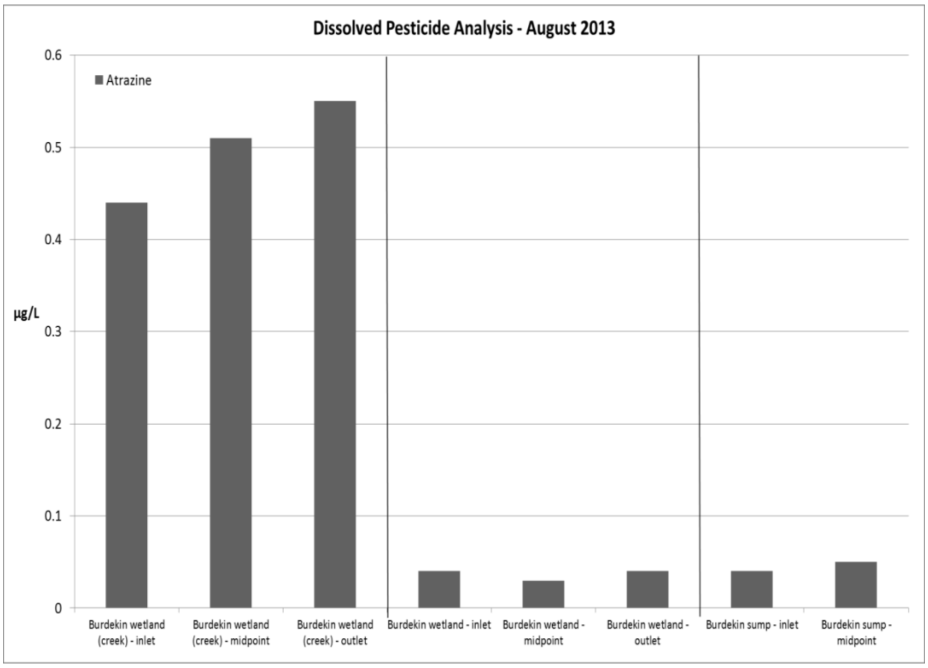


Figure 37: Dissolved pesticide analysis of water samples collected - Burdekin – Aug 2013.

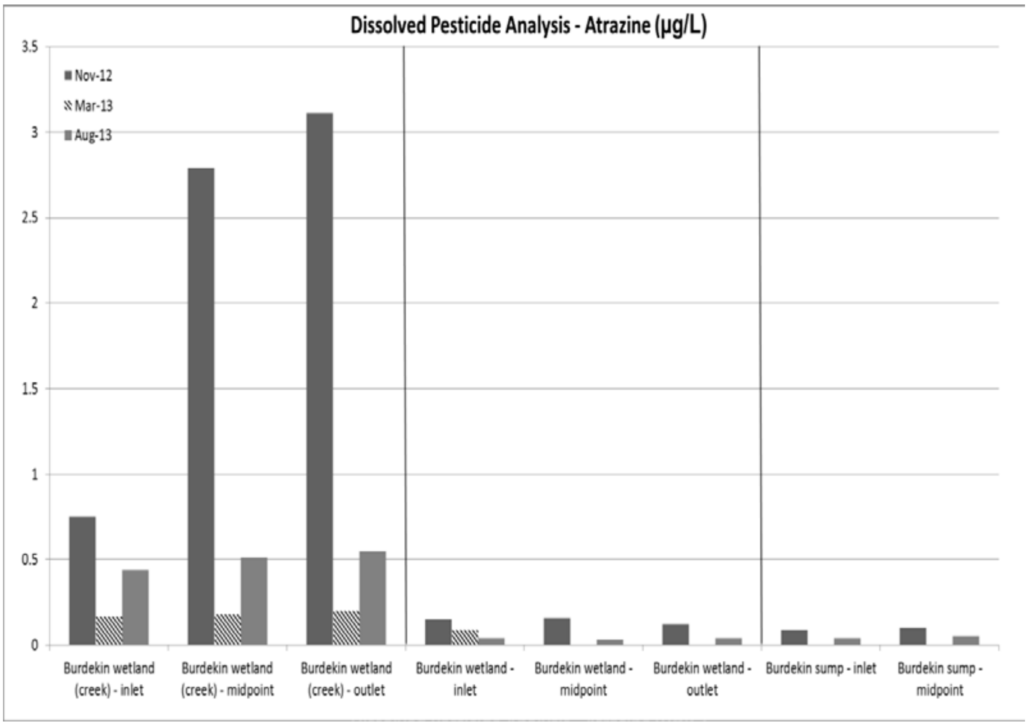


Figure 38: Dissolved pesticide analysis - Atrazine only - Burdekin - all sampling events.

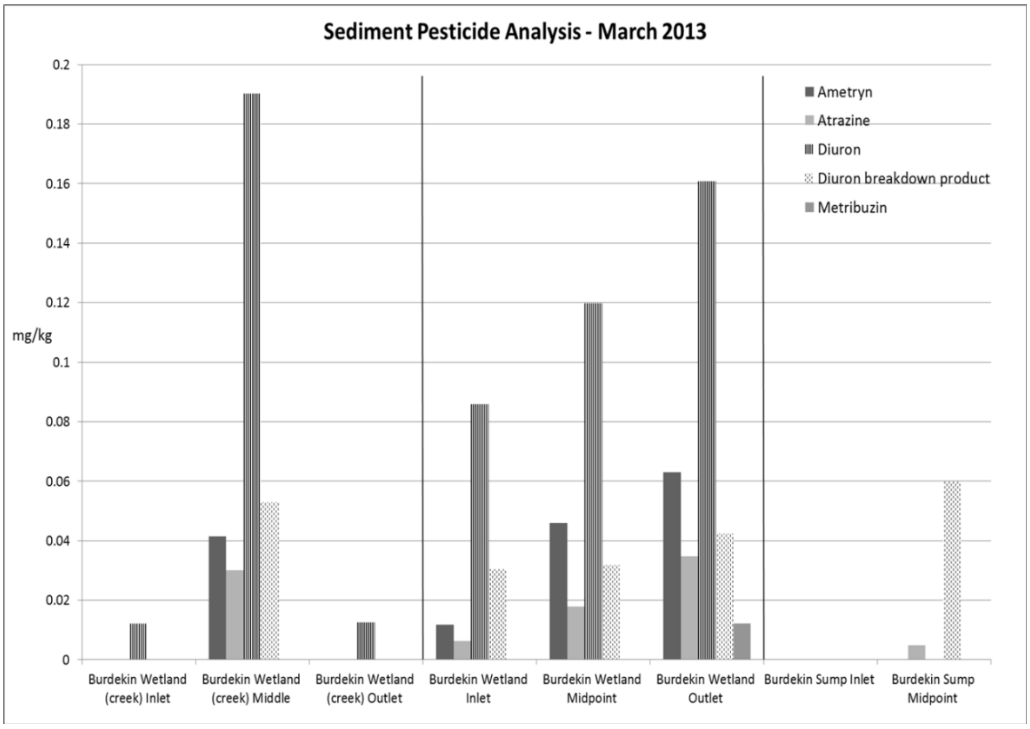


Figure 39: Pesticide analysis of sediments – Burdekin - Mar 2013.

Findings - discussion

Overall, there seems to be a pattern that particulate pollutants (nitrogen and phosphorus) may be settling in the wetlands, whereas dissolved pollutants appear at times to be increasing throughout the wetlands.

In the natural wetland, the increase in pesticides across the wetland suggests that there is input into the wetland other than from the “inlet collection site”. Farm practices on adjacent paddocks in the middle section of the wetland are potentially influencing the pesticide concentrations found. The farmer who provides access to this wetland (whose cane paddocks lie closer to the 'inlet' section of the wetland) had stopped using atrazine the year prior to sampling, so the increased detection of atrazine at the midpoint of the wetland might be due to inputs from the adjoining farm or from possible groundwater incursions.

In the Burdekin region, it should be noted, that irrigation and fertilising schedules dictate when the wetland will be receiving an influx of nutrients and pesticides. In the sampling events presented here, irrigations occurred prior to the November 2012 and August 2013 sampling, hence the higher levels of these pollutants in the wetland samples. Further, the Burdekin 'creek' wetland and reclamation sump sites are both designed for tailwater re-use, meaning that the levels of pollutants may be heightened by the repeated use (and addition of fertilisers/pesticides) of the water coming from off the paddock. This aspect of increasing pollutant loads in recycled water needs to be addressed in terms of safe nutrient and pesticide levels for both paddock and wetland health, but also in terms of reducing new applications given the amount already in the recycled water supply.

Metribuzin was detected in high amounts in the constructed wetland. This farmer recently replaced atrazine use with metribuzin and so, this high influx into the wetland could be testament to either high solubility and therefore loss from the paddock, or poor timing of application prior to a rainfall event.

7.2 Herbert River catchment sugarcane farms

The Herbert River wetland is approximately 3 hectares in area and between 0.5m and 0.75m in depth, draining between 60-80 hectares of sugarcane land (Figure 40). Drainage water into the wetland is derived solely from rainfall over the surrounding paddocks. The wetland is surrounded by grassed headlands and has a dirt road at the outlet end of the wetland. Because of the depth of the wetland invasion by aquatic weeds, including hymenachne, particularly during the dry season is an ongoing problem. A few macrophyte species, such as the native lily are present.

Figure 40: Constructed wetland in the Herbert region.

Though a suite of water soluble nutrients were analysed only data for filterable reactive phosphorus (FRP), dissolved inorganic nitrogen (DIN), and total suspended solids (TSS) are shown for the sampling event in August 2013 (Figure 41). FRP and TSS concentrations appear to decrease from the inlet to the outlet DIN however increased in concentration (approximately 3-fold) between the inlet and the outlet; the concentration at the midpoint was slightly lower than at the inlet. Dissolved organic nitrogen (DON), which is derived from organic production (e.g., wetland vegetation), did not show a wide range in variation between the inlet (565 µg/L), midpoint (575 µg/L), and outlet (557 µg/L).

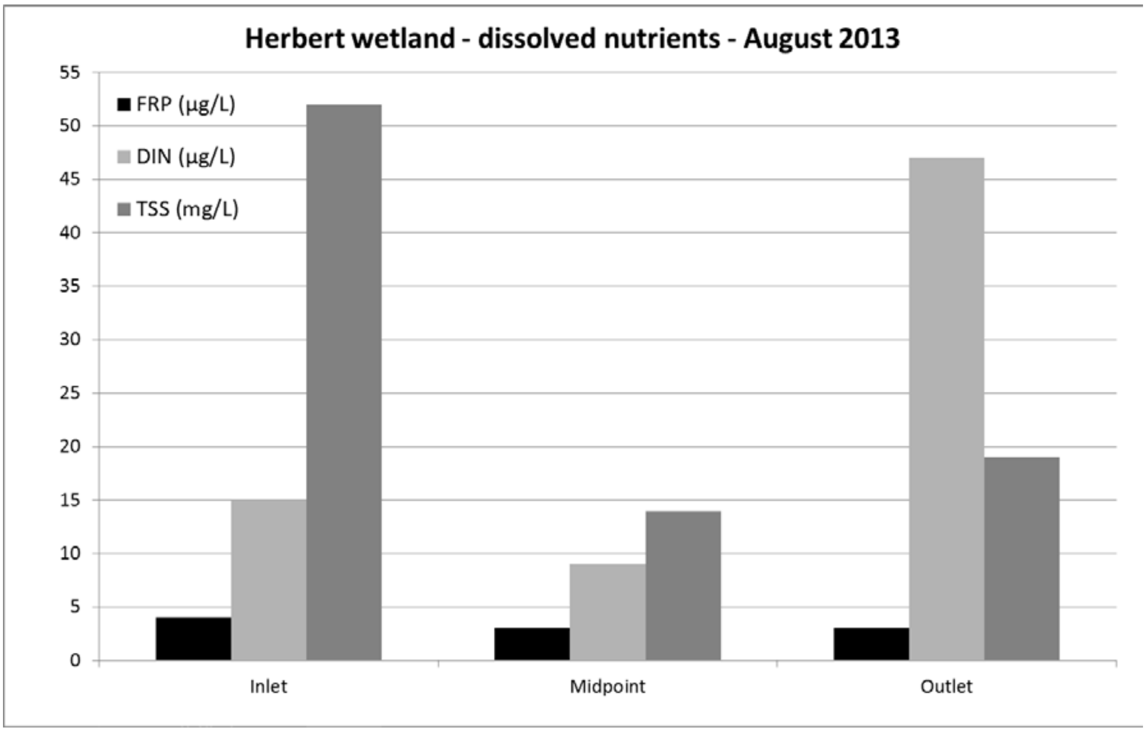


Figure 41: Nutrient sampling in the Herbert constructed - August 2013.

FRP - Filterable Reactive Phosphorus, DIN - Dissolved Inorganic Nitrogen, TSS - Total Suspended Sediments

In the Herbert wetland, the 'reverse dip' in the sediment nutrient concentrations from what was expected may be due to the almost homogeneous shallow depth profile of the wetland and the possible building up of sediments at the narrow outlet pipe. Since only one sampling event has been conducted and analysed for this wetland, it is difficult to draw conclusions as to the effectiveness of this particular wetland. Sediment sampling for total nitrogen and phosphorus showed a decrease in sediment concentrations in the midpoint of the wetland, but an overall increase in concentration of nitrogen between the inlet and outlet and a slight increase in phosphorus at the outlet (Figure 42).

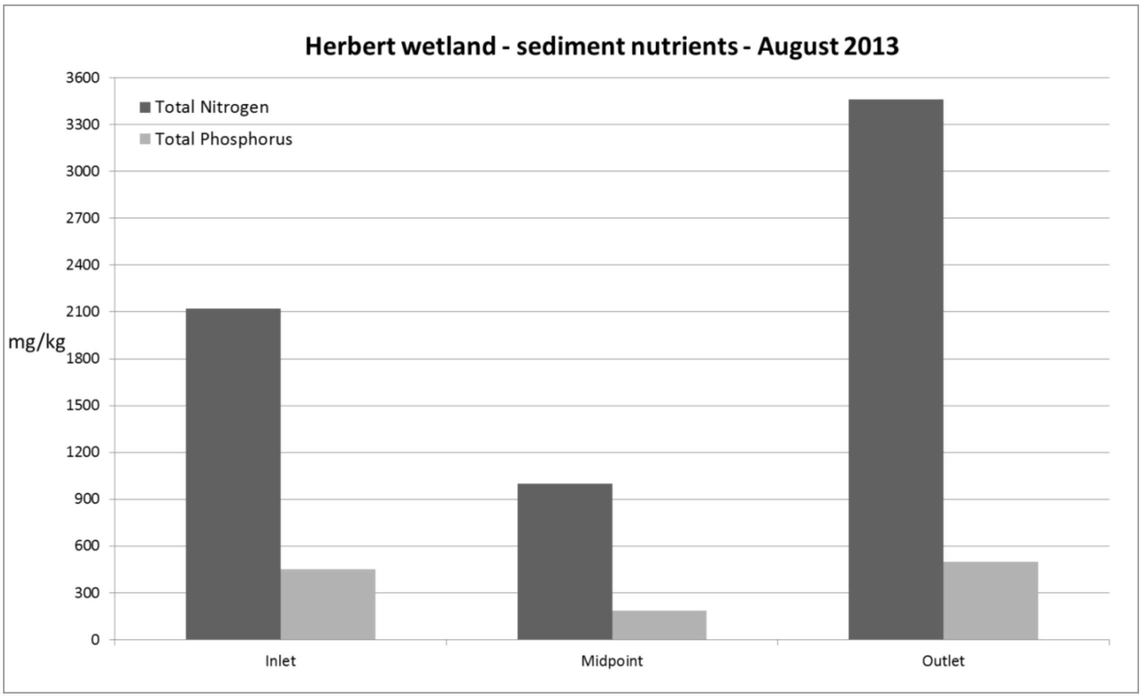


Figure 42: Sediment sampling in the Herbert constructed wetland - August 2013.

7.3 South Johnstone River catchment banana farm

The South Johnstone wetland is 2.45 hectares total area (350m long, 70m wide) and averages 1.3m deep, with three 2m deep pools; it was built in May 2010 (Figure 43). The wetland receives water from a 26 hectare banana farm catchment and it is a high rainfall area of the Wet Tropics with rainfall in the range of 3000 – 4000 mm/yr. There is a sediment basin and high flow bypass built into the design. It was designed to reduce total suspended solids and phosphorus by half and nitrogen by one third (Qld Wetlands Program). The macrophyte zone was planted with native aquatic grasses, reeds and sedges in August 2010, though the majority of these were subsequently removed by geese activity. The edges of the wetland were planted with native tree species and are routinely sprayed to remove Hymenachne infestations. The wetland has input from a barramundi farm directly upstream, as well as a composting site situated to the west/east about midway along the wetland. This site was sampled in December 2012 and March 2013 [77] at four sampling sites; Inlet/bypass drain, Inlet, Midpoint and Outlet (Figure 44).

Total suspended solids (TSS) decreased dramatically from the inlet-drain (41 mg/L) to the outlet (4.2 mg/L) in December 2012 (dry season), and decreased only slightly during March 2013 (wet season) (Figure 45). In December 2012 both dissolved inorganic nitrogen (DIN) (Figure 46) and phosphorus (FRP) (Figure 47) decreased across the wetland. However, in March 2013, though DIN and FRP both decreased across the wetland, FRP only decreased by approximately 20% as compared to 95% in December 2012. Also, in March 2013, initial concentrations of both DIN and FRP were higher than those in December 2012. Total nitrogen in sediments showed an increase at the midpoint of the wetland and a decrease at the outlet in December 2012 (Figure 48). In March 2013 however the sediment nitrogen concentrations varied throughout the wetland. Sediment phosphorus concentrations showed a similar pattern, with a peak concentration (2050 µg/L) in the midpoint of the wetland in December 2012 and lower concentrations at the inlet (980 µg/L) and outlet (810 µg/L) (Figure 49. In March 2013 however the concentrations varied across the wetland, though the inlet and outlet held slightly higher concentrations than the drain and midpoint (166 and 182 µg/L differences, respectively). No pesticides were detected (from a multi-residue pesticide suite analysis) in the South Johnstone wetland samples.

Figure 43: Constructed wetland in South Johnstone River catchment.

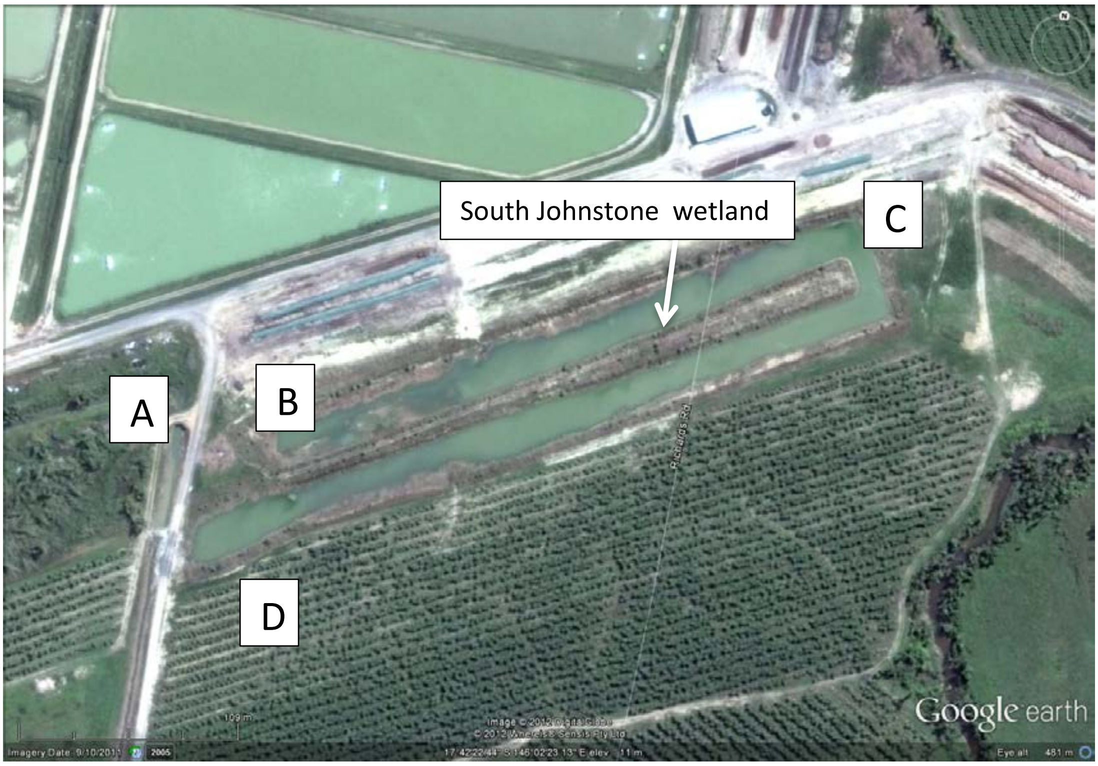


Figure 44: Sampling sites within the South Johnstone constructed wetland.

A - Inlet/bypass drain, B - Inlet, C - Midpoint, D - Outlet

The South Johnstone catchment has extreme rainfall and runoff, a high water table which prevents infiltration and potentially leads to exfiltration and has fast surface flows to streams, and possibly relatively fast sub-surface flows as well, thus transporting pesticides to streams even after infiltration. The combination of these factors means that in the Wet Tropics, buffer widths for effective trapping of these pollutants will need to be in excess of 10m (with good grass cover) and in some circumstances (with poorer grass cover) in excess of 30m. In addition for these particular herbicides, while infiltration may occur, no further trapping on infiltrated soil may occur 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 is the generally high water table which prevents infiltration and may promote instead exfiltration, returning previous upslope drainage to surface flow as seen in the Johnstone catchment studies of McKergow *et al*. [199, 200]. Connor *et al*. (2012), in the Mulgrave catchment, and McJannet *et al*. (2012), in the Tully-Murray catchment, also show that with the extreme rainfall and hydrological conditions present in these areas, trapping of sub-surface flow nitrate (a dissolved phase pollutant) through riparian areas or small wetlands is minimal.

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 Wet Tropics and mostly through infiltration. Trapping may improve to perhaps 30% where buffer widths are increased to 20m but still mainly by infiltration. The final fate of the infiltrated pesticides and nutrients is unclear but it is quite possible that transport to an adjacent drain or stream could be rapid with little further loss, thus minimising any net trapping [203].

Dry season sampling showed that the South Johnstone wetland can be effective in reducing the TSS and nutrient load exiting the wetland. This is most likely due to long residence times of the water within the wetland during these low flow periods. However, during higher flow events, during the wet season, there was minimal reduction, if any, to higher levels of nutrients entering into or being remobilised within the system. Interestingly, March 2013 sampling found less TSS entering the system than during December 2012, suggesting less erosion occurred in March 2013, though there were higher loads of nitrogen and phosphorus detected, which may point to remobilisation of nutrients within the wetland during the wet season.

Only relatively low K_{OC} value herbicides were 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. 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. For pesticides with low K_{OC} infiltration is more likely to 'remove' pesticides than sedimentation. Other sugarcane pesticides like chlorpyrifos and paraquat will be more likely to be trapped by sediment retention but were not the focus of this study.

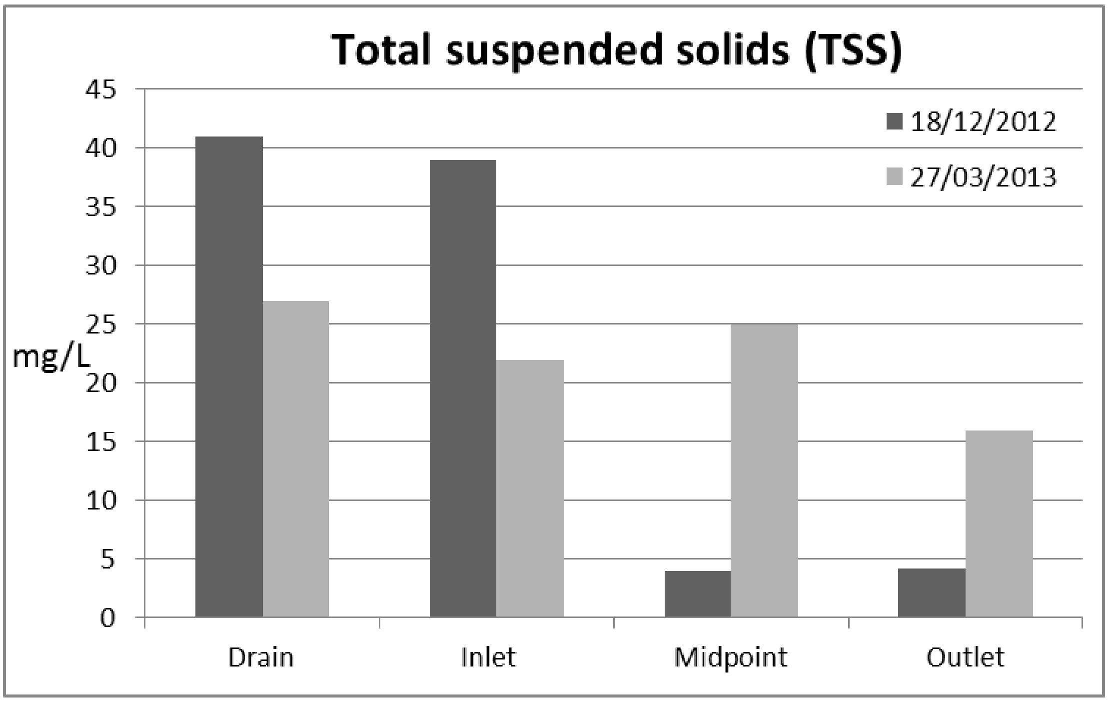


Figure 45: TSS in South Johnstone constructed wetland - Dec 2012 and Mar 2013.

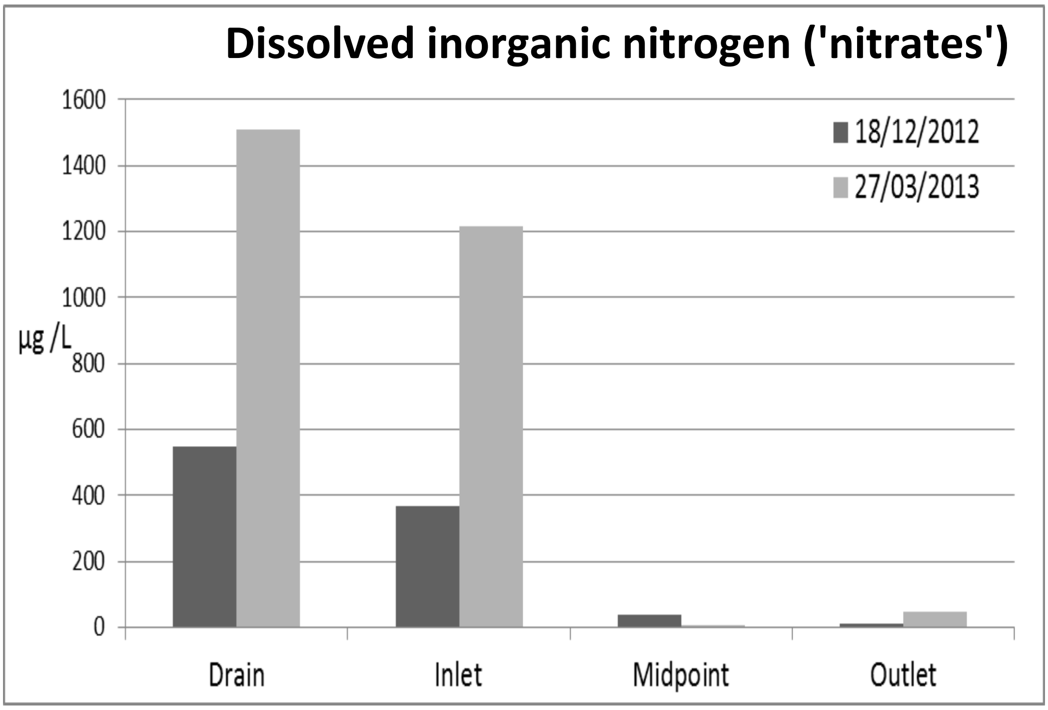


Figure 46: DIN in South Johnstone constructed wetland - Dec 2012 and Mar 2013.

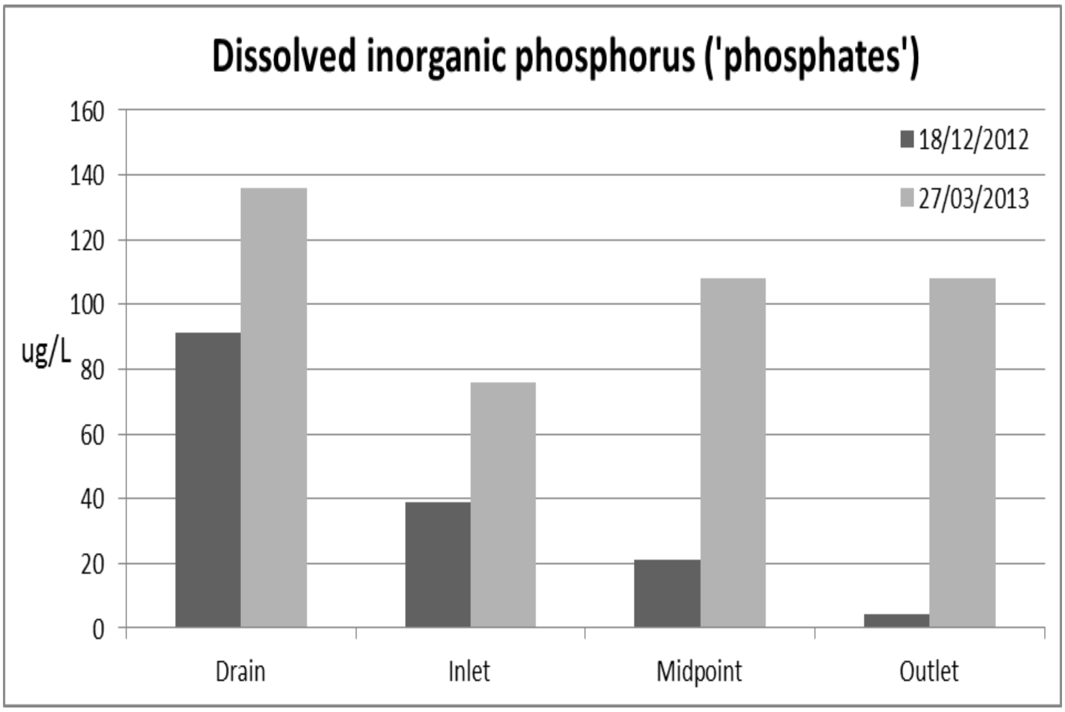


Figure 47: DIP in South Johnstone constructed wetland - Dec 2012 and Mar 2013.

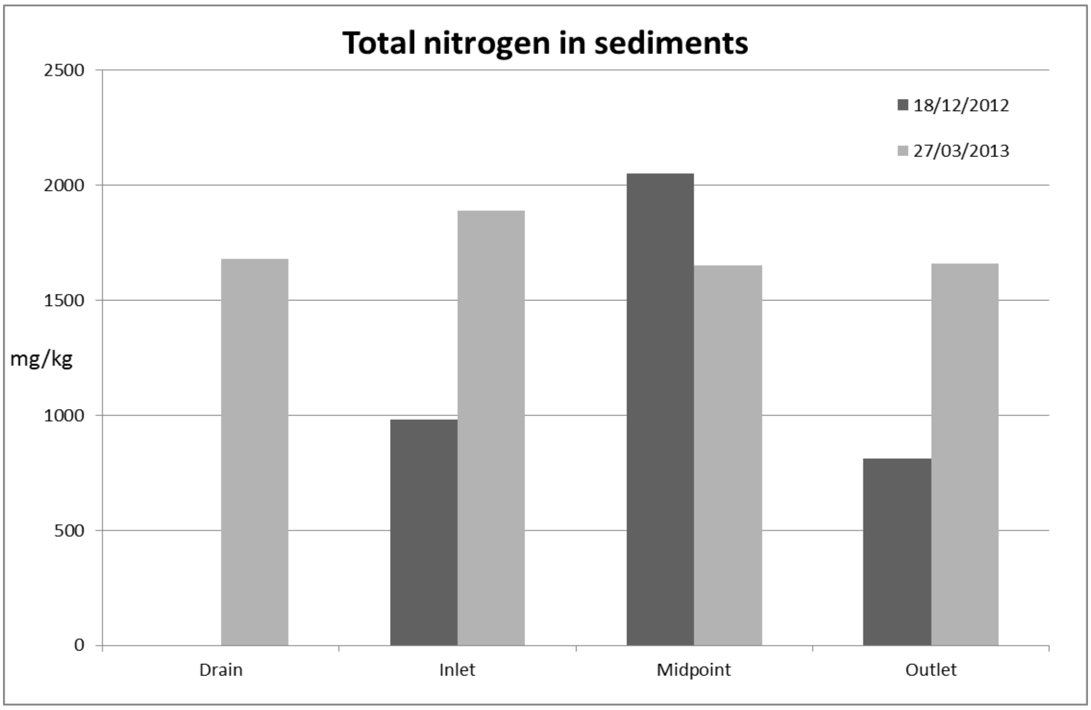


Figure 48: TN in sediments in South Johnstone constructed wetland - Dec 2012 and Mar 2013.

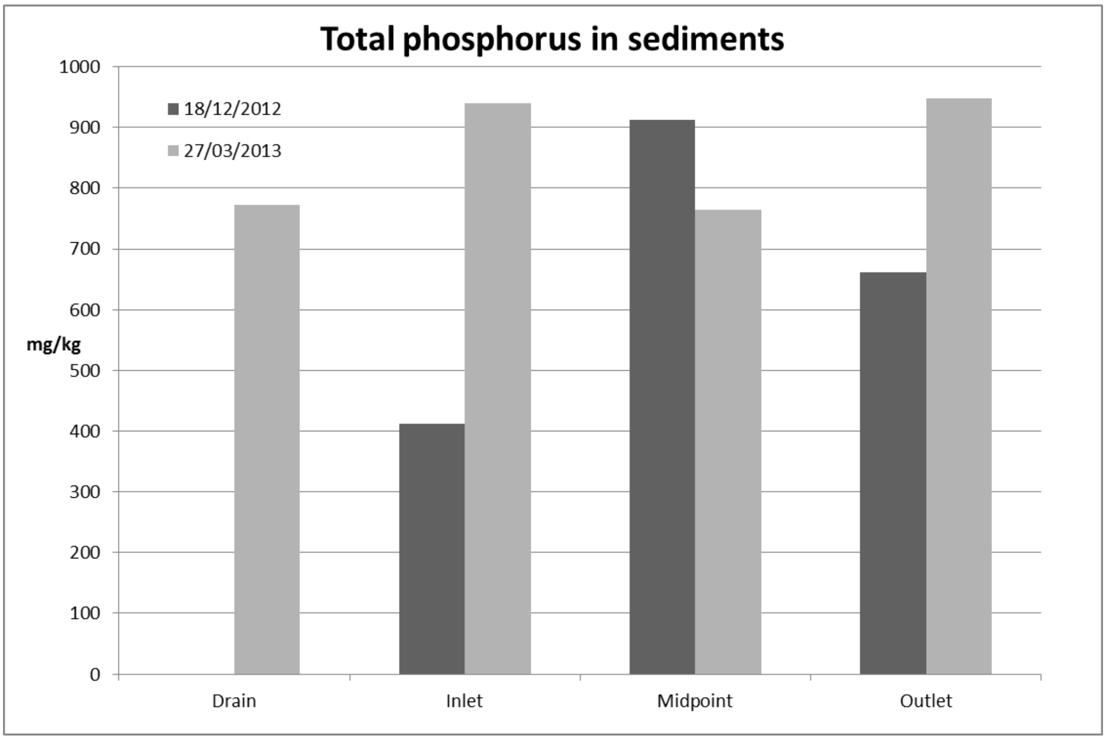


Figure 49: TP in sediments in South Johnstone constructed wetland - Dec 2012 and Mar 2013.

HYDRODYNAMIC MODELLING PROJECT

For this study we are using a two-dimensional floodplain hydrodynamic model (MIKE 21) to generate a range of water residence time in floodplains especially for sugarcane farms. The model was previously calibrated for the Tully floodplain and as a part of this study we have updated the model with boundary conditions for the recent flood in January 2013. We computed inflows to the floodplain and locally generated runoff using MIKE 11 rainfall-runoff model. The test run succeeded and primary checks were completed against stage heights at Euramo. Calibration against measured velocity data was carried out and then used to simulate floodplain residence time for different flow conditions.

Model description: The area of hydrodynamic modelling domain is 720 km^2 (30 x 24 km) and it covers almost entire floodplain (Figure 50), which is 32% of the total catchment area. Inputs to the model are land elevation, surface roughness and water sources. Model boundaries include inflows through the Tully and Murray Rivers, and through 4 creeks. Modelling domain was divided into 800,000 computational grids (30 x 30 m). Computational time increment was derived after satisfying numerical stability criteria. A time step of 4 seconds was used for this flood event.

Field Measurement

A field trip was conducted between 26-28 January 2013 while flood water was receding from the sugarcane fields. We visited a total of 53 sites in the floodplain between the Tully and Murray Rivers (locations are shown in Figure 50). In most places we accessed the site by a 4WD drive car and in some places where water depth was high we accessed the site by a small boat. At each location we took 3 to 5 readings and data were processed to calculate mean velocity at any particular site. We used a simple handheld velocity measuring device (FlowTracker Handheld-ADV, SonTek; http://www.sontek.com/flowtracker.php) to track flow direction and magnitude. Coordinates and time of measurement were recorded using the Garmin GPS 72H.

Measured velocity among 53 sites varied from 0.0 to 0.3 m/s and stagnant waters was present at many sites within sugarcane fields (Figure 51). High velocity was observed in places close to a cane drain. It is mentioned here that we conducted the measurements during the receding phase of flood water and in this case flow velocity is generally small everywhere. Currently we are using this information to calibrate the hydrodynamic model primarily by changing roughness parameter locally.

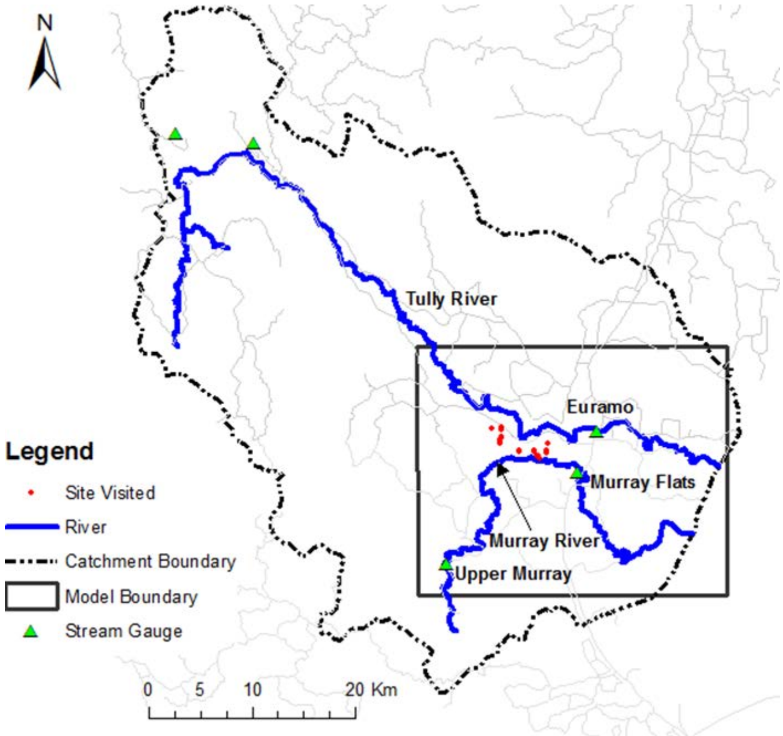


Figure 50: Sampling sites and extent of hydrodynamic model in Tully River catchment.

Figure 51: Measuring flow in sugarcane farms using a hand held flow tracker.

Photo: A. Palmer.

The full results of this study are currently being prepared for publication but a conference paper has been prepared, Karim *et al*. [147] and is presented in Appendix 3.

CONCLUSIONS

Vegetated systems (e.g. grassed strips, riparian vegetation, wetlands, sumps) are increasingly being incorporated into farming systems in north Queensland, especially in the catchments draining to the Great Barrier Reef (GBR) lagoon, to improve downstream water quality.

The objective of this review was to investigate the role and effectiveness of vegetated systems in trapping nutrients, pesticides and sediment in GBR catchments and hence preventing loading of downstream environments particularly the GBR. The following questions from DEHPs Reef Water Quality (RWQ) Research and Development program were addressed:

- What are the most effective methods for trapping loss of reef pollutants from sugarcane farms?
- What is the effectiveness of water quality filters like floodplains, riparian areas, grassed buffer strips and wetlands in reducing nutrients, sediments and pesticides?

The review investigated the effectiveness of a variety of vegetated systems at sites within the South Johnstone, Tully, Herbert and Burdekin catchments.

It included an evaluation of the likely performance of the different systems in different parts of GBR catchments (freshwater and estuarine) and between catchments and in different rainfall and hydrological conditions. This included some modelling of residence times as the main explanatory factor in the ability of systems to trap different materials. The systems reviewed include:

a. Grassed drains, buffer strips, headlands, inter-rows, etc.
b. Riparian vegetation
c. Natural wetlands (freshwater and estuarine)
d. Constructed wetlands
e. Reclamation sumps
f. Floodplains

The project had three main components:

1. An extensive literature search of relevant studies from Australia and overseas. This component included an analysis of where the review studies were applicable in the North Queensland context.
2. Field studies of the effectiveness of various sorts of constructed wetlands in trapping pollutants under different flow conditions.
3. Modelling water residence times in overbank flow conditions on the Tully-Murray flood plain and making preliminary conclusions as to the degree of likely trapping/removal of pollutants in such conditions.

A principal finding of the study is that the residence time of water in trapping mediums is an important measure of likely effectiveness of any vegetated area. Long residence times lead to effective trapping while short residence times are unlikely to trap anything. The trapping efficiency is also critically determined by the nature of the material (correlated with residence times) – in general the order of potential trapping is:

- Coarse particulate material (sediment – sand and gravel) – high efficiency.
- Medium particulate material (sediment - silt and adsorbed/absorbed contaminants) – moderate efficiency.
- Fine particulate material (sediment – fine silts and clay and the adsorbed/absorbed contaminants) – low efficiency.
- Dissolved material (e.g. nitrate, atrazine) – very low efficiency.

As a result of this relationship only at floodplain scales are residence times long enough to achieve some trapping of dissolved and fine particulate material in the wet season. Trapping in smaller vegetated systems is only effective in the dry season or in low flow conditions. The low flow conditions of irrigation tailwater flows is a special case in the lower Burdekin where higher levels of trapping of fine particulate and dissolved material can occur.

Permanent trapping of contaminants is also dependent on the trapped material not being removed by flushing on the next or subsequent high flows. Sump systems which recycle the trapped material back on to the paddock can achieve high trapping effectiveness. Essential to this working are well designed high flow bypass systems so the trapped material is not flushed downstream.

Long residence times of materials like atrazine and nitrate in the trap are necessary to allow processes like denitrification (and hence removal of nitrogen as N_2) and pesticide chemical degradation to benign chemical forms to occur. Degradation half-lives of pesticides commonly used in the sugar industry are in the order of 50 – hundreds of days. Hence to degrade significant amounts of these chemicals, they must be held in the trap for long periods.

The lower Burdekin field studies showed that in the wetlands studied there is evidence that suspended sediment and particulate nutrients (nitrogen and phosphorus) may be settling in the wetlands, whereas dissolved pollutants appear at times to be increasing throughout the wetlands. The increase in pesticide concentrations across the wetland suggests that there is input into the wetland other than from the "inlet collection site". Farm practices on adjacent paddocks in the middle section of the wetland are potentially influencing the pesticide concentrations found. Further, the Burdekin 'creek' wetland and reclamation sump sites are both designed for tailwater re-use, meaning that the levels of pollutants may be increased by the repeated use (and addition of fertilisers/pesticides) of the water coming from off the paddock. This aspect of increasing pollutant loads in recycled water needs to be addressed in terms of safe nutrient and pesticide levels for both paddock and wetland health, but also in terms of reducing new applications given the amount already in the recycled water supply.

It was also evident that pesticide (PSII herbicide) concentrations exceeded relevant water quality guidelines frequently in the wetlands during the study. This was most notable for atrazine and diuron matching the elevated concentrations of these herbicides found regularly in Barratta Creek and other waterways of the lower Burdekin [70-72]. Also notable were the concentrations of metribuzin found where one farmer had moved from using atrazine to this 'alternative' herbicide. Metribuzin is now frequently being found in sugarcane growing areas in waterways as its use increases in place of the restricted-use herbicide diuron, but also where it replaces atrazine [72].

The South Johnstone wetland study of suspended sediment and particulate nutrients showed considerable trapping in low flow conditions but little trapping in the wet (wetter!) season. The South Johnstone catchment has extreme rainfall and runoff, a high water table which prevents infiltration and potentially leads to exfiltration and has fast surface flows to streams, and possibly relatively fast sub-surface flows as well, thus transporting dissolved contaminants including pesticides to streams after infiltration. Another factor mitigating against effective trapping through infiltration is the generally high water table which prevents infiltration and may promote instead exfiltration, returning previous upslope drainage to surface flow as seen in the Johnstone catchment studies of McKergow *et al.* [199, 200]. Connor *et al*. [64], in the Mulgrave catchment, and McJannet *et al*. [194, 195], in the Tully-Murray catchment, also show that with the extreme rainfall and hydrological conditions present in these areas, trapping of sub-surface flow nitrate (a dissolved phase pollutant) through riparian areas or small wetlands is minimal.

Dry season sampling showed that the South Johnstone wetland can be effective in reducing the TSS and nutrient load exiting the wetland. This is most likely due to long residence times of the water within the wetland during these low flow periods. However, during higher flow events, during the wet season, there was minimal reduction, if any, to higher levels of nutrients entering into or being remobilised within the system.

In the Herbert catchment wetland, the 'reverse dip' in the sediment nutrient concentrations from what was expected may be due to the almost homogeneous shallow depth profile of the wetland and the possible building up of sediments at the narrow outlet pipe. Since only one sampling event has been conducted and analysed for this wetland, it is difficult to draw conclusions as to the effectiveness of this particular wetland. Sediment sampling for total nitrogen and phosphorus showed a decrease in sediment concentrations in the midpoint of the wetland, but an overall increase in concentration of nitrogen between the inlet and outlet and a slight increase in phosphorus at the outlet.

While residence time of catchment runoff in the GBR lagoon before it is transported to the open ocean is reported in many studies, quantitative estimates of water residence time in the river-floodplain system for majority of the GBR catchments is generally unknown. The floodplain study focused on the Tully-Murray catchment in the Wet Tropics which is frequently flooded (2 to 3 floods in each year) and carries a large quantity of land sourced contaminants to the GBR lagoon during overbank flow events. A two-dimensional floodplain hydrodynamic model (MIKE 21) was used to simulate spatial and temporal variations of velocities across the floodplain. This information was used to estimate mean residence time on the floodplain before flood water from agricultural lands reaches coastal waters. The model was calibrated using measured inundation depths and velocities at 53 locations on the floodplain for a recent flood in 2013, which was about 2.3 times bigger than a mean annual flood. A range of water residence times has been extracted for in-channel and floodplain waters for different floods. Typically in-channel residence times are of the order of one day while floodplain times can be in the order of several to 15 days. This information can be used to estimate denitrification, pesticide degradation and sedimentation by combining residence time with pollutant decay rules to assess the effectiveness of vegetated areas.

While it is clear that only constructed wetlands/sumps/vegetated areas with long residence times are capable of significant levels of trapping of all pollutants (except coarse sediment) further research is needed to better be able to accurately quantify the potential degree of trapping in the varying circumstances across the GBR catchment. Experimental work in the current project only focussed in the Wet Tropics and lower Burdekin areas. While the lessons learnt here and from the literature survey may be applicable in other parts of the GBR catchment, some of the conclusions would need to be validated in the actual region e.g. the Fitzroy catchment. To be able to predict accurately the effectiveness of particular designs in a Queensland context more research is needed. In particular the role of vegetation on river floodplains in slowing up flow in overbank flow events needs to be quantified in order to be able to predict residence times and likely degree of trapping through sedimentation of fine sediments, denitrification and pesticide degradation. This may be very important in assessing the effects stemming from changes to the Vegetation Management Act (1999) where increased clearing of riparian and frontage country vegetation may eventuate.

APPENDICES

Appendix 1. Raw data from regionally specific cases.

SITE	Sample Date	Total Kjeldahl Nitrogen as N (mg/kg)	Total Phosphorus as P (mg/kg)	Total Organic Carbon (%)
Burdekin natural wetland - inlet	23/11/2012	1780	150	2.10
Burdekin natural wetland - midpoint	23/11/2012	2990	314	1.31
Burdekin natural wetland - outlet	23/11/2012	1490	205	8.56
Burdekin constructed wetland - inlet	23/11/2012	5690	244	1.16
Burdekin constructed wetland - midpoint	23/11/2012	1990	181	0.81
Burdekin constructed wetland - outlet	23/11/2012	310	213	0.59
Burdekin sump - inlet	23/11/2012	400	164	0.46
Burdekin sump - midpoint	23/11/2012	1680	241	0.60
S. Johnstone inlet	18/12/2012	980	412	0.77
S. Johnstone midpoint	18/12/2012	2050	912	2.15
S. Johnstone outlet	18/12/2012	810	523	0.72
S. Johnstone drain	27/03/2013	1680	773	
S. Johnstone inlet	27/03/2013	1890	939	
S. Johnstone midpoint	27/03/2013	1650	765	
S. Johnstone outlet	27/03/2013	1660	947	
Herbert wetland - outlet	27/03/2013	3460	498	
Herbert wetland - inlet	27/03/2013	2120	451	
Herbert wetland - midpoint	27/03/2013	1000	187	
Burdekin natural wetland - inlet	28/03/2013	1670	222	
Burdekin natural wetland - midpoint	28/03/2013	2890	522	
Burdekin natural wetland - outlet	28/03/2013	2400	261	
Burdekin constructed wetland - inlet	28/03/2013	1930	389	
Burdekin constructed wetland - midpoint	28/03/2013	4860	753	
Burdekin constructed wetland - outlet	28/03/2013	2280	376	
Burdekin sump - inlet	28/03/2013	1200	324	
Burdekin sump - midpoint	28/03/2013	1620	376	
Burdekin natural wetland - inlet	12/08/2013	1630	186	
Burdekin natural wetland - midpoint	12/08/2013	4330	548	
Burdekin natural wetland - outlet	12/08/2013	1400	694	
Burdekin constructed wetland - inlet	12/08/2013	270	223	
Burdekin constructed wetland - midpoint	12/08/2013	2030	294	
Burdekin constructed wetland - outlet	12/08/2013	2640	451	
Burdekin sump - inlet	12/08/2013	670	596	
Burdekin sump - midpoint	12/08/2013	1160	412	

Site	Sample Date	Temperature °C	pH - quanta reading averaged over all depths	EC (µS/cm) - quanta reading averaged over all depths	Total Suspended Solids (mg/L)	Turbidity (NTU) - Quanta reading @ 0.25m	Total Nitrogen (µg N/L)	Total Filterable N (µg N/L)	Ammonia (µg N/L)	Total Phosphorus (µg P/L)	Total Filterable P (µg P/L)	Filterable Reactive P (µg P/L)	NOX	Particulate N (µg N/L)	PN proportion of TN	DON (µg N/L)	NOX proportion of TN	Particulate P (µg P/L	DOP (µg P/L)	TN:TP Molar Ratio	DIN (µg N/L)
Burdekin natural wetland - inlet	23/11/2012				10		931	626	8	158	104	33	15	305	32.8%	603	1.6%	54	71.0	13.0	23
Burdekin natural wetland - midpoint	23/11/2012	26.5	7.70	177	5.4	5.7	790	608	15	93	71	26	188	182	23.0%	405	23.8%	22	45.0	18.8	203
Burdekin natural wetland - outlet	23/11/2012	26.5	7.60	173	3.1	15.7	914	595	21	115	66	19	104	319	34.9%	470	11.4%	49	47.0	17.6	125
Burdekin constructed wetland - inlet	23/11/2012				8.4		8003	7822	471	31	19	2	6885	181	2.3%	466	86.0%	12	17.0	570.9	7356
Burdekin constructed wetland - midpoint	23/11/2012	29.6	9.55	526	1.8	7.9	1577	1376	19	43	20	1	20	201	12.7%	1337	1.3%	23	19.0	81.1	39
Burdekin constructed wetland - outlet	23/11/2012				5		2402	2167	45	41	16	4	1089	235	9.8%	1033	45.3%	25	12.0	129.6	1134
Burdekin sump - inlet	23/11/2012				14		955	605	16	66	20	4	21	350	36.6%	568	2.2%	46	16.0	32.0	37
Burdekin sump - midpoint	23/11/2012	29.1	8.19	317	7.4	11.8	699	536	8	34	12	3	20	163	23.3%	508	2.9%	22	9.0	45.5	28
S. Johnstone drain	18/12/2012				41		1282	890	18	183	102	91	530	392	30.6%	342	41.3%	81	11.0	15.5	548
S. Johnstone inlet	18/12/2012	28.7	7.58	74.5	39	72	1006	730	102	125	51	39	265	276	27.4%	363	26.3%	74	12.0	17.8	367
S. Johnstone midpoint	18/12/2012	30.6	7.66	72.3	4	13	482	382	22	56	40	21	15	100	20.7%	345	3.1%	16	19.0	19.0	37
S. Johnstone outlet	18/12/2012	30.3	7.31	67.1	4.2	11.5	570.5	438.5	1.5	49	24.5	4.5	7	132	0.233	430	0.012	24.5	20	25.7	8.5
S. Johnstone drain	27/03/2013				27		1695	1573	16	192	157	136	1492	122	7.2%	65	88.0%	35	21.0	19.5	1508
S. Johnstone inlet	27/03/2013	28.8	6.73	58	22	117	1595	1282	26	169	120	76	1191	313	19.6%	65	74.7%	49	44.0	20.9	1217
S. Johnstone midpoint	27/03/2013	30.6	6.70	64	25	126	622	448	6	250	146	108	2	174	28.0%	440	0.3%	104	38.0	5.5	8
S. Johnstone outlet	27/03/2013	30.2	7.11	74	16	36.4	795	702	44	239	140	108	3	93	11.7%	655	0.4%	99	32.0	7.4	47
Herbert wetland - inlet	27/03/2013	32.1	8.33	748	52	912	602	580	13	54	20	4	2	22	3.7%	565	0.3%	34	16.0	24.7	15
Herbert wetland - midpoint	27/03/2013	30.3	8.73	779	14	13.8	656	584	8	31	15	3	1	72	11.0%	575	0.2%	16	12.0	46.8	9
Herbert wetland - outlet	27/03/2013	30.1	7.48	628	19	29.9	619	604	38	70	24	3	9	15	2.4%	557	1.5%	46	21.0	19.6	47
Burdekin natural wetland - inlet	28/03/2013	26.2	7.18	316	15	19	886	587	48	309	112	101	1	299	33.7%	538	0.1%	197	11.0	6.3	49
Burdekin natural wetland - midpoint	28/03/2013	26.5	7.20	332	9.8	18	773	518	60	255	150	133	2	255	33.0%	456	0.3%	105	17.0	6.7	62
Burdekin natural wetland - outlet	28/03/2013	26.8	7.35	343	8.8	9	620	459	15	234	166	139	2	161	26.0%	442	0.3%	68	27.0	5.9	17
Burdekin constructed wetland - inlet	28/03/2013	29.9	9.40	258	82	234	1414	845	17	120	29	6	7	569	40.2%	821	0.5%	91	23.0	26.1	24
Burdekin constructed wetland - midpoint	28/03/2013	27.6	8.03	234	5.1	11.7	649	524	21	34	11	2	1	125	19.3%	502	0.2%	23	9.0	42.2	22
Burdekin constructed wetland - outlet	28/03/2013	27.8	8.52	226	5.5	9.2	1122	572	7	65	15	3	3	550	49.0%	562	0.3%	50	12.0	38.2	10
Burdekin sump - inlet	28/03/2013	26.0	8.20	231	35	108	543	364	6	73	34	23	5	179	33.0%	353	0.9%	39	11.0	16.4	11
Burdekin sump - midpoint	28/03/2013	27.2	8.22	226	14	48	592	429	8	60	23	7	9	163	27.5%	412	1.5%	37	16.0	21.8	17
Burdekin natural wetland - inlet	12/08/2013	21.2	8.09	1183	21	1816	614	520	13	387	377	335	81	94	15.3%	426	13.2%	10	42.0	3.5	94
Burdekin natural wetland - midpoint	12/08/2013	21.8	7.30	1183	4.2	34.3	622	600	29	131	119	107	85	22	3.5%	486	13.7%	12	12.0	10.5	114
Burdekin natural wetland - outlet	12/08/2013	21.7	7.56	1186	3.3	31.6	631	507	9	106	74	53	83	124	19.7%	415	13.2%	32	21.0	13.2	92
Burdekin constructed wetland - inlet	12/08/2013	22.1	6.71	517	7.1	18.4	3797	3305	350	61	16	5	1614	492	13.0%	1341	42.5%	45	11.0	137.6	1964
Burdekin constructed wetland - midpoint	12/08/2013	22.5	9.03	451	5	32.1	3891	3663	127	26	13	4	2384	228	5.9%	1152	61.3%	13	9.0	330.9	2511
Burdekin constructed wetland - outlet	12/08/2013	21.1	7.60	455	20	78.3	2475	2136	44	48	20	4	1045	339	13.7%	1047	42.2%	28	16.0	114.0	1089
Burdekin sump - inlet	12/08/2013	24.9	7.13	614	5.1	13.8	3421	3365	1208	73	42	11	130	56	1.6%	2027	3.8%	31	31.0	103.6	1338
Burdekin sump - midpoint	12/08/2013	22.4	7.91	561	11	65	2649	2474	378	44	25	5	433	175	6.6%	1663	16.3%	19	20.0	133.1	811

Appendix 2. Hydrodynamic modelling project (Karim *et al*., 2013).

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Hydrodynamic modelling of floodplain flow residence time in a wet tropical catchment, north eastern Australia

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Abstract: Vegetated systems (e.g. grassed strips, riparian vegetation, wetlands, sumps) are increasingly being incorporated into farming systems in north Queensland, especially in the catchments draining to the Great Barrier Reef (GBR) lagoon, to improve downstream water quality. The residence time of water in trapping mediums is an important measure of likely effectiveness of any vegetated area. While residence time of catchment runoff in the GBR lagoon before it is transported to the open ocean is reported in many studies, quantitative estimates of water residence time in the river-floodplain system for majority of the GBR catchments is generally unknown. This study focused on the Tully-Murray catchment in the wet tropics which is frequently flooded (2 to 3 floods in each year) and carries a large quantity of land sourced contaminants to the GBR lagoon during overbank flow events. A two-dimensional floodplain hydrodynamic model (MIKE 21) was used to simulate spatial and temporal variations of velocities across the floodplain. This information was used to estimate mean residence time on the floodplain before flood water from agricultural lands reaches coastal waters. The model was calibrated using measured inundation depths and velocities at 53 locations on the floodplain for a recent flood in 2013, which was about 2.3 times bigger than a mean annual flood. A range of water residence times has been extracted for in-channel and floodplain waters for different floods. This information is useful to estimate denitrification, pesticide degradation and sedimentation by combining residence time with pollutant decay rules to assess the effectiveness of vegetated areas.

Keywords: *Floodplain, residence time, vegetated system, water quality, GBR*

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1. INTRODUCTION

Excess terrestrial loads of sediment, nutrients and pesticides derived from agriculture have been recognised as the major cause of water quality degradation in estuaries and marine waters of the Great Barrier Reef (GBR) lagoon (Brodie et al. 2012a). Since European settlement the discharge of pollutants to the GBR lagoon has greatly increased; the sediment yield has risen by an estimated factor of 5.5, while total nitrogen and phosphorus loads have increased by factors of about 6 and 9, respectively (Kroon et al., 2012). Deterioration of water quality reaching the GBR lagoon and subsequent degradation of marine habitats continues to be attributed to land use modifications and land management practices in GBR catchments (Brodie et al., 2012a). In order to protect the GBR, efforts to mitigate and prevent any further degradation are increasing. Introducing a vegetated patch adjacent to farming lands or revegetating floodplains and wetlands could be an option to reduce agricultural loads to GBR. However, only limited research has been conducted into the ability and efficiency of vegetated systems to trap pollutants. It is well documented that the degree of pollutant removal greatly depends on flow residence time (e.g. Wang et al., 2007; Brodie et al., 2012b). An important issue is to estimate the flow velocity on the floodplain, which greatly differs from in-channel flow (Helton et al., 2012).

The bulk of land derived pollutants to the GBR lagoon are delivered by river floods (Wallace et al. 2009) and much of the rivers' freshwater discharge occurs in short-lived flow events, with on average 2 to 3 floods per year for rivers in the wet tropics and 1 flood per year for the dry tropical rivers (Furnas, 2003). The residence times of these flow events varies between catchments in the range of few days for a small river catchment (e.g. Ross River, Tully River) to a few weeks and up to a few months for the two largest GBR catchments, the Burdekin and Fitzroy (Brodie et al., 2012b). While there have been a number of previous studies to estimate residence times of water in the GBR lagoon (e.g. Luick et al., 2007; Wang et al., 2007; Choukroun et al., 2010) using hydrodynamic modelling and remote sensing technologies, studies on estimating residece time on the floodplain environment are still limited. A few studies estimated the river flood times for the selected catchments of GBR lagoon (e.g. Lambrechts et al., 2010; Webster and Ford, 2010). The residence time in the floodplain however differs greatly from the mean flood speed due to the complex nature of floodplain flow (Helton et al., 2012). In this study we have investigated water residence time in the Tully-Murray catchment which is a relatively small catchment but discharges large quantities of contaminants to the GBR lagoon during floods.

2. STUDY AREA AND DATA

2.1. Location and hydrology

The Tully-Murray catchment is located in the Wet Tropics region of the north-east coast of Australia (Figure 1) and is one of the many catchments that drain into the GBR Lagoon. It covers an area of 2072 km^2, of which 832 km^2 is floodplain (Karim et al., 2008). Topography varies from steep rainforest-covered mountains in the upper catchment to low-relief floodplains which are largely developed for agriculture (mainly sugarcane and bananas) and grazing.

The Tully and Murray Rivers are the two main waterways on the floodplain (130 and 70 km long, respectively) and receive catchment runoff through numerous tributaries. The length of floodplain varies from 38 km along the Tully River and 42 km along the Murray River. The Tully River has well developed natural levee banks which have been

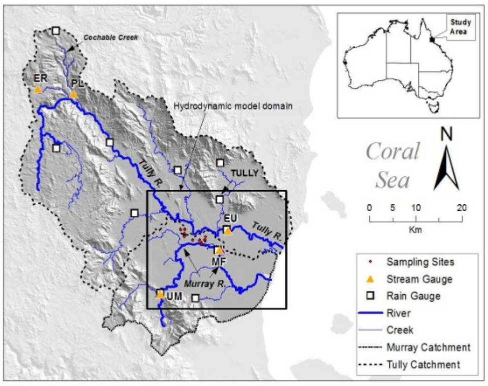

Figure 1 Study area map showing the Tully and Murray catchments and major streams. The rectangle shows the hydrodynamic modelling domain (30 × 24 km). The location of stream gauges in the upper catchment (ER, Ebony Road, PL, Power line) and in the floodplain (UM, Upper Murray; MF, Murray Flats; ER, Euramo) is shown. The red dots are the sampling sites where water depth and velocity were measured.

extended artificially in places. The general slope of the floodplain between the Tully and Murray rivers is from the elevated levees adjacent to the Tully to lower elevations adjacent to the Murray. This slope results in the flow of floodwater from the Tully into the Murray during overbank floods. The catchment receives a mean annual rainfall of between 2000 and 4000 mm depending on location, with most rainfall (60-80%) occurring during the wet season between December and April. During this period floodplains are inundated by an average of 3 to 4 floods per year (Wallace et al., 2009). The mean annual flood has a discharge about twice the bank-full discharge. Because the topography of the Tully–Murray floodplain is very flat and the rivers are quite close, water from the two rivers often merges during a flood.

2.2. Stream flow

Gauged data were used to specify model boundaries and to calibrate the rainfall-runoff and hydrodynamic models. Mean daily discharge and stage height data were obtained from the Queensland Department of Environment and Resource Management (DERM) for the period of 1972-2013. The Tully River has two gauges in the upper catchment and one gauge on the floodplain, while the Murray River has one gauge in the upper catchment and one on the floodplain (Figure 1). Gauged data in the upper catchment were used to estimate inflow at the hydrodynamic model boundaries and floodplain stage-height records were used to calibrate the model. Flows recorded in the lower Tully (at Euramo) were used to test the hydrodynamic model predictions of flood speed and peak arrival at Euramo. We also conducted field sampling between 26-28 January 2013 to measure the inundation depth and velocity across the floodplain when flood waters were receding from the sugarcane fields. These data were used to test the hydrodynamic model predictions of water depth and velocity at different locations. A total of 53 sites were visited in the floodplain between the Tully and Murray Rivers (locations are shown on Figure 1). In most places we accessed the site by a 4WD car and in some places where water depth was high we accessed the site by a small boat. At each location we took 3 to 5 readings and data were processed to calculate mean velocity at that point. We used a simple handheld velocity measuring device (FlowTracker Handheld-ADV, SonTek) to track flow direction and magnitude. Coordinates and time of measurement were recorded using the Garmin GPS 72H.

2.3. Topography and surface roughness

The topography of the study area used in the hydrodynamic model was a 30 m grid digital elevation model (DEM). This DEM was primarily based on one coarse resolution (± 0.7) areal photogrammetry data for the entire floodplain area and one fine resolution (± 0.15m vertical accuracy) data set along the main highway and railway. The bathymetry of the Tully and Murray Rivers and major creeks was added to the DEM using surveyed cross-sections. As creek widths are relatively small (10 to 70 m) and at many locations less than the model grid size, the creek width was adjusted to ensure a continuous creek section until it met with a river or another creek. Fine scale details for the main wetlands in the floodplain were embedded into the 30 m DEM using re-sampled 3 m LiDAR data. Bathymetry of the wetlands was estimated using a combination of LiDAR data (i.e. above their end of dry season water level) and field surveys of the submerged bathymetry. Wetlands were reproduced in the model using a set of rectangular grids ensuring the surface area was kept as close as possible to the actual wetland area.

We used Manning's roughness coefficients n to represent land surface resistance to the propagating flood wave. A surface roughness map was developed for the hydrodynamic domain with the same size grid as the hydrodynamic model using the Queensland land use map (Pitt et al., 2007). Initial roughness coefficients were estimated based on land use and then refined as a part of the model calibration process. Land use in the Tully-Murray floodplain is dominated by sugarcane plantations, interspersed with some grazing land. The next largest land use is banana farming, which is concentrated in the upstream reaches of the Tully floodplain. To produce a hydraulic roughness map, vegetation cover was classified as sugar cane, banana, grazing, cereal and urban. The water bodies were categorised as wetlands, creeks, and rivers. Sugarcane roughness is very dependent on the cane growth stage at the time of flooding (i.e. a fallow field can create a flow path while a fully mature cane field can act as a strong impediment to flow). River flow records show that most of the overbank events occur between January and March (Wallace et al., 2009), when cane fields are generally fully covered by plants, so a high roughness value was adopted for cane areas.

3. HYDRODYNAMIC MODELLING

3.1. Model configuration

The hydrodynamic model was configured for the combined Tully and Murray floodplains including estuaries at the downstream end. The computational domain was 720 km^2 (30 × 24 km) covering the entire floodplain

(Figure 1), which is 32% of the total catchment area. Inputs to the model were land elevation, surface roughness and water sources. Model boundaries include inflows through the Tully and Murray Rivers, and through 4 creeks. At the downstream, seaward boundary water levels equal to the Mean High Water Spring (MHWS) tide were used. The downstream boundaries were set sufficiently distant from the floodplain so that boundary effects (if any) were insignificant on floodplain flows. The inflows at the intersection of streams and hydrodynamic model boundaries were estimated using a calibrated rainfall-runoff model. The detail of rainfall-runoff modelling can be found in Karim et al. (2012). The upstream boundaries were set well above the floodplain to capture and define the upper catchment flows onto the floodplain.

We used a two-dimensional floodplain hydrodynamic model (MIKE 21; DHI, 2008) to simulate flood wave propagation and to estimate flow velocity across the floodplain. The MIKE 21 model is fully dynamic and is based on the depth-averaged Saint-Venant equations to describe the evolution of water levels, and two Cartesian velocity components. The model produces grid-based water level and velocity components in two horizontal axes (the x and y) over the entire computational period.

3.2. Simulations

Water sources on the floodplain include locally generated runoff and inflows from the upper catchments. We estimated local runoff as well inflows through the stream using the previously calibrated NAM rainfall-runoff model (Karim et al., 2012). The Tully and Murray catchments were divided into a number of sub-catchments based on land topography. Sub-catchment boundaries and stream networks were generated using ArcGIS Hydro Tools. The area above the hydrodynamic model domain was divided into 15 sub-catchments with an average area of 96 km^2, based on stream networks that carry upper catchment runoff to the floodplain. Runoff for individual sub-catchments was estimated separately and then propagated through sub-catchments further downstream. Runoff within the hydrodynamic domain was simulated using much smaller sub-catchments with an average area of 9.7 km^2. Sub-catchment boundaries and location of their outlets were obtained from previous hydrodynamic modelling studies of Karim et al. (2012). A total of 66 sub-catchments, 19 linked with the Tully River and 47 linked with the Murray River, were used in the floodplain. Modelled runoff was added to the hydrodynamic model as a point source at the outlet of each sub-catchment.

The hydrodynamic model domain was divided into 800,000 computational grids each 30 m by 30 m. The computational time increment was derived after satisfying numerical stability criteria. A time step of 4 sec was used as this produced a stable solution for floods with a return period of up to 50 year. Simulation of each flood event was carried out for 12 days to include the full flooding period of the largest flood. Computed time-varying water depth and velocity were recorded hourly at some selected points and two-hourly for all computational points. Spatial variation of flow velocity across the floodplain was extracted using MIKE 21 toolbox.

3.3. Calibration

The hydrodynamic model was calibrated for a recent flood in 2013 (22-28 January) which was about 2.3 times bigger than a mean annual flood by comparing observed and simulated stage heights and velocities. During the calibration process, floodplain topography was slightly modified at some locations to rectify model instability due to high velocities that occurred at sharp gradients. Surface roughness coefficients (Manning's n) were varied iteratively for the major land uses (sugarcane, banana and grazing) within the recommended range to attain close agreement between measured and simulated water heights in the river and on the floodplain. The calibrated n value for sugarcane is 0.20 which is the maximum among the land uses followed by urban area ($n = 0.12$), banana ($n = 0.10$) and grazing ($n = 0.09$).

3.4. Sensitivity of flood magnitude

Flood scenarios were estimated for 3 storm events for the ARI (average recurrence interval) of 1, 20 and 50 years. Design rainfalls for these storm events were estimated using rainfall frequency analyses for the Tully area. These estimates were based on the CRC-FORGE method (Durrant and Bowman, 2004). Predicted annual rainfall for 1, 20 and 50 ARI storm events were 408, 672 and 813 mm respectively. Temporal distributions of rainfall for the above events were obtained using 4-hourly temporal pattern hyetographs (Pilgrim et al., 2001). The critical storm duration for floods in the Tully-Murray floodplain is 72 hours. Combining this with temporal patterns hyetographs gave rainfall distributions for 72-hour storms that were divided into 18 periods each of 4 hours duration. Runoff values were then simulated using the previously calibrated NAM runoff model (Karim et al., 2012).

4. RESULTS AND DISCUSSION

4.1. Flow velocity

As part of model calibration a comparison between simulated and observed stage heights at the main catchment outlet (Euramo on the Tully River) was performed to ensure that model simulated speed of flood wave propagation that is representative of observed flood speed. Agreement between simulated and observed stage heights was reasonably good and the differences at any time were less than ±7%. The difference between simulated and observed peaks was less than 3%.

Figure 2 shows a comparison of simulated velocity at 53 locations where velocities were measured during the 2013 flood. Velocities between locations on floodplain vary in the range of 0 to 0.6 m/s. This is due to land slope and surface resistance. For example flow velocity at the entry point of sugarcane field is much higher than the velocity of flow leaving the sugarcane primarily due to resistance to flow by sugarcane. At 4 locations simulated velocities were found to be zero. Though there are large differences in point to point comparison overall the coefficient of determination is reasonable (r^2=0.76). Discrepancies are large for the low velocities while for large velocities simulated velocities are very close to observed velocity. Difficulties in measuring low flow in the field have contributed to the difference between simulated and observed velocity.

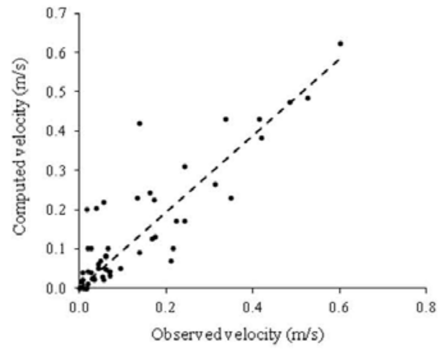

Figure 2 Comparison of measured versus simulated flow velocity on the floodplain between Tully and Murray Rivers. The dashed line (---) is the fitted linear regression line (V_{sim}=0.97 V_{obs} with r^2=0.76).

We investigated how flow velocity changes within in floodplain environment with respect to channelized flow. This gave us an indication of residence time in the floodplain. Figure 3 shows the simulated mean velocity at different locations along the Tully River starting from the upstream end of the hydrodynamic model boundary. It also shows discharge hydrographs at 2 locations (one at the upstream boundary and other at 15 km downstream). While discharge varies very little between locations, mean velocity can be greatly different based on location along the stream. At the upstream boundary flow is confined within the river bank, therefore velocity increases with increasing discharge (or stage height). The velocity curve at 5 km downstream indicates flow is still within riverbanks but river sections are greatly enlarged. The velocity curves at 10 and 15 km downstream indicate overbank flow conditions where velocity first increases with increasing discharge and then decreases due to spreading of flow overbank. The reason is that when it starts overflowing cross sectional area increases therefore mean velocity decreases. It can be seen that velocity at the lower end of the floodplain can be as small as ¼ of the upstream velocity. This indicates water residence time on the floodplain is much higher than it is for channelized flow.

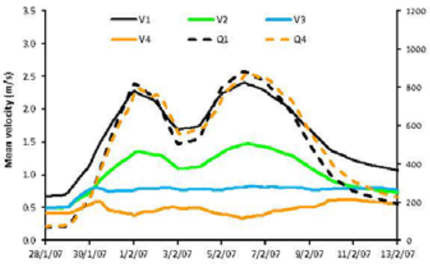

Figure 3 Spatial and temporal variation of flow velocity along the Tully River for the flood event in 2007 (V1, V2, V3 and V4 are mean velocity at 0, 5, 10 and 15 km downstream respectively from upstream boundary of the model, Q1 and Q4 are discharge at 0 and 15 km downstream respectively).

Figure 4 shows an example of relative magnitude of floodplain flow velocities with respect to in-stream flow at a mid-section on the floodplain. Apart from mean velocity, grid based velocity is smaller on the floodplain and it varies greatly between locations. It is noticed that the difference between floodplain velocities are more pronounced when in-stream velocity is relatively low. This is a situation when inundation depth is small and flood wave propagation is influenced more by surface resistance. Result shows velocities are high for locations close to river (e.g. FP1, FP2). With distance from the river velocity decreases due to surface resistance and it produces very small or zero velocity for a distant location (e.g. FP5). However this pattern could be different if topography and/or land use differs greatly.

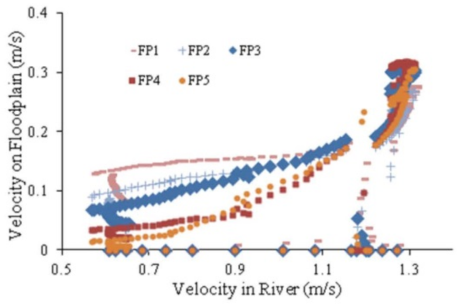

Figure 4 Comparison of velocity at different floodplain sites with respect to main channel (FP1, FP2, FP3, FP4 and FP5 are 5 locations on the floodplain, 1 indicates closest to and 5 indicates furthest from the Tully river).

4.2. Water residence time

The residence time for in-stream flow in the Tully catchment is less than a day from the upstream end of the floodplain before flood water reaches to GBR lagoon. The length of the Tully River for its floodplain part is approximately 38 km and if we assume a grid velocity of 1.3 m/s as wave celerity then flood water takes about 8 hours to reach the coast. If we consider mean velocity of 0.5 m/s then residence time is approximately 1 day. Residence time on the floodplain differs greatly between locations. However it is still within couple of days as water finds its way into cane drains and allows it to move faster. Figure 5 shows an example of inundation duration across the floodplain for a flood event of 1-year ARI. While much of floodplain shows less than 6 days of inundation for a 3 day storm event, some areas in the vicinity of the Murray River shows about 12 days of inundation. These are the places where residence time could be several days. Flat land slope is one of the reasons for this inundation behaviour. Large floods (e.g. ARI 20 and 50 years) produced longer duration of inundation but actual residence time on the floodplain is less due to high flow velocity for a large flood event. In this study we used 30 m DEM to represent land topography and thus couldn't reproduce levee banks along the river adequately. A finer resolution DEM (e.g. 5 m) could be useful to improve estimates of flow exchange between river and floodplain and thus residence time on the floodplain.

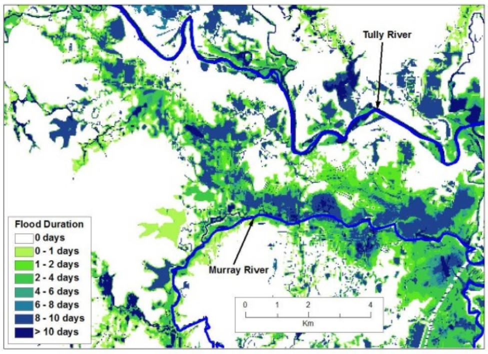

Figure 5 A typical example of spatial variation in floodplain inundation duration across the Tully-Murray floodplain for an annual average flood (ARI of 1-year).

5. CONCLUSIONS

In this study we have calibrated a two-dimensional floodplain hydrodynamic model to estimate water residence time on the floodplain. During a flood event, estimated residence time for in-bank flow is less than a day while it can be several days on the floodplain based on location. Land slope and land cover with high resistance to flow (e.g. sugar cane) are the major factors contributed to residence time. If time permits further validation of the model will be performed for a separate flood. We will use the calibrated model to estimate changes in residence time for different flow events and land uses. This information will be combined with decay rules to assess effectiveness of different vegetated system to remove pollutant from flood water before it reaches to a stream.

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