Government of **Western Australia**
Department of **Water**

Proposed guidelines for treating acidic drain water in the Avon catchment, Western Australia: adapting acid mine drainage treatment systems for saline acidic drains

Looking after all our water needs

Salinity and land use impacts series

Report no. SLUI 54
December 2009

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Adapting acid mine drainage treatment systems for saline acidic drains

Brad Degens

Department of Water

Salinity and land use impacts series

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Department of Water
168 St Georges Terrace
Perth Western Australia 6000
Telephone	+61 8 6364 7600
Facsimile	+61 8 6364 7601
www.water.wa.gov.au

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For more information about this report, contact

Dr Brad Degens
Salinity and Water Resource Recovery, Resource Management Division,
email: brad.degens@water.wa.gov.au

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Summary

Using deep drainage in the WA Wheatbelt to manage shallow saline groundwater in broad valley floors can lead to discharge of acidic saline water that may require treatment of the acidity and removal of metals and minor elements before disposal or further use. The low pH and high concentrations of metals and other elements can pose environmental risks where water is discharged to creeks, floodways and lakes. The sustainable use of deep drains in managing dryland salinity will depend on meeting the pressing need to develop low-cost, practical, and effective options to treat acidity.

The Avon Catchment Council funded an initiative to trial potential treatment options and to develop guidelines for designing and installing acidity treatment systems in Wheatbelt drains. The proposed guidelines are intended to provide landholders, contractors and others involved in managing acidic drain waters with sufficient information to design and install a treatment system consisting of a single option or combination of treatment options and to manage the operation of the system. The options have been adapted for use in Wheatbelt drains from an exhaustive range of international options developed to treat acid mine drainage and acidic drainage from acid sulfate soils.

The guidelines start with a brief review of the constituents that may be found in acidic saline drain waters, why these are of concern to waterways and infrastructure, and some treatment principles. Current understanding of acidity and its impacts in saline surface waters in the WA Wheatbelt is restricted to a few studies and requires further investigation. Nevertheless, expected impacts can be broadly drawn from research in fresh water systems elsewhere in the world.

Before beginning the design process it is necessary to collect or estimate information on acidity loads from drains. While the preferred approach is to obtain this information through monitoring water quality and flows from a drain, this is rarely obtainable at the outset so may need to be estimated. Factors controlling acidity loads such as flows and minor element concentrations are discussed, based on the outcomes of investigations carried out as part of the Engineering Evaluation Initiative. These can be used to guide estimates.

A step-by-step process is provided to assist with designing treatment systems for acidic saline water discharging from deep drains. Advice is given on deciding what water from a drain needs to be treated, to what quality and how to choose a suitable treatment option or combination of options. In the following sections, engineering and construction details for a range of individual treatment options adapted for use within drains or at the end of drains are presented, including designs to manage sludges produced by treatment. These focus on using neutralising materials such as lime-sand and alternative approaches using microbial-based treatment systems such as composting wetlands. While most options focus on passive treatment with low maintenance and water flowing under gravity, several treatment options involving pumping and dosing systems are discussed for situations where this may be possible. Where possible, case studies of pilot treatment options are presented to provide some insight into local design and performance. These case studies are based on on-ground evaluations conducted in drains at Doodlakine and Beacon between 2006 and 2008.

The treatment options that may be suitable for use in acidic saline drains include lime-sand beds, subsoil carbonate beds, in-drain composting beds and diversion wells. Lime-sand basins, composting wetlands, lime-sand reactors and hydrated lime dosing units are suited for treatment at drain ends. These options range in effectiveness from full treatment of a range of acidic waters to partial neutralisation, with costs for common drain water acidity levels varying from 10 cents per kL for some treatment options to more than 90 cents per kL, although all costs exclude managing sludges and/or decommissioning treatment systems.

The final section presents suggestions for creating an integrated treatment setting using various combinations of treatment units.

1 Introduction

Dryland salinity due to rising saline groundwater affects around 1 million hectares of the WA Wheatbelt and threatens up to 3 million hectares over the next 20–200 years (McFarlane et al. 2004). Increasing water use by vegetation has been proposed as the main long-term strategy for dealing with rising groundwater (Clarke et al. 2002). This, however, depends on widespread changes in land use before an improvement in valley floor salinity occurs (George et al. 1999; George et al. 2002).

There has been increasing farmer interest (particularly in the eastern Wheatbelt) in using deep open drains to manage shallow saline watertables in valley floors (Kingwell & Cook 2007). This interest has probably arisen because of mixed confidence in alternative options for managing salinity in broad valley floors. Concerns about the rate of spread of salinity and the perceived failure of other options to manage rising watertables in broad valley floors have added to interest in deep open drainage (Ali & Coles 2002).

More than 90 000 kilometres of salinity earthworks have already been constructed in parts of the Wheatbelt (ABS 2002, of which at least 4000 km are deep (at least 2 m) open groundwater drains. There is increasing interest in expanding the use and scale of drains to deal with the expanding threat of salinity in valley floors (Ruprecht et al. 2004). However, widespread use of deep open drains is widely acknowledged as requiring careful planning and management of the water discharged from these drains (Ali & Coles 2002; Ruprecht et al. 2004, Stewart et al. 2009).

The sustainable use of deep drains will require safe, practical, and cost-effective options for managing the disposal of saline, often acidic, water. Deep open drains can intercept shallow acidic groundwater and accelerate its discharge to surface environments (Shand & Degens 2008). Without treatment and management, such water poses a threat to aquatic and terrestrial ecosystems (Degens et al. 2008a). Managing the disposal of water from deep open drains will therefore require treatment of acidity in some drain waters.

Acidic drain waters in the Wheatbelt are very similar to acidic waters draining from abandoned mines (usually termed acid mine drainage or AMD) and drainage from acid sulfate soils (ASS drainage), only more saline (Degens et al. 2008b). Worldwide, there is a range of chemically and biologically based technologies developed for treating AMD and, to a lesser extent, ASS drainage (for an overview see Gazea et al. 1996, Skousen et al. 1998 or Taylor et al. 2005). These treatment systems are broadly categorised as being either active or passive.

Active treatments commonly require continual input of (bio)chemicals, energy and maintenance and are often intended for short-treatment periods or large acidity loads. By contrast, passive treatments involve low maintenance and energy inputs after establishment, are frequently gravity-driven (i.e. do not include pumping systems) and are intended for long-term operation with generally smaller acidity loads (Younger et al. 2002; Taylor et al. 2005).

A review of treatment options highlighted a number of potential materials and technologies that held some promise for treating acidic waters in WA drains (Douglas & Degens 2006), though many were not yet in a position to be used immediately in the Wheatbelt at the time of preparing these guidelines. Further research and development of some of these options may see them as providing additional treatment options. In view of this, these guidelines focus on treatment systems that rely on either the use of locally available neutralising materials and/or the use of sulfate reduction by microbes to generate alkalinity.

These guidelines provide advice on treatment options for **net** acidic waters discharged by deep open drains in the WA Wheatbelt where these are constructed in low-gradient valley floors.

The design principles for treatment systems outlined are adapted from those used for AMD as detailed by Hedin et al. (1994), Younger et al. (2002), PIRAMID Consortium (2003) and Watzlaf et al. (2004). A limited selection of treatment options is presented to meet the following criteria:

- being most suited to the water chemistry commonly occurring in Wheatbelt drains
- having the greatest likelihood of compatibility with low-gradient Wheatbelt drains
- being practicable
- using materials that are easily handled and most likely to be locally available.

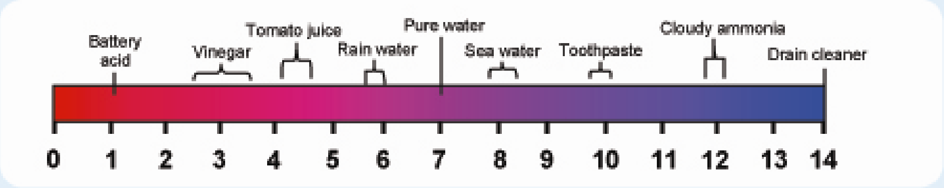
These guidelines begin with a review of the constituents of acidic saline drain waters, why these are of concern and where acidity treatment might be considered in relation to other potential management approaches. Users are then taken through treatment principles before the range of available treatment options is presented. These include the pros and cons of each in relation to WA Wheatbelt conditions and a decision-making tree to assist users in designing their own system.

Case studies of pilot treatment units (including construction and engineering details) are outlined to give guideline users some insight into the local designs and performances. Many of the adapted designs are based on results of on-ground trial evaluations of pilot treatment trials carried out in the Avon catchment between 2006 and 2008 under an initiative supported by funding from the Avon Catchment Council. At the end are suggestions for creating an integrated treatment setting using various combinations of treatment processes.

A final point is that these guidelines focus on interventions to treat acidic drain water within the confines of a drain or contained receiving system and do not cover options for managing acidic drain waters that do not involve specific treatment *per se*.

pH and why it is important to water

pH is a measure of the acid or alkaline property of water commonly ranging from 0 to 14, where a pH<7 is acidic, pH 7 is neutral and pH>7 is alkaline. pH reflects the balance of hydrogen ions (acid) and hydroxide ions (alkaline) which make up part of the acidity or alkalinity of water that is often due to other things dissolved in the water such as metals, bicarbonate or carbon dioxide gas.



Low pH or acidic water is corrosive and harmful to aquatic life and can contain metals that can also affect aquatic life.

2 Background

2.1 Origins of acidity

Most shallow and deeper saline groundwater in the eastern WA Wheatbelt is acidic (pH <5.5) and can contain elevated concentrations of dissolved metals and other elements, particularly those associated with broad valleys (Shand & Degens 2008; Lillicrap & George 2008). This groundwater was known to be acidic before active management (tree planting, drainage and pumping) was instituted to control groundwater rise and was probably acidic before the landscape was cleared. In some Wheatbelt valleys, groundwater has pH 3–3.5 from the watertable to bedrock (often >20 m below the surface).

The origins of the acidity are not clear, although the most accepted explanation is that iron-rich water formed in the topsoil horizons of landscapes during waterlogged conditions and percolated to the groundwater leaving the alkalinity behind in the form of nodules of calcium carbonate or limestone in the soils (Gray & Lillicrap 2008). This process is aided by trees and shrubs growing in these soils and is thought to have occurred over tens of thousands of years (Lillicrap & George 2008). The iron-rich water is believed to have acidified soils immediately beneath the carbonate layers resulting in groundwater that is acidic but still containing some unreacted iron (Gray & Lillicrap 2008). The exact mechanism in the soils in the plant root zone is still the subject of research, but may explain observations that alkaline (calcareous or carbonate-rich) soils often overlie areas of low pH groundwater (Lillicrap & George 2008).

The construction of deep open drains (or groundwater pumping) can intercept and increase the mobility of this water in the landscape (Shand & Degens 2008). During movement through soils to drains, through drain sediments and down the length of drains a complex series of chemical reactions can modify the composition of the groundwater (Shand & Degens 2008). Although acidic before discharge into a drain, the pH can further fall due to the reaction of dissolved iron with oxygen on exposure of the water to air.

2.2 General composition of acidic saline drain water

Acidic drain waters in the WA Wheatbelt are saline, commonly ranging from 30 000 mg total dissolved salts (TDS)/L to 80 000 mg TDS/L (Degens et al. 2008b; Fig. 1). During summer the salinity can exceed 100 000 mg TDS /L, particularly in trickle flows. The pH in flowing water can be as low as 1.8 in summer–autumn and up to 4 in winter–spring. After rain, the pH of some drain water can increase to more than 5 (Section 4.2). Few drain waters are consistently pH 4–6 (Fig. 1).

Surveys have found that more than half of the acidic drains contain flowing water with high concentrations of soluble iron (Fe, at >25 mg/L) and aluminium (Al, commonly at >80 mg/L) (Degens et al. 2008b). Aluminium is frequently the dominant dissolved metal: more than 70% of acidic drain samples contained aluminium in higher concentrations than iron – generally more than double. Manganese (Mn) can also be present, but generally does not exceed concentrations of 6 mg/L in flowing water. Dissolved minor elements such as lead (Pb), copper (Cu), zinc (Zn), nickel (Ni), uranium (U) and rare earth elements (lanthanum, cerium) can also occur in high concentrations (>0.5 mg/L) in some, but not all, acidic drain waters (Degens et al. 2008b). Dissolved cadmium (Cd), arsenic (As) and selenium (Se) can also occur, though are less common (see Appendix A for the distinction between metals and minor elements).

The concentrations of nutrients in drain water are often very low, with nitrate and ammonia generally less than 2 mg/L and 3 mg/L respectively. Soluble phosphate is generally less than 0.2 mg/L and frequently cannot be detected (Shand & Degens 2008). These nutrients can be significantly concentrated, along with salts, in trickle flows occurring during summer. In contrast to nutrients, concentrations of dissolved organic carbon can be quite high, with many acidic drains containing water with more than 15 mg dissolved organic carbon/L (Shand & Degens 2008). Despite being open to the atmosphere, drain waters are often poorly oxygenated.

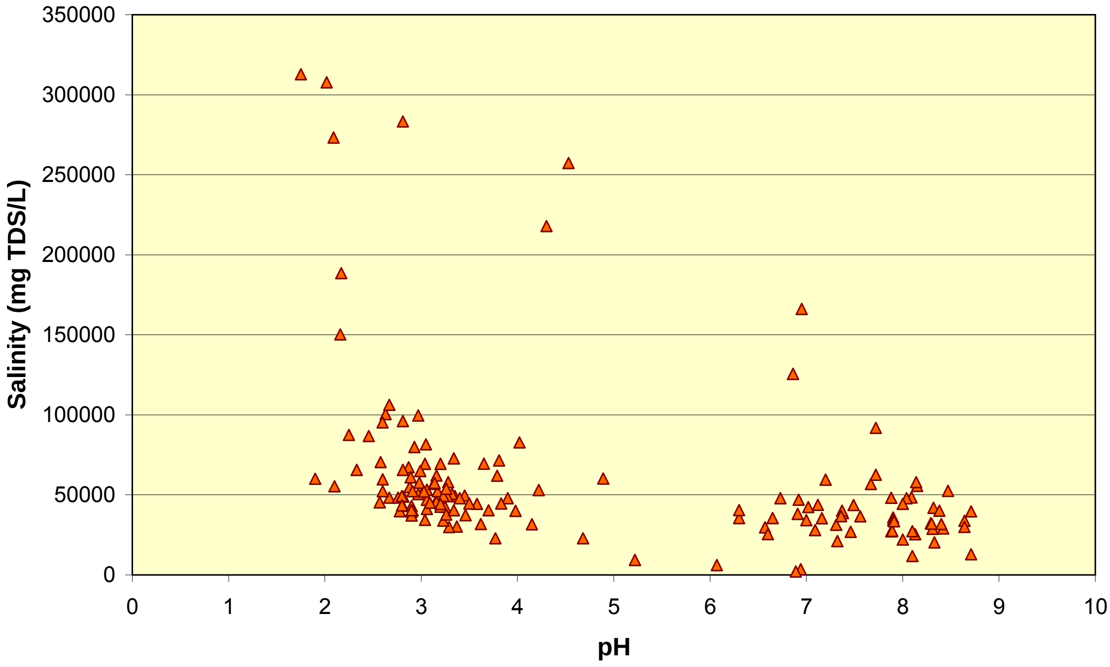


Figure 1 The distribution of drain water salinity in relation to pH for 158 drain water samples in the Avon and Yarra Yarra basins (59 drainage systems)

2.3 Why acidic water may be a problem

The main risks of untreated acidic water are defined by the hazards to lakes and waterways and the pathways by which these might be expressed. Aquatic environments in this context include creeks, floodways and lakes as well as the riparian ecosystems associated with these. Hazards are categorised in two groups: (1) acidity and (2) mobilisation of metals and minor elements, and their potential impacts depend on how, when and how much acidic water reaches surface environments.

Acidic water can pose a hazard to:

- infrastructure in waterways, such as bridges and culverts; this is discussed at the end of this section
- aquatic life in often saline lakes, creeks and floodplains
- riparian vegetation, soils and sediments in waterways and lakes
- human health, where water accumulates in lakes used for recreation.

In addition to salinity and waterlogging impacts, acidification of waterways and lakes is likely to reduce the biodiversity in these areas. Acidification of flowing water will directly and immediately affect organisms (including algae, macro-invertebrates and aquatic plants) living in the water and on the beds of lakes and waterways. Waterways with acidic water (due to discharge of acidic drain water) have less biodiversity than similar alkaline waterways (Stewart et al*.* 2009). Likewise, acidic saline lakes have less aquatic diversity than alkaline saline lakes (Halse 2004). Acidification of sediments is also likely to affect the capacity for organisms to survive in these sediments and recolonise them during fresh-flow events, known to be important in Wheatbelt waterway ecology (Davis et al*.* 2003; Sim et al*.* 2006a, b).

In contrast to aquatic impacts, acidity impacts on riparian ecosystems are likely to be less obvious and more delayed. The possible impacts on riparian vegetation are largely unknown, but experiences with ecosystem acidification (by acid rain) elsewhere in the world indicate that significant changes in diversity can occur when soils and sediments become more acidic (Driscoll et al. 2007). These effects are mostly caused by direct toxic effects of aluminium on roots in acidified soils and impacts on the availability of nutrients, particularly cations such as magnesium, calcium and potassium (Driscoll et al. 2007; Vance 2006). The latter may be less critical in saline ecosystems where the concentrations of these ions are unlikely to be limiting because of high inputs from saline groundwater.

The aluminium and iron in acidic water can have short-term impacts on aquatic organisms in creeks, floodways and lakes where drainage water directly discharges (and evaporates) or is 'flushed' by catchment runoff. In freshwater ecosystems, this effect is often more than just the effects of low pH and includes the effects of dissolved metals (Sutcliff & Hildrew 1989; Gensemer & Playle 1999). The metals in acidic water are indicated in measures of acidity (see Section 3.2.1) that indicate more about the toxic effects of acidic waters beyond the effects of pH. The same is expected in saline environments.

For dissolved aluminium, the impacts on aquatic organisms are largely due to the formation of toxic ion-complexes (Gensemer & Playle 1999), mostly when acidic waters are partially neutralised by alkaline waters; for example, during mixing with flood waters (Fig. 2; Degens et al. 2008a). However, high concentrations of sulfate, calcium and dissolved organic carbon in many acidic saline waters in the Wheatbelt (Shand & Degens, 2008) are commonly known to reduce the toxic effects of the metal (Gensemer & Playle 1999). Fine sludge precipitating from water after neutralisation may physically affect aquatic organisms, such as insects, by clogging breathing organs and mouthparts, although there is no evidence to support this (Gensemer & Playle 1999).

In contrast, dissolved iron can generate acidity in downstream environments as a result of oxidation by air and form iron sludges or precipitates (Fig. 3) that can smother any organisms in the water and on the bottom of lakes and waterways (Phippen et al. 2008). Oxidation of iron can cause the pH of water to fall well downstream of where water discharges, even when the pH first appears to be greater than 6. Some of the iron-rich sludges forming in acidic waters can also trap acidity in minerals (such as schwertmannite, Sullivan & Bush 2004) that can continue to lower the pH of water for many decades (see Section 3.2.2 Acidity in soils and sediments). Formation of sludges after iron oxidation may have large impacts on mats of algae growing on the bottom of lakes and creeks and on submerged aquatic plants that are important parts of saline aquatic ecosystems (Davis et al. 2003; Sim et al. 2006 a, b). Furthermore, iron sludges can readily be resuspended into lake water during windy conditions, creating very turbid water likely to directly affect all aquatic organisms (Phippen et al. 2008).

Figure 2 Formation of cream, meringue-like froth characteristic of aluminium minerals precipitating in alkaline floodwaters mixing with acidic drain waters (reaching pH 4.6)

Hazards associated with minor elements in acidic water include the transport and accumulation of elements such as arsenic, cadmium, copper, lead, nickel, uranium, and selenium in surface environments. These dissolved minor elements may cause immediate toxic effects on aquatic organisms in waterways and lakes; for example, reduction in growth, reproduction or even mortality (Wang 1987; Rainbow 1996). There may also be longer-term risks to bird life if the metals accumulate in lakes (e.g. Diamond 1989) and begin to bioaccumulate within food chains. This is known to occur with elements such as lead, cadmium, uranium, arsenic and selenium in some aquatic ecosystems elsewhere in the world when concentrations of the elements entering the systems have increased (e.g. Maret et al. 2003; Gosavi et al. 2004; Simmons & Wallschlager 2005).

Figure 3 Sediments and organic debris in a creek smothered by iron precipitates formed from acidic drain water

Bioaccumulation in lakes involves the concentration of metals in the food chain as higher organisms consume algae, water insects and plants. High metal concentrations accumulate in organisms that eat other organisms that have accumulated metals either directly or indirectly from lake waters. Accumulation can occur to the extent that the health of animals at the top of the food chain (such as aquatic birds) is affected. In some ecosystems, this can extend to genetic damage to resident bird populations (e.g. Pastor et al. 2004). Factors that can affect bioaccumulation include salinity, a key factor in saline waters as well as the concentrations of dissolved organic matter, sulfate and calcium (Wang 1987; Hare 1992).

A general principle is that the timing of minor metal solubility plays a major role in whether there is a risk of bioaccumulation, with a higher risk when the highest concentrations of dissolved minor metals are during times when aquatic organisms emerge, grow and reproduce (Hare 1992; USEPA 2007). In ephemeral lakes, this is likely to be when consumers such as birds arrive and reproduce.

In the Wheatbelt, minor element concentrations in lake waters are likely to depend on stores that have accumulated in their shallow muds between flood-filling events and whether these metals are released in any significant concentrations during or after filling. Many ‘naturally’ acidic Wheatbelt saline lakes may have concentrations of dissolved minor metals and other elements in the drying phases similar to those in lakes from which the acidic saline drain waters have been discharged (Degens et al. 2008a). However, the metal concentrations can be slightly higher in the uppermost few millimetres of muds in some lakes receiving acidic drainage (Degens et al. 2008a) and these muds will probably play a key role in what metals are released when the lakes are filled by floodwaters.

The acidity hazard of acidic water extends beyond aquatic ecosystems to include corrosion hazards for waterway infrastructure such as culverts, bridge work and water supply pipes. While salinity is widely known to corrode concrete and steel infrastructure, additional exposure to acidic water may greatly accelerate this (ABCB 2004). Acidic drainage from acid sulfate soils has been shown to corrode concrete infrastructure in waterways of coastal environments (Sammut 2000; Morgan 2005) and acidic saline waters would have similar effects on any similar infrastructure in the Wheatbelt. This risk particularly applies to infrastructure not installed to standards that would resist acid and salt attack (see Standards Australia for standards on concrete and steel infrastructure in saline and/or acidic environments).

2.4 Treatment in the context of acidic drain water management

There is a range of potential management options of which treatment is one option. Deciding not to construct deep drains could be considered an option that avoids generating acidic drainage waters and the need to manage them, although this may conflict with other objectives of managing shallow saline watertables. *These* guidelines focus on purpose-built interventions to specifically treat acidic drain waters within the confines of a drain or contained receiving systems. However, there are various alternative options that might be suitable for some sites that do not involve specific treatment intervention or involve use of natural treatment capacity. These are outlined below, along with options that involve purpose-built treatment systems.

- **Drain design to minimise acidic water export** – There is some evidence that, where acidic drainage water is likely, drains can be designed to minimise interception of the acidic groundwater, while achieving control over shallow watertable levels (Cox & Tetlow in press). This option may effectively enable management by avoiding unnecessary interception and transport of acidic groundwater.
- **Drain design to intercept alkaline groundwater** – Although there is some uncertainty about the likely success of this approach, drains could potentially be placed to intercept alkaline groundwater in the landscape (e.g. upper reaches of some catchments or margins of valleys) with this water neutralising the acidic groundwater intercepted, generally in the lower reaches of the drainage system before discharge. This occurred to some extent in drains at Pithara (Cox in press).
- **Disposal to acidic lakes with no treatment or containment** – This option involves disposal to lakes already acidified by regional groundwater. Assessment of suitability would include evaluating risks to the riparian vegetation associated with the site, the
- Risk of downstream export of acidity and minor metals (dissolved and particulate), and the fate of these in lakes and waterways downstream, and potential neutralisation by floodwaters.

- **Disposal to lakes or waterways with treatment by landscape alkalinity** – Water might be contained within (for example) lakes with a natural capacity to neutralise acidic water or with a likelihood of neutralisation by floodwaters. Natural neutralising capacity in lakes can be due to carbonate materials stored in lake sediments or by microbial treatment processes in muds at the bottom of the lakes (principally sulfate reduction, see Sections 3.1 & 6.3.1). Risk assessment would include evaluating risks to the riparian vegetation associated with the site, the risks of downstream export of acidity and minor metals (dissolved and particulate), the fate of these in lakes and waterways downstream and risks to aquatic ecosystems. Where neutralisation depends on floodwater alkalinity it will be necessary to establish the frequency, effectiveness and fate of neutralisation products.
- **Disposal to lakes with augmented treatment capacity** – This option involves disposal to natural sites but taking action to bolster natural processes to achieve treatment. These actions might include adding carbonates from surrounding dunes or soils (see also Section 6.2), introducing lime-sand, or adding organic matter or nutrients (stimulating algal growth) to enhance microbial treatment processes such as sulfate reduction in sediments (see Sections 3.1 & 6.3.1). Information needed to assess risks includes rates of neutralisation in the receiving site (based on adding neutralising materials or microbial activities), the long-term fate of neutralised materials, the risk of inland ASS-like conditions developing and the risk of acidity being flushed downstream during flow events (including floods).
- **Disposal without treatment to constructed evaporation basins or natural basins (and possibly alternate uses)** – Water may be disposed of into purpose-built or naturally well-contained sites with no treatment other than evaporation to reduce the volume and concentrate acidity on-site. It may also be possible to use untreated acidic water in solar ponds before final disposal. Solar ponds involve constructing deep ponds that allow collection and storage of heat energy in hypersaline water for direct use (e.g. drying ovens) or generating electricity (e.g. using heat-exchange systems to drive a Stirling cycle engine coupled with an AC generator; Ahmed et al. 2000). Information needed to assess risks includes rates of off-site leakage and risk of structure failure (particularly during floods). For natural sites, there would also need to be an assessment of risks to any riparian vegetation, development of inland ASS-like conditions and the impacts of groundwater discharge on any adjoining aquatic and riparian ecosystems.
- **Treatment with containment** – Water is treated and contained for further use. The assessment of risks principally focuses on salinity issues (e.g. mounding of local groundwater and associated impacts).
- **Treatment with discharge** – Water is treated and discharged to natural environments. The assessment of risks principally focuses on salinity issues for the receiving environment (e.g. changes in timing of water volumes and salt loading).

Deciding whether an option might be suitable to manage acidic water depends on the level of case-specific risks and what risk reduction management is carried out. Naturally, treatment to neutralise acidity and remove all dissolved metals and other elements achieves complete minimisation of risks associated with acidity and dissolved elements. The minor risks in this case would include the implications of the treatment system being effective less than 100% of the time throughout the life of the drain discharge.

Note that these options only focus on managing acidity and dissolved metals. Managing salt and water volumes or other drainage-associated issues (such as nutrient and sediment export) would also need to be considered. Options such as containment with no treatment may deal with all issues, whereas options such as treatment with discharge are not likely to mitigate salt and water volume risks.

3 Acidic water treatment principles

3.1 Basic chemistry of acidic water treatment

Treatment of acidic water requires dealing with both the acidity and dissolved metals (see Appendix A for the distinction between metals and minor elements).

More precisely, this involves raising pH to neutralise acidity **and** remove metals dissolved as a result of the low pH. These metals commonly include iron, aluminium and to a lesser extent manganese in saline acidic drain water (as indicated in Section 2.2), but can also include elements such as copper, nickel, lead, zinc and uranium and occasionally cadmium, arsenic and selenium. The metals and other elements do not all behave similarly with respect to the pH, salinity and oxygen status of water and, if present, some will require more effort to remove than others (for example, zinc, cadmium, arsenic and nickel; PIRAMID Consortium 2003, Gazea et al*.* 1996).

It is important to note that whether metals and other elements need to be removed depends on whether the concentrations are acceptable for the proposed water use (e.g. disposal, discharge to the environment, desalination or aquaculture; as outlined in Section 3.2.3).

Treating acidic drain water commonly involves raising pH by neutralisation, after which dissolved trace metals and other elements are commonly removed from water in sludges. These two treatment steps are dictated primarily by the chemistry of the dissolved metals, where pH is the main controlling factor in many cases (Younger et al. 2002).

Neutralising acidity requires adding alkalinity in chemical form as a carbonate or hydroxide material or producing alkalinity as a by-product of microbiological activity in an anaerobic (zero-oxygen) composting system (Younger et al*.* 2002). Water pH strongly influences the solubility and behaviour of many elements (particularly dissolved metals) and raising this allows chemical reactions in which the elements precipitate and can be removed as solids (or sludges). The main processes that lead to minor element removal include:

- precipitation as a solid (often observed as a fine sludge)
- adsorption onto sludges formed in the water after neutralisation
- adsorption onto clays, iron oxides and organic matter suspended in the water or present in underlying soils in contact with the water
- co-precipitation with other elements (i.e. trapped within iron or aluminium sludges as these form after neutralisation).

Iron and aluminium are the main metals in the precipitates formed during neutralisation of acidic Wheatbelt drain water. Soluble aluminium can be removed directly by raising pH and generally forms gelatinous, amorphous white precipitates (which can be aluminium hydroxides or hydroxy-sulfates). In contrast, removing dissolved iron by precipitation requires both increasing the pH and aerating the water. It is necessary first to raise pH to speed up the rate at which dissolved iron reacts with oxygen as this reaction is often very slow in water with pH less than 4 (common in most Wheatbelt drain water), although specialised acid-tolerant bacteria (e.g. *Thiobacillus ferroxidans*) can greatly increase the reaction rate (Kirby et al*.* 1999; Baker & Banfield 2003).

In microbiological treatment systems such as composting beds (see later Sections 6.3 & 7.2), dissolved iron is precipitated as a fine black ooze smelling faintly of rotten-egg gas (hydrogen sulfide) and eventually becomes a fine-grained pyrite mineral (one form of a mineral commonly known in rocks as iron pyrite or 'fools gold'). Metals such as nickel, lead and zinc are precipitated in a similar way in this black ooze.

Both aluminium- and iron-based precipitates can interact with dissolved minor elements in water during neutralisation and remove them from solution. The extent to which this occurs can be highly dependent on the nature of individual minor elements and is not outlined here (see Langmuir 1997; Langmuir et al. 2004; Espana 2007). An important point to note is that the precipitates formed during neutralisation are the early stages of mineral formation and can be unstable (Cravotta & Trahan 1999). Many will often rapidly redissolve in water if pH or oxygen concentration (in the case of iron precipitates) decrease, shedding any elements that have been adsorbed or co-precipitated (McDonald et al. 2006).

Effective removal of sludges from treated waters is a key part of treatment. As indicated above, sludges commonly trap or adsorb the dissolved metals and other minor elements in acidic water and will need to be separated and stored in a place that prevents them from re-entering the treated waters again.

3.2 Acidity and target treatment values

Treatment of acidic drainage water requires consideration of some of the chemical aspects of the water that are somewhat different from strong acid–base chemistry. Understanding these differences provides the basis for setting targets for water treatment (covered at the end of this section).

3.2.1 pH, acidity and alkalinity

The acidity of water is commonly discussed in terms of pH values but this may under-estimate the acidic properties of water in natural environments. pH is a measure of the concentration of hydrogen ions (H^+) in water and may only represent the 'tip' of the acidity 'iceberg' in acidic Wheatbelt drain water. The total acidity *includes* the amounts of dissolved 'free' metals that contain a type of acidity not detected by just measuring pH. The 'free' metals commonly include aluminium, iron and manganese, with minor contributions from some minor metals (e.g. copper or zinc). Measuring acidity reveals significant additional dimensions to the quality of acidic waters that have significant implications for their behaviour and aquatic toxicity (see Section 2.3).

In low pH water (less than about pH 5.5) concentrations of 'free' metal ions are usually much higher than in neutral and alkaline water and effectively behave as 'hidden' or 'potential' acidity not reflected by pH. The extent of this acidity only becomes apparent when trying to raise the pH or when the water mixes with higher pH water or mud. Then, the metal ions begin to hydrolyse and precipitate as sludges, releasing hydrogen ions which have the effect of consuming alkalinity in the higher pH water and mud. When the alkalinity is exhausted or cannot react fast enough (particularly in mud), soil and water pH can decrease sharply. The capacity to consume alkalinity is used as a measure of the total acidity in acidic water. It is also helpful to consider total acidity as a measure of the tendency of water to buffer against *a rise* in pH.

The common method for describing the amount of acidity in water, sediments and soil is to express it in terms of the amount of pure calcium carbonate ($CaCO_3$) needed to bring the pH to near neutral. For water, the standard way of measuring this is by quantifying the mass of a strong neutralising material such as sodium hydroxide (also known as caustic soda) needed to raise water to pH 8.3 (Kirby & Cravotta 2005; see Appendix B). The result is described as an **equivalent** amount of calcium carbonate to neutralise the water per litre (e.g. grams $CaCO_3$ equivalent/L or grams $CaCO_3$/L). An outline of methods for measuring acidity in the field is given in Appendix B and the range of acidity levels likely to be encountered in Wheatbelt drains is illustrated in the information box – Acidities of WA saline drain waters).

Acidities of WA saline drain waters

Acidities likely to be encountered in acidic saline drain water in the WA Wheatbelt generally range 0.1–1.6 grams calcium carbonate ($CaCO_3$) per litre (Fig. 4) or, in larger volumes, 0.1–1 kg $CaCO_3$ per kL water. The median acidity of the drains in this survey was 0.6 g CaCO3/L. Wheatbelt drain water rarely contains any unreacted alkalinity, though the groundwater feeding the drains can sometimes contain some dissolved alkalinity.

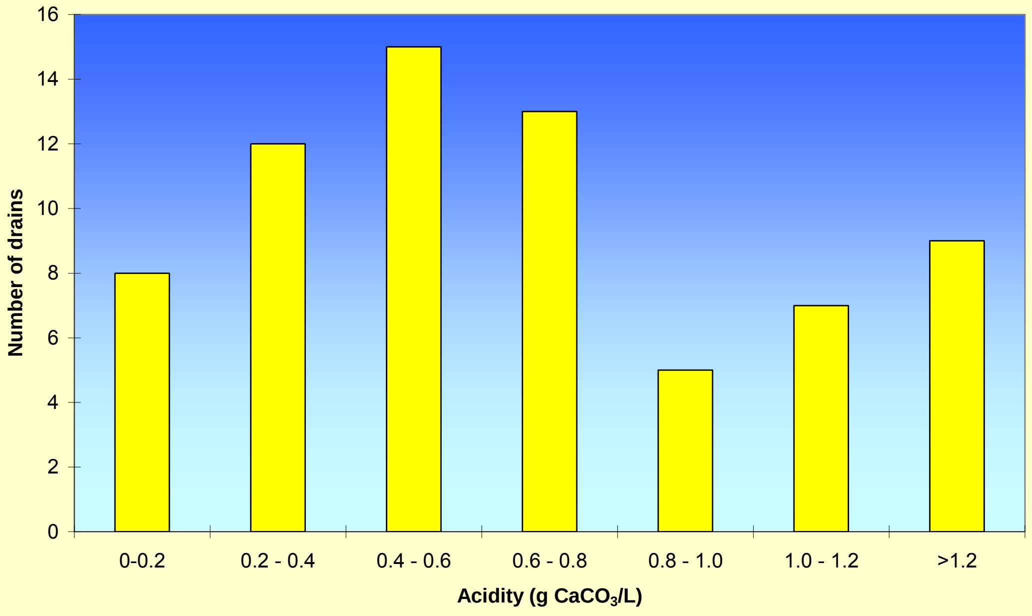


Figure 4 Distribution of acidity concentrations for flowing drains with water pH less than 4.5 (69 sites)

A useful point to note is that the amounts of calcium carbonate acidity are equivalent to the amounts of sulfuric acid (e.g. an acidity of 1 g calcium carbonate per litre is equivalent to 1 gram of 100% sulfuric acid per litre).

The analysis end-point of pH 8.3 is the upper limit at which water is considered to contain no significant residual acidity of any form, particularly that due to dissolved carbon dioxide gas, which forms carbonic acid (Langmuir 1997). Collectively, the acidity due to the dissolved metals is often termed 'metal' acidity (Langmuir 1997; Kirby & Cravotta 2005) to distinguish this from acidity due to dissolved carbon dioxide gas or organic acids (the latter occurring in organic rich environments).

The alkalinity of water can be thought of as reflecting the strength of buffering against a fall in pH if mixed with water containing acidity or coming in contact with acidic sediments. This is in effect the opposite of acidity. Water with high pH can be referred to as alkaline water and can contain chemical alkalinity. This is mostly present as dissolved bicarbonate when pH is between 4.5 and 8.3 and mostly carbonate when pH is greater than 8.3. Alkalinity is measured in the reverse way to acidity by determining how much of a strong acid such as hydrochloric acid is needed to lower pH to the end-point of 4.5. As with acidity, the degree of alkalinity is also expressed as an **equivalent** amount of calcium carbonate per litre of water, though this refers to the amount that is **preventing** pH from falling to 4.5.

Some water can have hidden or potential acidity at the same time as having some alkalinity (see information box – Net acidity). The net effect of these on how water will behave if exposed to air can be determined as the difference between the acidity and alkalinity. If water has potential acidity in excess of alkalinity, the water will become acidic in streams and lakes in the long term. Conversely, if water has potential acidity in deficit of alkalinity, and if no other reactions occur from sediments, the water will become alkaline in streams and lakes in the long term.

Net acidity

Some water has acidity (as dissolved metals) and alkalinity (mostly as bicarbonate) at the same time. This frequently occurs where iron is present in poorly aerated drain water or groundwater and sometimes in drain water just after treatment. Such water contains alkalinity in conjunction with acidity either because poor aeration allows iron to remain dissolved or the chemical reactions are slow and have not reached an end-point.

Naturally, acidity and alkalinity have the potential to cancel each other out. To work out whether cancelling occurs it can be useful to calculate **net acidity** (Kirby & Cravotta 2005; also see Glossary).

Net acidity = acidity – alkalinity

Most drain water in the eastern Wheatbelt is net acidic (Shand & Degens 2008) but some groundwater can contain alkalinity before discharging into drains.

3.2.2 Acidity in soils and sediments

Acidity stored in soils and muds in drains and lakes needs to be accounted for when considering treating a site where acidic water has been present for some time and the underlying sediments and soils have become acidic. Discharge of acidic drainage to lakes and floodways in the Wheatbelt can result in gradually acidifying sediments in these sites (Degens et al. 2008a). The acidity stored in soils and sediments can be more than 100 times that held in overlying water (Degens et al. 2008a) and will counteract attempts to treat surface water acidity as the acidity slowly 'leaks' back into the water.

The layers of acidic mud within drains can also add to the acidity load after rainfall events. This is particularly likely when flows increase sufficiently to move mud in the channels.

Sediment and soil acidity can be considered (in a way) similar to that of water acidity, although there is a wider range of 'reactivity'. Acidity in sediments is generally described as:

- soluble
- exchangeable
- insoluble.

Soluble acidity is generally the most reactive of these (reacting over minutes to hours), whereas exchangeable acidity is less reactive (reacting over minutes to days) and insoluble acidity is least reactive (hours to decades, sometimes centuries; see McElnea et al. 2002 for more detail). Insoluble acidity includes acidic minerals forming in soils and sediments such as jarosite, schwertmannite and basaluminite (McElnea et al. 2002). The amounts of acidity are expressed in terms of the mass of pure lime-sand to raise the pH of a tonne of the acidic materials to 6.5 (McElnea et al. 2002). Each component of acidity can be estimated by sampling and analysis by methods developed for Acid Sulfate Soils (Ahern et al. 2004).

Details on soil treatment are not covered in this guideline. Broad guidance for assessing and treating extreme soil and sediment acidity (for aquatic ecosystem protection) can be derived from guidelines for the treatment of acid sulfate soils (Ahern et al. 2004; Department of Environment and Conservation 2009).

3.2.3 Acidity treatment targets – moving beyond pH

The final water quality to be achieved by treatment will ultimately depend on the environment to which the water will be discharged or the purpose for which it will be used. It is reasonable to expect that treatment targets for water discharged to natural sites will differ from those of water disposed to purpose-built evaporation basins. Further uses of saline water after treatment may include aquaculture (George & Coleman 2002), desalination (URS 2002) or solar ponds (for heat and/or electricity generation; Ahmed et al. 2000). Each use will have different requirements – for example, a high level of treatment may be required for aquaculture while no treatment may be needed for water used in solar-pond electricity or heat production.

The use of treated water may also determine whether the targets are fixed or occasional deviations are acceptable (and when). For example, high-value environments such as ski lakes or lake conservation reserves may demand no breaches of pH and acidity targets while for lesser value or perhaps more resilient receiving environments the occasional low pH and higher acidity may be acceptable for some times in the year or during certain flow events when there is some capacity for surface flows to neutralise acidity.

Measuring pH provides a poor indication of acidity (see box – 'How pH can fail to show all of the acidity') and is therefore not suitable as a reliable target for treatment of Wheatbelt acidic drain water. Certainly, pH is a key, easily measured property of water and an important factor influencing aquatic ecology – but 'hidden' acidity needs to be accounted for to guarantee successful treatment and nullify the relative toxic effects of acidic waters on aquatic ecosystems, particularly due to dissolved aluminium and iron (see Section 2.3). Not all low

How pH can fail to show all of the acidity

Although all low pH water contains acidity, the amount can vary widely, from negligible to extreme (see Fig. 5). For example, at pH 3.5, drain water and groundwater in the WA Wheatbelt have been found to contain as little as 0.08 grams total acidity (as equivalent $CaCO_3$) per litre to 1.2 grams total acidity per litre. Although similar pH, the water will behave significantly differently in treatment systems and if discharged to creeks, floodways or lakes.

Conversely, while free hydrogen ions (measured as pH) also contribute to the acidity of water, this is often only a small fraction of total acidity in acidic saline drain and groundwater. This is illustrated in Figure 5 as the amount of acidity due to hydrogen ions alone (measured as pH) in various drain and groundwater samples from the eastern Wheatbelt.

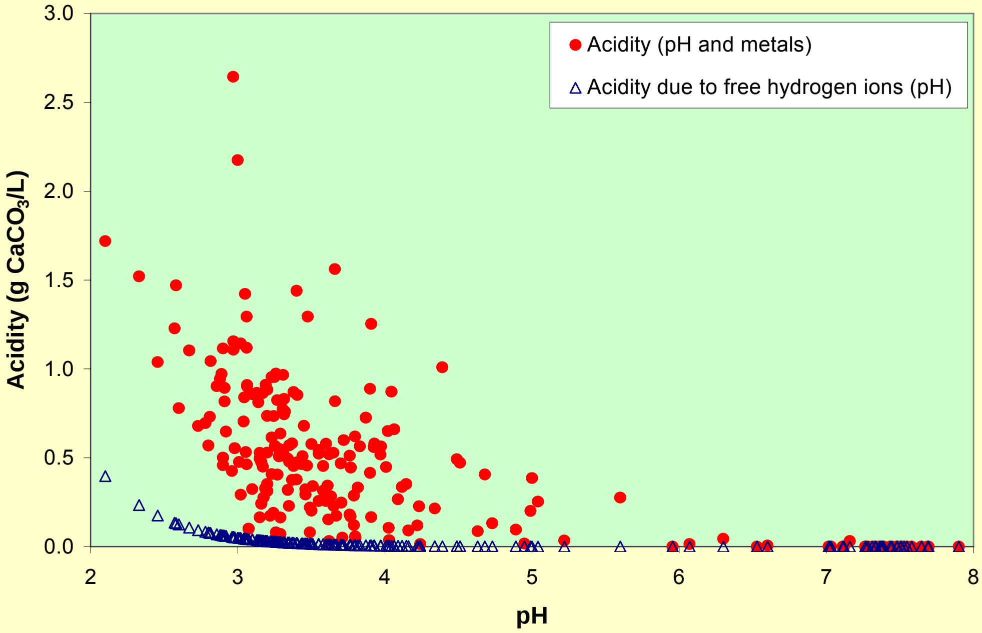


Figure 5 Hydrogen ion acidity (pH) and total acidity (from hydrogen ions and dissolved metals) in drain and groundwater in the WA Wheatbelt (214 sites)

pH waters are the same in this respect (Gensemer & Playle 1999; Gray & Delaney 2008).

Treatment of acidity in Wheatbelt drains is achieved when waters contain **no net acidity** and preferably some level of soluble alkalinity (e.g. bicarbonate).

Removing the 'metal' acidity, mostly associated with dissolved iron and aluminium, will eliminate the overwhelming effect of this on pH and allow water to find a new pH balance. For water held in an isolated tank open to the air, this pH will be around 6.3. However, the eventual pH of the water in the environment will be ultimately depend on pH buffering exerted by sediments and organic materials with which the water comes in contact. If water with no net acidity is discharged into environments with acidic sediments the water will most likely (unfortunately) be re-acidified.

The behaviour of pH in water after treatment of 'metal' acidity highlights the challenges in identifying pH targets. The target pH for natural ecosystems may vary widely since riverine and floodplain water (and therefore the associated aquatic ecosystems) can have pH slightly lower than that of some lakes (Degens et al. 2008b). Naturally acidic lakes also exist in the Wheatbelt (Degens et al. 2008a) where the target pH for discharge may be low, though the composition of soluble metals in the water will need to be considered.

Subsequent water uses such as desalination and aquaculture will require different pH targets. Potential target pH values for various aquatic ecosystems (high conservation value versus multiple-use systems) are currently part of ongoing work by the Departments of Water and Environment and Conservation.

It may not be necessary to achieve full water treatment if it is discharged to a contained environment. Discharge of a lower pH water (with residual acidity) to an evaporation basin where water is contained and managed may be more acceptable than if the water is to be discharged directly to an alkaline waterway and lake system.

Although the focus of these guidelines is on treating acidity, it will also be necessary to consider other aspects of water quality that can be inadvertently modified during treatment processes. These include changes in nutrient concentrations, increases in salinity and changes in ionic composition (e.g. increased calcium concentrations relative to other ions such as sodium, magnesium and potassium). As with acidity, the acceptable levels of each will depend on where the water is discharged or the requirements to enable water re-use.

A process for determining locally relevant pH and acidity limits that could be applicable for Wheatbelt waterways and lakes is outlined in the national water quality management strategy (ANZECC & ARMCANZ 2000). The strategy suggests a pH default value of 6.5 for upland waterways (this does not include inland lakes) with the requirement for further investigations should discharge waters be less than this (ANZECC & AMRCANZ 2000). However, this is only intended to be an interim level, is based mostly on data from coastal and fresh–brackish waterways (ANZECC & AMRCANZ 2000) and may be inappropriate for ephemeral inland saline waterways and lake systems where a higher pH may be required in some systems.

Similar problems arise for determining safe concentrations of minor elements. While the ANZECC and ARMCANZ guidelines offer 'default' values, these are based on long-term exposure risks to aquatic organisms in **fresh and marine** waters (see Appendix C for guideline limits) and may not be applicable to inland salt lakes or saline waterways. The guidelines are conservative and intended to trigger where further investigations will be needed to ensure protection of aquatic species at different levels in conjunction with a broad water quality management strategy (95% and 80% are given in Appendix C). This may be construed to mean that safe concentrations may be higher in many cases, particularly for temporary and intermittent lakes and waterways (Smith et al. 2004). Concentrations below analytical detection limits could be regarded as safe concentrations, though this is impractical given that many minor elements occur naturally in salt lake ecosystems, even those containing alkaline water (Degens et al. 2008a). These could be used to define suitable end-point concentrations.

In summary, while pH is often used in many existing targets and guidelines, acidic water treatment to achieve net acidity of zero with some excess alkalinity will significantly reduce most risks associated with disposal or re-use of acidic drain waters (Department of Environment and Conservation 2009). It will be necessary in some cases to include treatment targets for final pH, specific soluble metals and perhaps even nutrients, salinity and ionic composition. Establishing these water-quality targets will require consideration of the environment to which the water is discharged or the water-quality needs for re-use. Further guidance is provided within guidelines published as part of the National Water Quality Management Strategy (e.g. ANZECC & AMRCANZ 2000).

4 Information for designing treatments – what controls acidity loads?

A significant factor in the successful treatment of acidic drainage is matching the capacity of treatment systems with the drain acidity loads.

Designing treatment systems for acidic drainage involves gauging (or estimating) the flows and measuring the concentrations of acidity and dissolved elements that a potential treatment system may need to handle. Estimates of acidity loads and the timing of these are also required since treatment systems are generally sized to handle daily loads rather than concentrations.

The daily acidity load for a period is basically estimated from the total water volume flowing from the drain during a day multiplied by average acidity measured in the flow. Ideally, this estimate should be based on several measurements but often only one is used (see also Glossary).

Scoping daily acidity loads requires at least a broad understanding of the rate and timing of water *entering* drains to understand the timing and quantities of acidity and dissolved elements *leaving* drains. At the next level, it can be useful to bear in mind how acidity and dissolved element concentrations change depending on season and drain flows.

The following sections explore some of the main factors involved in flow and water chemistry to provide a backdrop for estimating how daily acidity loads from drains may change over several years.

4.1 Water flows

Predicting the timing of flows from drains is one of the main components of ascertaining the daily pattern of acidity loads. This can be achieved by estimating flows due to groundwater discharge along the length of the drain and flows due to surface runoff.

Drains with flows dominated by groundwater discharge generally have a flow pattern consisting of:

- daily variations due to rainfall events and atmospheric pressure causing flow directly in the drain
- seasonal and year-to-year variations in groundwater inflow due to rainfall recharging groundwater in the catchment (wet years vs dry years)
- seasonal variations due to increased evaporation in summer and reduced evaporation in winter.

This was well illustrated in investigations of drainage performance at Beynon Rd, east of Dumbleyung (Cox &Tetlow in press) and other sites near Morawa, east Pithara and Beacon (Cox in press).

Patterns of surface runoff are superimposed on the patterns of flow due to groundwater discharge and can vary widely depending on drain design. Where drains can carry runoff from large areas, there may be significant seasonal and year-to-year variations in drain discharge due to the patterns of catchment runoff.

In contrast, where the runoff area is restricted to a few metres either side of a drain with bunds, the variations in drain flows due to runoff are much smaller and often more predictable. Simple estimates of expected runoff during events can be made by rainfall runoff calculations (e.g. Davies & McFarlane 1987) with the catchment area defined by whether the drain is leveed or not.

4.2 Factors affecting acidity and dissolved element concentrations

Acidity, pH and dissolved element concentrations vary between seasons and rainfall events as results of a complex series of chemical reactions modifying the chemistry of drain water moving through deep subsoil horizons to drains, through drain sediments and along the drains.

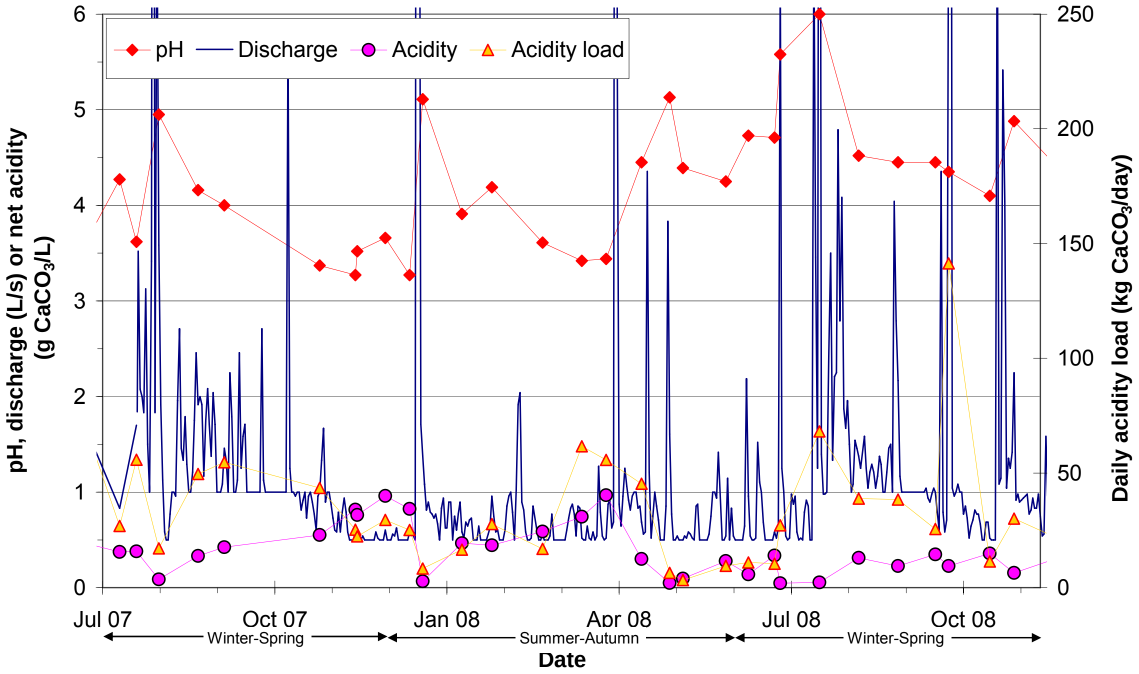


Figure 6 pH, net acidity, discharge rates (as daily average L/s) and daily acidity loads for water from a 9.6 km leveed deep drain in the Wallatin Creek catchment (constructed June 2006)

Acidity, pH and dissolved element concentrations in drain flows generally vary little during baseflow conditions which occur when groundwater inflows dominate. Acidity can be higher and pH slightly lower (sometimes as low as 2.5) in summer and autumn baseflows compared with winter–spring baseflows (Figs 6 & 7).

Concentrations of aluminium, iron and minor elements such as lead and zinc may also generally follow the same pattern of being higher in summer–autumn than winter–spring flows. These patterns are most likely due to evapoconcentration of water within the drains, increased contact time with sediments (contributing to higher aluminium concentrations) and increased oxidation and settling of iron-dominated minerals.

Formation and oxidation (drying) of black muds in the base of some drains can also influence dissolved metal composition in drain water, particularly during low flows. These black muds contain sulfide minerals that generally form when drains are flowing and the muds are waterlogged (Fitzpatrick et al*.* 2008). If several centimetre thick layers of sulfides form in drain sediments, they may begin to reduce acidity and dissolved metal concentrations (Fitzpatrick et al*.* 2008). Drying of the muds and increased penetration of air when drains stop flowing can result in the oxidation of sulfide minerals which will contribute additional acidity to drain water (Fitzpatrick et al*.* 2008).

Although the rates of acidity and dissolved element movement are dominated by groundwater discharge to drains most of the time, these can change rapidly during rainfall events when surface runoff can greatly alter water chemistry, acidity and element loads in drain flows. This can occur as inflow neutralises and to some extent dilutes drain water and as flows flush out some of the minor elements and acidity accumulated during low flows.

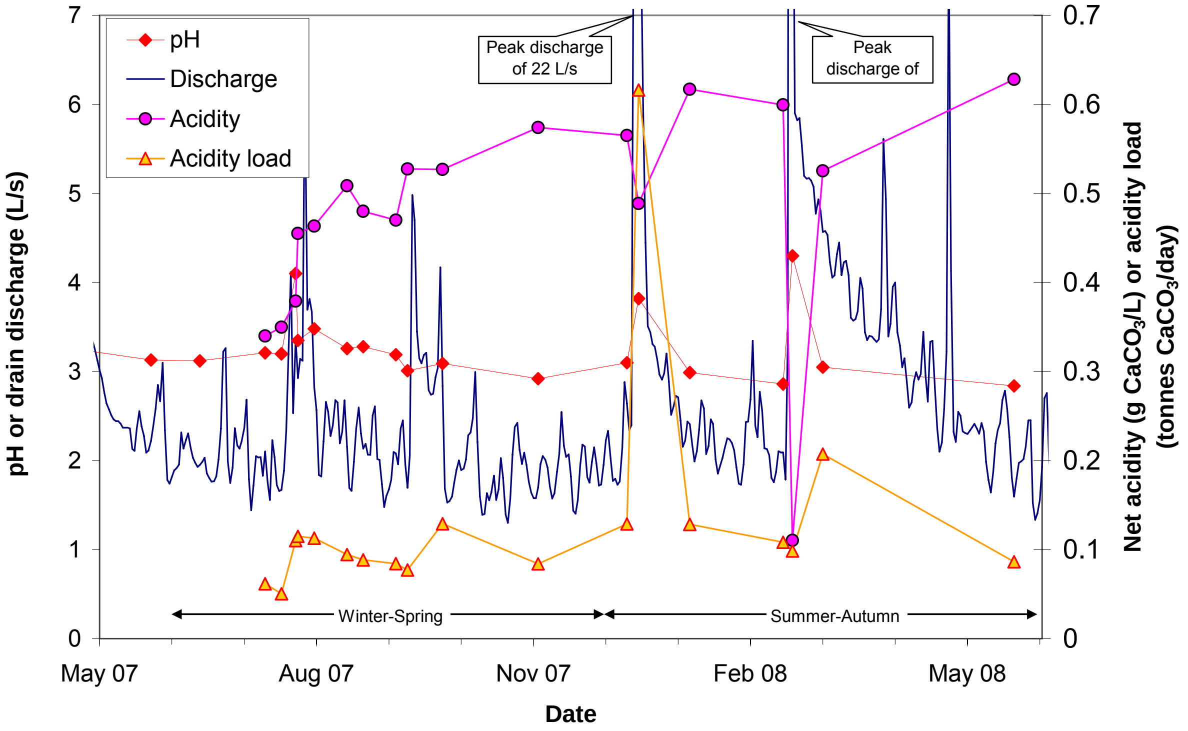


Figure 7 Seasonal and flow effects on pH, net acidity, discharge rates (as daily average L/s) and daily acidity load for water from a 6.5 km section of leveed deep drain in the Beacon River catchment (measured at Scotsman Road)

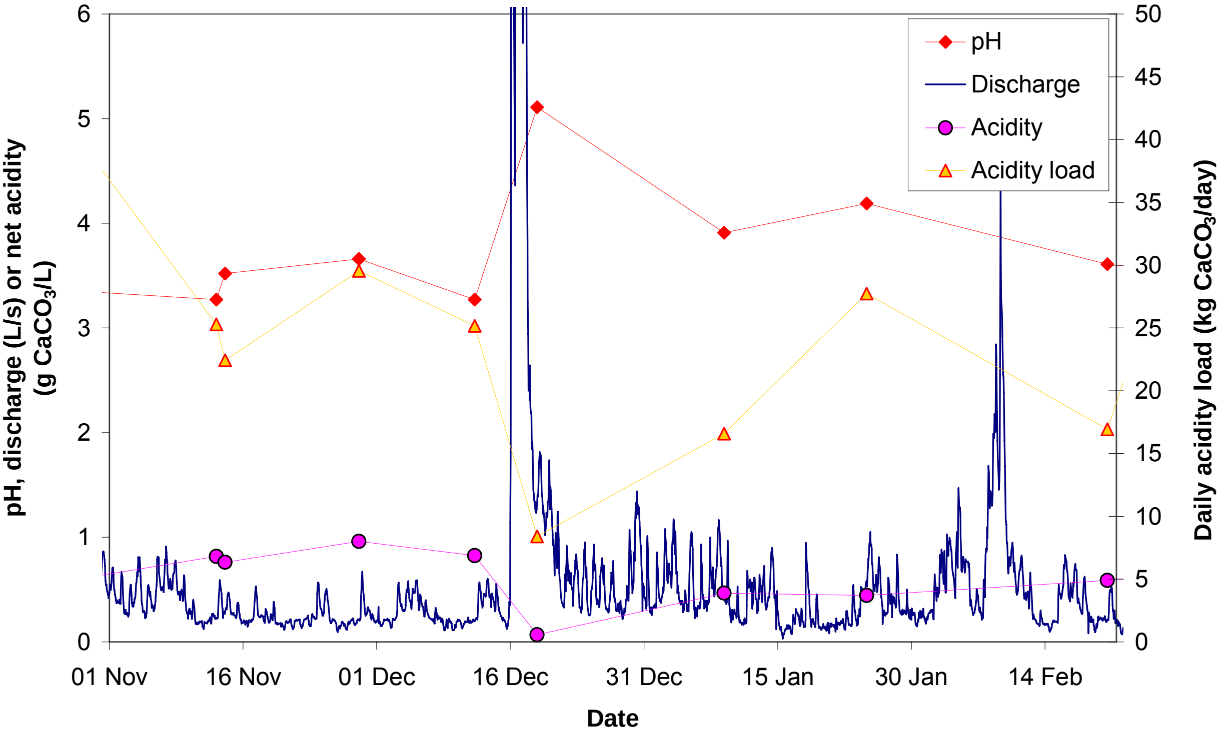


Figure 8 Changes in pH, net acidity, daily acidity load and flows from a 9.6 km leveed deep drain in the Wallatin Creek catchment before and after 36 mm rainfall over three days

Partial and sometimes complete neutralisation of drain discharge occurs during flow events when runoff from drain berms and walls may carry carbonate-rich materials into the drain. Flows in some leveed drains may become less acidic and the overall acidity load may decrease. For example, flows in a deep drain near Doodlakine after rainfall in mid-December 2007 resulted in pH exceeding 5.5, acidity decreasing to <0.07 g $CaCO_3$/L and the acidity load decreasing to less than 30% of that in preceding days (Fig. 8).

The rainfall dilution and neutralisation effect during high flows does not occur in all drains. Although some dilution and neutralisation can occur the net result of increased flows after rainfall may be increased flushing of acidity from drains, despite these being higher in pH. An example of this occurred in a deep drain at Beacon when, although pH increased to almost 4.5 and acidity decreased or remained unchanged during higher flows (July, November and December 2007), the acidity load could increase by up to 4 times compared with baseflows (Fig. 7).

Where large flows can occur in drains, particularly unleveed drains, sediments accumulated in drains during low flows may be scoured out. Many of these sediments can contain minor metals (Fitzpatrick et al. 2008) and will increase the total load of metals in drain water during high flows. Such events are known to occur when floodwaters flush through other acidic environments (Hammarstrom et al. 2005; MacDonald et al. 2007) similar to those in WA Wheatbelt drains. Drain design that minimises areas of surface runoff (with levees) is likely to reduce the likelihood of scouring events.

Where drain flows are seasonally intermittent (that is, flow ceases in summer and recommences in winter) acidity loads may become dominated by a first flush, most likely to occur early in winter as a pulse of acidity and dissolved metals concentrated during summer. First flushes with high concentrations of acidity and dissolved metals are typical where acidity and minor metals accumulate in minerals and salts rather than water: common in many areas with acid rock minerals (Harris et al. 2003; Nordstrom 2008) or acid sulfate soils (Green et al. 2006). Although monitoring of water chemistry in Wheatbelt drains has not been able to capture such events, it is expected that similar patterns would also occur.

Reliable flow and acidity information to design a treatment system is rarely obtainable at the outset. The above summary can be used to broadly estimate likely water quality and flows where treatment systems are designed before drain construction. This would need to be complemented with ongoing drain flow and water quality monitoring (after construction of a treatment system) with further adaptations and improvements to address any short-falls. While monitoring drains for some time before constructing a treatment system could provide a better basis for design, this means accepting discharge of untreated water. After the treatment system is constructed there may need to be some remediation of the discharge environment.

5 Designing treatment systems

The design of treatment systems for acidic drain water centres on selecting one or a combination of treatment options based on understanding the flows and acidity with which the system must cope. This design can become an iterative process where constraints with individual options might see the need to use alternative options or combinations of options to improve reliability of treatment or cover a range of flow conditions (also see integration of treatment options in Section 8). The initial steps include deciding which flows will need treatment, what approaches might be needed to handle these flows and which sites are available for constructing treatment systems. Addressing these points provides the basis for selecting which treatment options might be appropriate before starting to calculate sizes and designing system construction (covered in later sections). Steps to guide these decisions are illustrated in Figure 9.

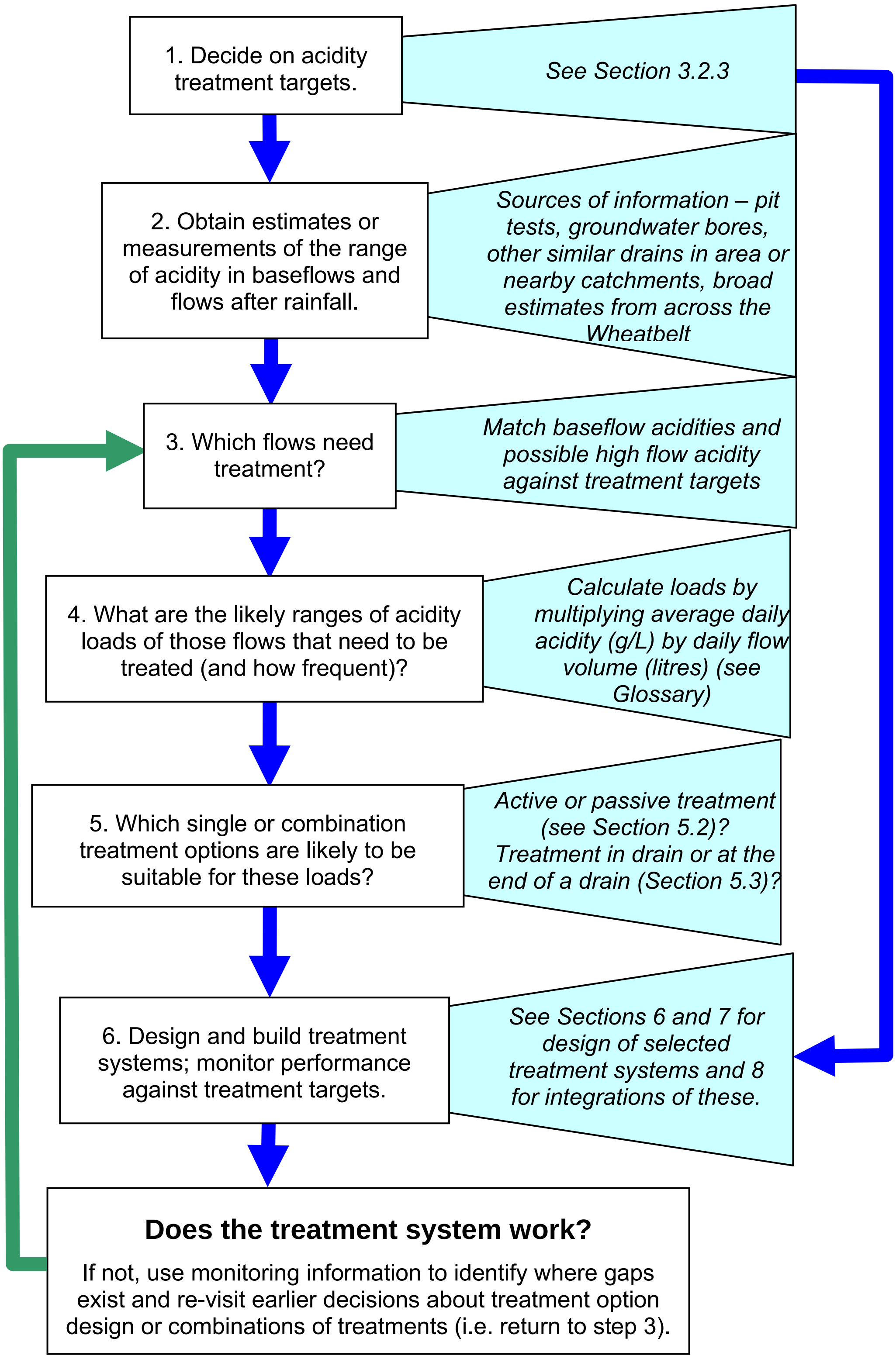
Adaptive management is considered important for acidic drain water treatment because of the large uncertainties regarding the acidities and volumes of drain flows combined with the limited knowledge about the long-term performance of many treatment systems. The process of designing treatment systems is not a one-way path and needs to be linked with management and monitoring to achieve continuous improvement (Fig. 9). This follows the principles of adaptive management where a lack of information to design the best treatment system (one that treats all flows that need treatment) can be offset by using the monitoring of the treatment performance to continuously improve the design (or add more treatments). Designing the best treatment system, where there is great confidence that it will work effectively, requires detailed information on the acidity and flows to be treated and detailed information to predict how the system will perform. Neither is currently available and probably never will be for all situations.

5.1 Deciding which flows need treatment

To avoid unnecessary treatment costs or impacts from failing to treat unexpected acidity in drain flows it is important to identify whether all or only some of the flows from a drain need to be treated. Some key questions to consider include:

- Is any of the drain water likely to be acidic (that is, pH <5.5 and total acidity >0.1 g $CaCO_3$/L)?
- Will acidity be treated only in flows during the first months after construction (during the dewatering phase) or over the long term in the years after construction?
- Will all acidity be treated in all flows including high flows after rainfall events or only baseflows in the weeks–months between these times?
- Is it necessary to treat acidity loads in all high flows following rainfall or just those that are smaller than a certain size?

Note that deciding which flows need to be treated is greatly helped by understanding the targets the water needs to meet with respect to acidity (see Section 3.2 & Fig. 9).



1. Decide on acidity treatment targets.

See Section 3.2.3

2. Obtain estimates or measurements of the range of acidity in baseflows and flows after rainfall.

Sources of information – pit tests, groundwater bores, other similar drains in area or nearby catchments, broad estimates from across the Wheatbelt

3. Which flows need treatment?

Match baseflow acidities and possible high flow acidity against treatment targets

4. What are the likely ranges of acidity loads of those flows that need to be treated (and how frequent)?

Calculate loads by multiplying average daily acidity (g/L) by daily flow volume (litres) (see Glossary)

5. Which single or combination treatment options are likely to be suitable for these loads?

Active or passive treatment (see Section 5.2)? Treatment in drain or at the end of a drain (Section 5.3)?

6. Design and build treatment systems; monitor performance against treatment targets.

See Sections 6 and 7 for design of selected treatment systems and 8 for integrations of these.

Does the treatment system work?

If not, use monitoring information to identify where gaps exist and re-visit earlier decisions about treatment option design or combinations of treatments (i.e. return to step 3).

Figure 9 Key steps in making decisions that help in the design (including where information can be obtained) and adaptive management of treatment systems

Obtaining enough information to make sound decisions about required treatment and its timing before drain construction is important. Other drains in the area or in similar catchments elsewhere can provide some background, particularly if these are monitored for at least a year. When this is not possible, acidity can be estimated from pit tests or pump sampling of shallow monitoring bores in the area where a drain is planned. Flows are also commonly estimated to support the drain design. If no other sources of acidity information are available, it is best to use the conservative estimates of acidity (e.g. using average or upper limit acidities in existing Wheatbelt drains described in Section 3 and illustrated in Fig. 4) and flows in this guide and collated elsewhere for Wheatbelt drains (Shand & Degens 2008).

5.2 Choice of approaches

Deciding which flows will need treatment and when will help in deciding which treatment approaches might be needed. Generally, active treatment systems are more suitable for drainage water with high acidity (>0.8 g $CaCO_3$/L) and/or consistently high acidity loads (>150 kg $CaCO_3$/ day; Taylor et al. 2005) and passive treatment systems more suitable for drainage waters with lower acidity and/or mostly lower acidity loads.

Active treatment systems require continual input of (bio)chemicals and energy (Younger et al. 2002) and usually require regular servicing and maintenance. These are typically intended for short treatment periods or large acidity loads and can be very responsive.

Passive treatment systems involve low maintenance and energy inputs after establishment, are usually gravity-driven (i.e. do not involve pumping) and are intended for long-term operations spanning many years (Younger et al. 2002). Passive treatment systems can be slow to respond to acidity changes but do not tend to fail suddenly. From an economic perspective, passive treatments may incur smaller ongoing operating and maintenance costs, but incur greater establishment costs.

Various active treatment options have been developed nationally and internationally to remediate acidic waters (often to deal with acid mine drainage). The majority broadly involve oxidation, dosing with alkali, and sedimentation (Younger et al. 2002). Those potentially suitable for acidic water in Wheatbelt drains include:

- **Calibrated reagent applicating blender (CRAB)** – principally using hydrated lime, but can be tailored for other neutralising materials (Green et al. 2007)
- **Hydro-active limestone treatment (HALT)** – portable limestone grinding mill for direct dosing of waterbodies/ways (Taylor et al. 2005)
- **Aqua-fix system** – granulated quicklime (CaO) direct dosing system with the option of being powered by water flows (Taylor et al. 2005)
- **Hydrated lime dosing** – controlled dosing of hydrated lime slurry to achieve rapid and controlled rise in pH (Taylor et al. 2005).
- **Pulsed limestone reactor** – semi-trailer mounted throughflow limestone reactor with CO_2 injection and recovery systems (Sibrell et al. 2003).

- **Closed tank reactor (limestone filled reactor)** – limestone aggregate-filled tank through which waters are intermittently or continuously pumped (Green et al. 2008).

Recent local research has highlighted that ethanol-fed bacterial bioreactors might also have some potential as an active treatment option (Franzmann et al. 2007). However, this process is currently more expensive than any chemical neutralisation treatments and may only become cost-competitive with further development to enable efficient harvesting of economically valuable metals.

A complementary suite of passive treatment options has also been developed internationally over the past 20 years to treat both net alkaline and net acidic waters (Hedin et al. 1994; Skousen et al. 1998). Many are still undergoing continual investigation and improvement (Younger et al. 2002; Watzlaf et al. 2004; Doshi 2006). The passive treatment options suited for treatment of **net-acidic** waters with high iron concentrations typical in many eastern Wheatbelt drains include:

- **mixing with waters containing alkalinity** – see Section 2.4
- **anoxic limestone drains** – where water flows through limestone sealed within a channel (with a slope of at least 10%) covered by compost
- **oxic (aerobic) limestone drains** – an open channel lined with coarse limestone where water flows through fast (often with a slope of at least 10%)
- **composting wetlands (sometimes called anaerobic wetlands)** – a permanently waterlogged organic mixture through or over which water flows
- **limestone ponds** – shallow ponds filled with limestone where water pools until treated or treated by infiltration
- **limestone diversion wells** – concrete wells filled with limestone through which water diverted from a flow channel is forced with enough pressure to agitate the limestone
- **reducing and alkalinity producing systems (RAPS)** – a pond of composting mixture overlying limestone through which water flows downwards; generally suited to areas with >1% gradients or where water can be ponded 1–2 m above ground
- **permeable reactive barriers** – porous reactive materials mounted on racks that chemically remove dissolved metals (and other elements) and neutralise waters directly from flow channels (though normally applied to treat groundwater).

The above list accounts for the fact that many drain waters contain high concentrations of iron and aluminium, nil or negligible alkalinity and are often poorly oxygenated (see Section 2.2). There is limited information available to distinguish which of the passive treatment options might be suitable for WA Wheatbelt drains of differing water qualities or provide a decision tree to assist this as is commonly provided in guides for design of treatment options for acid mine drainage (see decision trees in Hedin et al. (1994), Younger et al. (2002) and PIRAMID Consortium (2003)).

Anoxic limestone drains and RAPS may have limited application in Wheatbelt drains since these are only suited to use in treating acidic water with less than 2 mg aluminium/L and limited dissolved oxygen (Younger et al. 2002; PIRAMID Consortium 2003). Most Wheatbelt drain water contains much higher concentrations of aluminium and is often highly oxidised (Shand & Degens 2008). Under these conditions, the aluminium will often precipitate within and may eventually clog anoxic limestone drains and RAPS without regular flushing (Watzlaf et al*.* 2004; Cravotta & Ward 2008). For further information on RAPS design, consult Watzlaf et al*.* (2004) or PIRAMID Consortium (2003).

Most passive treatments have been developed in the northern hemisphere where conditions are somewhat different to the Wheatbelt and so their performance may differ under WA conditions.

Conditions providing opportunities and constraints for the use of passive treatment systems include:

- **low relief** – There is little opportunity for significant head-driven flows in drainage systems. The low gradient of many Wheatbelt valley floors (commonly less than 0.1%; Beard 1999) and design to minimise scouring flows restricts the gradient of many drains.
- **higher temperatures** – Average maximum summer air temperatures often exceed 30 ºC (Bureau of Meteorology 2008), which can have the benefit of speeding up chemical and microbial reactions in soils and waters (provided that soils remain moist). Average winter minimum air temperatures are mostly above 5 °C (Bureau of Meteorology 2008) which can reduce the cooler season limitations that may occur for biological treatment systems (PIRAMID Consortium 2003).
- **higher salinity** – There are no reports of treatment systems handling saline waters, particularly with high concentrations of sodium and chloride. The saline nature of most acidic Wheatbelt drain waters may influence the behaviour of some dissolved metals and some of the chemical reaction rates. On the other hand, the saline waters inherently contain high concentrations of sulfate, a benefit to microbial sulfate-reducing treatment systems (covered in later sections).
- **higher evaporation rate** – High evaporation rates during summer and autumn in the WA Wheatbelt (with daily average evaporation commonly exceeding 10 mm) can result in increased acidity and dissolved element concentrations in waters. Evaporation restricts the maximum time water can circulate through treatment systems. Where treatment times are significant, evaporation from a treatment system can result in no outflow and the treatment system behaving as an evaporation basin.

Active and passive treatment approaches are not mutually exclusive and a combination might be needed to achieve treatment, with each targeting different types of flows. Most cannot be used in isolation and often require settling and/or oxidation ponds. Given that many drains are intended to operate with little need for landholders to continuously check or maintain flows, passive treatment systems are more likely to be appropriate.This does not excllude the option of all treatment for a drainage system being handled by a single active treatment system, perhaps managed by a regional service provider.

Guidance on matching treatment approaches to combinations of average daily flows and acidities is illustrated in Figure 10. For example, an average daily baseflow discharge of 3 L/s with an average acidity of 0.5 g $CaCO_3$/L may be treatable with passive treatment methods (point (a) in Figure 10) but if the acidity for these baseflows was 0.7 g $CaCO_3$/L active treatment may be necessary (point (b) in Figure 10). Likewise, baseflow discharge averaging 3 L/s with an average acidity of 0.5 g $CaCO_3$/L (point (a)) could be handled by passive treatment systems, but if flows after rainfall increase to more than 5 L/s with little decrease in acidity (point (c) in Figure 10), active treatment during these events may be needed to complement passive treatment.

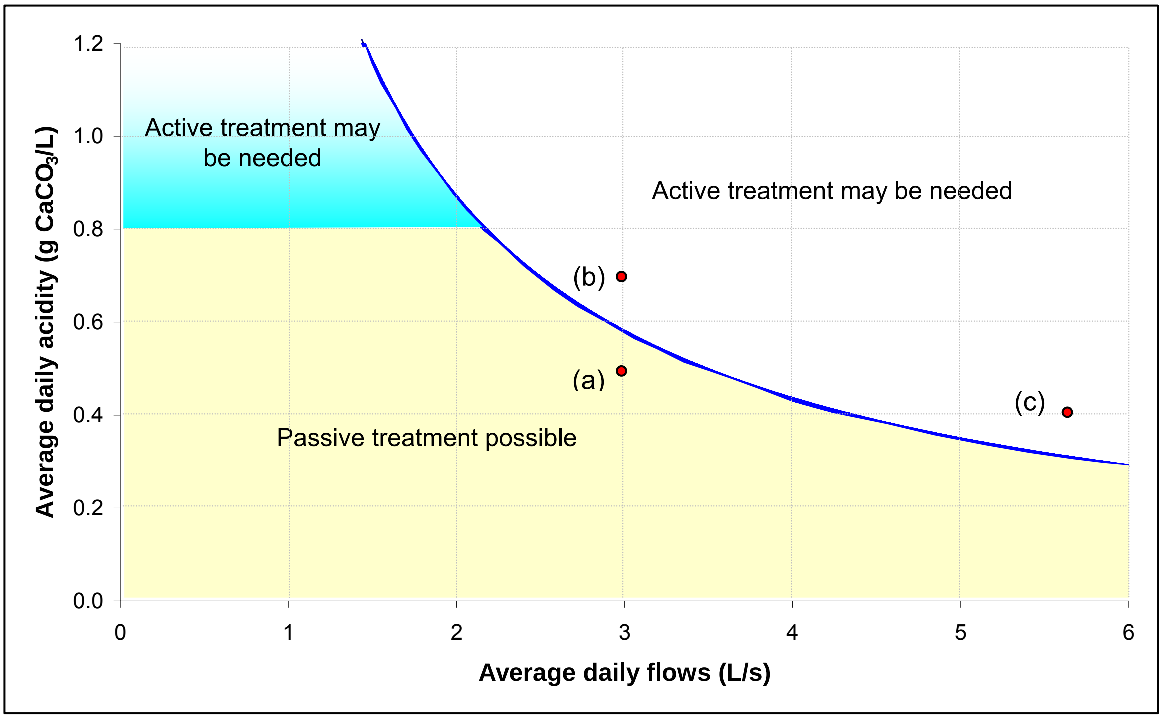


Figure 10 Treatment options as a function of average daily flow and average daily acidity (after Taylor et al. 2005)

5.3 Where can treatment be carried out?

The choice of available sites for treatment can largely control the most appropriate treatment options. Although water acidity places considerable constraint on the choice of approaches (as covered in the above section), the additional limitations posed by site selection can sometimes override these, particularly when constrained by the availability of land area and electrical power.

Active treatment depends on available (and reliable) power supply and all-weather access; which may not be achievable without considerable additional cost. This is particularly relevant since many terminus points for drains are low-lying floodplains or lakes prone to winter waterlogging and generally at some distance from existing infrastructure. However, there are situations when active treatment may be favoured where pumping is carried out (between drains or from drains to evaporation basins) and opportunistically within drains (where powerlines cross drains or nearby to sheds with power).

Selection of sites for passive treatment depends on the preference for a single end-of-drain treatment site or for distributed treatment sites throughout a drain. The latter approach effectively involves treating water as it seeps into and flows within drains, reducing the size of the treatment system needed at the drain end (this may only be required for the final polishing of water, see Section 8 Integration). These preferences will be selected according to whether landholders can commit to building and periodically maintaining a number of smaller in-drain treatment systems or will instead concentrate efforts on a single larger treatment system at the drain end. From a design perspective, a single point of treatment will be required to handle larger flows and acidity loads than many smaller in-drain sites.

Factors to consider for treatment within drainage channels include:

- **channel gradient** – The change in gradients along the length of the drain affects where water ponds (targeted for some in-drain treatments such as composting beds or avoided for lime-sand beds). These ponds could perhaps be built into the design of a drain before construction.
- **channel width** – The width of the drain floor sets the area available for in-drain treatment.
- **bank slopes** – Steep slopes may result in slumping or pipe erosion, with sediment deposition onto the drain floors. This may limit the effectiveness of some treatments.
- **access points** - Bench width and bank height may restrict machinery access for periodic maintenance/cleaning and sludge management. Confluence points and farm/road crossings offer sites for point treatment.

Factors to consider for siting end-of-drain treatments include:

- **area** – Find sufficient area for containment of waters and treatment (plus future expansion of treatment systems if required).
- **access** – Consider all-weather access if this is required for service and maintenance vehicles (particularly active treatment sites adjoining waterlogged flats). Access will also be required for periodic sludge removal and management.
- **gradient** – Land slope across the site will control the volumes of soil that may need to be excavated, whether water can be gravity-fed into, through and out of a treatment system and where the discharge points might be placed.
- **available power (only for active treatment systems)** – Consider location of treatments in relation to power lines and the security of the power supply, particularly the risk of power outages during thunderstorms and consequent reconnection times.
- **flooding risks** – Consider placement in relation to floodways, avoiding structures that impede flood flows and that, if flooded, will not cause off-site impacts.
- **public access/safety** – Purpose-built treatment systems require consideration of public health and safety, particularly when sited near towns or public amenity areas.
- **long-term management of materials** – Siting of treatment wetlands requires considering whether the watertable will remain at levels that maintain saturation of sediments over the long term or be sited where local soils contain carbonates that would offset risks of acid release (see Section 7.2.2).

- **aesthetics** – The visual amenity of treatment systems may need to be considered in some situations (e.g. if close to town sites).
- **approvals** – The treatment system may require notifying the Commissioner for Soil and Land Conservation, obtaining permits from the Department of Water to modify beds and banks in watercourses (most areas of the Avon Basin) and permits from the Department of Environment and Conservation to clear land, if perennial vegetation is to be removed or flooded to construct a treatment system.

5.4 Selecting treatments – matching options with treatment requirements

The penultimate stage of designing treatment systems requires selecting a treatment option or combination of options appropriate to handle the flows requiring treatment (baseflows or both baseflows and high flows) and considering whether the flows can be treated by the approach taken (i.e. active or passive or a combination) and the site constraints for the different options. While making the final choice of treatment system it is often necessary to consider also some of the broad capabilities and limitations of each. Table 1 is a summary of common treatment options regarded as suitable for the WA Wheatbelt. As mentioned previously, all options primarily consist of an acidity neutralisation stage (mostly dealing with pH, aluminium and iron) and removal of dissolved metals stage, which can occur at the same time or after the neutralisation stage. Figures 11 and 12 present schematically the likely relative performances of a range of treatment options under various acidity and flow conditions. The figures are based on the limited available information for single trials of individual treatment options under WA conditions and will change as more evaluations are carried out. It is also important to note that all treatment options involve a need to manage the sludges produced by treating acidic waters, since these contain the metals and other minor elements trapped from the drain water.

The final stage of the design process involves sizing, selecting materials and detailing construction to ensure optimal operation. Design details (including case studies) are covered in Sections 6 and 7, and Section 8 discusses how options can be integrated or used in combination.

Table 1 Overview of applications, best performance outcomes and limitations and risks of treatment options most appropriate for treating saline acidic Wheatbelt drain water in low relief landscapes where some performance information is available (see case studies in Sections 6 & 7) (continues on next page)

Option	**Application**	**Best performance outcomes**	**Limits and risks**
Lime-sand beds*	In-drain with limited sedimentation rates	Continuous low-level neutralisation of baseflows with low pH and low iron concentrations throughout a drainage system; no excavation costs	Iron coating of sand greatly reduces effectiveness (increases with water containing high soluble iron concentrations) Burial by sedimentation Flushing of sludges from drain
Subsoil carbonate beds*	In-drain when available during construction	Continuous low-level neutralisation of baseflows with low pH and low iron concentrations throughout a drainage system; no excavation costs	Not available at all sites Iron coating of carbonates greatly reduces effectiveness (increases with water containing high soluble iron concentrations) Burial by sedimentation Flushing of sludges from drain Marginally increases cost of drain construction
Subsoil carbonate lining	On berms when available during construction	Provides additional loading of alkalinity to runoff into drains, potentially reducing acidity flushing in high flows	Not available at all sites May contribute to sedimentation in drain
Composting beds	In-drain, but sections not subject to high flows	Continuous treatment (acidity and a broad-spectrum of dissolved metals) of baseflows particularly with high iron and/or aluminium concentrations throughout a drainage system; limited construction costs Minimises flushing of acidity in higher flows	Sizing limited by area available in drain base Greater risk of failure in drains with high acidity loads in baseflows Loss of drain depth (but may be comparatively cheaper than alternatively large end of drain wetland) Potential flushing of sludges in drains with high flows Treatment rates decline with time
Lime-sand reactors	In-drain where power available or solar power possible	Continuous mid–high level point neutralisation of baseflows	Power supply needed or solar feed, continuous supply of lime-sand required High iron concentrations may reduce efficiency of lime-sand use Not effective for all dissolved metals

Option	Application	Best performance outcomes	Limits and risks
Lime-sand reactors	End-of-drain coupled with holding/settling ponds	Continuous mid–high level treatment of drain flows (neutralisation and removal of metals).	Power supply needed or solar feed; continuous supply of lime-sand required Pump capacities unlikely to handle flows varying across 2 orders of magnitude unless used in conjunction with large containment basins High iron concentrations may reduce efficiency of lime-sand use Not effective for all dissolved metals
Lime-sand basins (ponds)	End-of-drain, requiring large pre-treatment holding areas	Batch neutralisation of water	Large area required for oxidation treatment prior to lime-sand treatment Risk of iron coating of lime-sand reducing effectiveness greater with water containing high soluble iron concentrations (>10 mg/L). Slow, fixed rate of treatment during which water can become more saline Not effective for all dissolved metals
Composting wetland	End-of-drain, sometimes requiring large areas	Continuous high level treatment (neutralisation and removal of a broad-spectrum of dissolved metals) of flows if sized accordingly Suitable for water with high iron concentrations	Treatment rates decline with time Water can become more saline if treatment rates slow May require large areas of land and large excavation pits
Hydrated lime dosing plants	End-of-drain coupled with settling ponds/tanks	High certainty treatment of acidity and dissolved metals for end-of-drain discharge (across a range of iron and aluminium concentrations)	Power supply needed or solar feed; continuous supply of hydrated lime required Pump capacities unlikely to handle flows varying across 2 orders of magnitude unless used in conjunction with large containment basins

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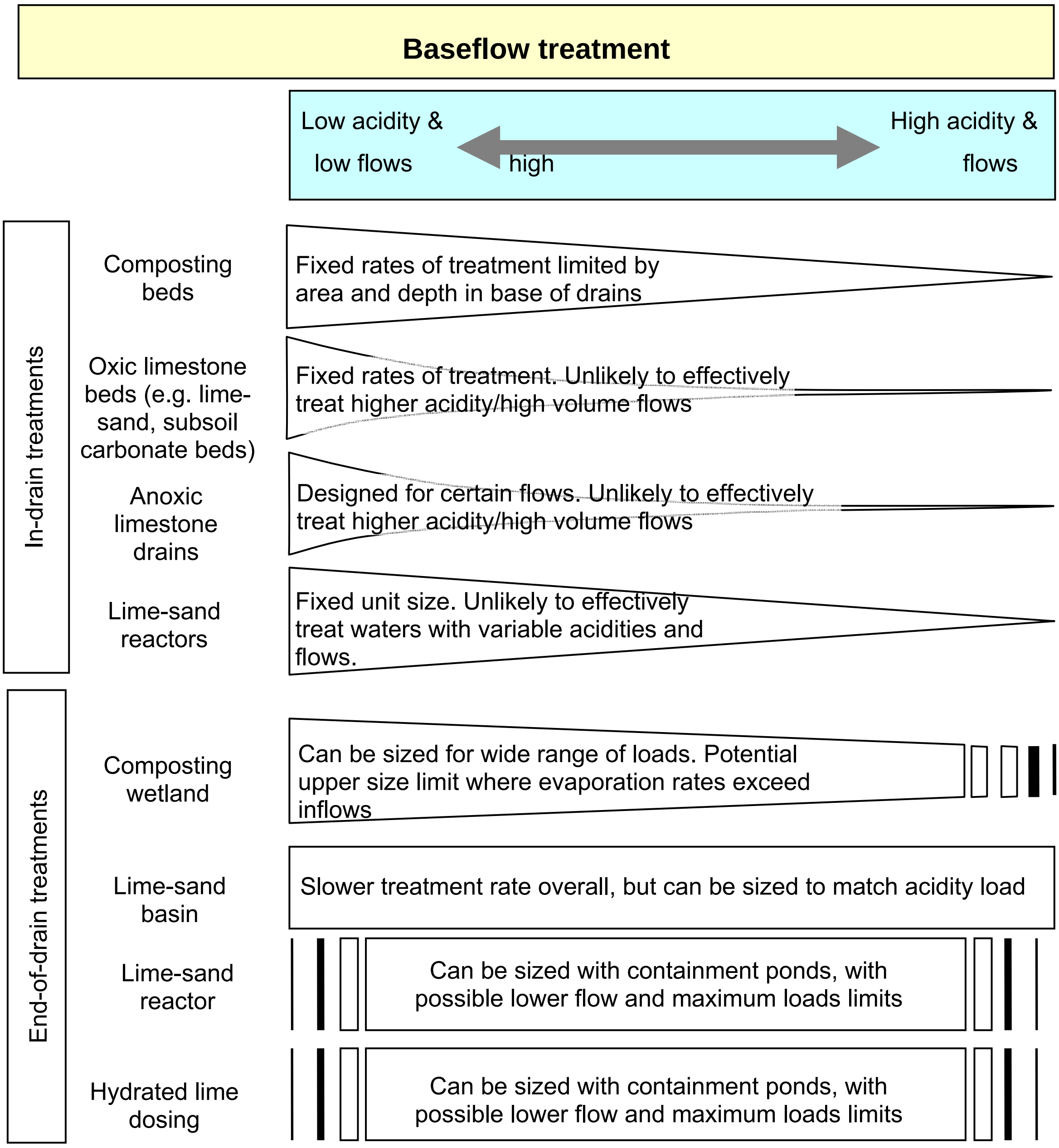
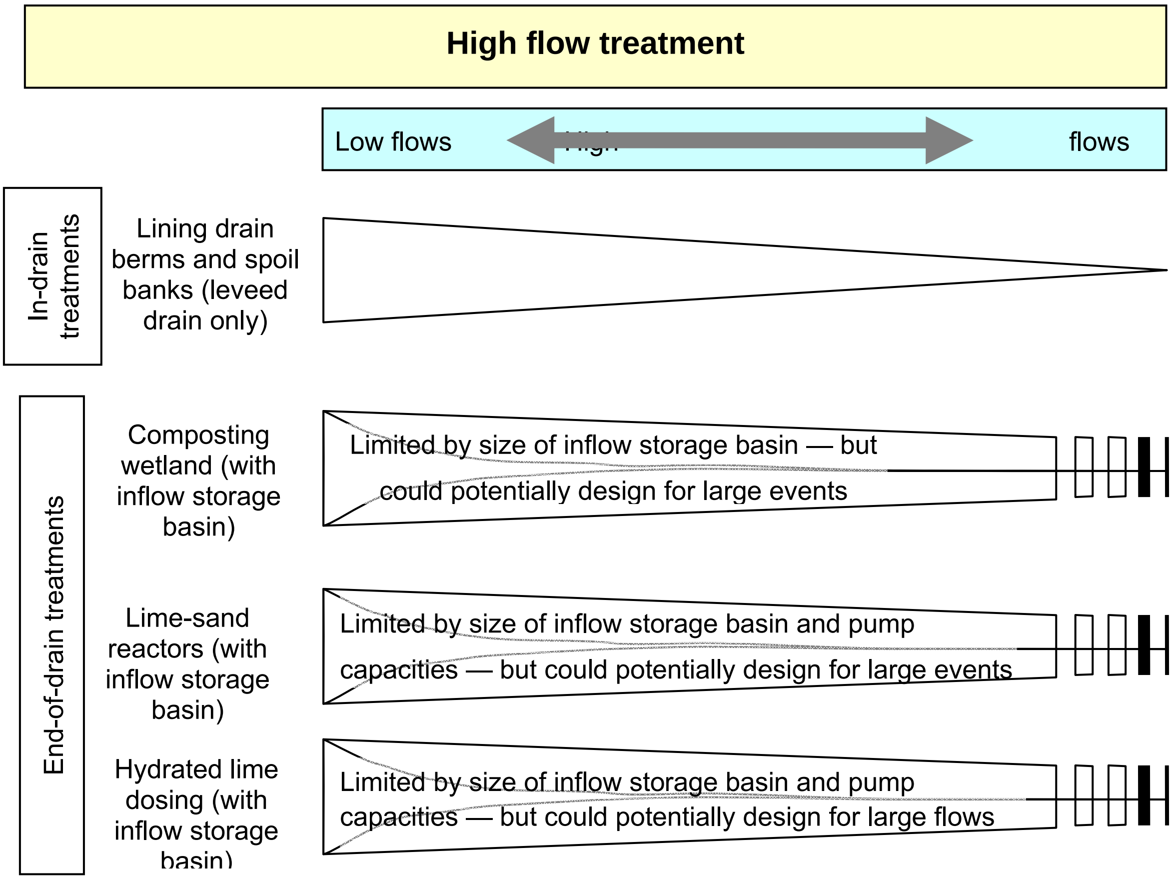
* *Lime-sand and subsoil carbonate beds are a variation of the limestone drains used internationally although these are usually purpose-designed for high through-flow and commonly use coarse aggregate limestone.*



Note: Likely success is represented by the width of bars and uncertain success represented by broken bars at each extreme.

Figure 11 Diagrammatic comparison of the suitability of individual treatment options for a range of baseflow conditions



Note: Likely success is represented by the width of bars and uncertain success represented by broken bars at each extreme. The likely success of end of drain treatment options with and without storage basins is represented by the smaller and larger bar widths, respectively.

Figure 12 Diagrammatic comparison of the suitability of individual treatment options for a range of high flow conditions

5.5 Reducing the risk of failure

It is extremely unlikely that any system designed has no risk of failure. When designing treatment systems for acidic saline drain water, the main problems are likely to be those that limit treatment rates or unexpected variations in acidity or drain flows.

Factors that limit treatment rates are linked with either the neutralisation rates (chemical or microbiological) or physical flow restrictions such as blockages with silts, sludges and slimes that result in acidic water bypassing neutralisation sites. Neutralisation rates can decrease when chemical neutralising materials get coated with mineral crusts or are buried. Both result in the neutralising materials reacting more slowly with acidity in water due to the decrease in reactive surface areas.

The neutralisation rate in microbiological treatment systems (e.g. composting treatment systems, Sections 6.3 & 7.2) might decrease if the organic mixture rapidly decomposes and cannot support high rates of microbial activity or if acidity affects the microbial activity (i.e. acidity inflow exceeds the rate of microbial neutralisation). Further details on managing these risks are covered individually for each of the treatment options in these guidelines.

Volumes of water and acidity concentrations can vary widely in drains with seasonal conditions (as outlined in Section 4). This variability and the risks to treatment effectiveness can be accommodated by some of the following design modifications:

- **over-sizing** – This can be done to allow for some decline in treatment effectiveness with time and also provide excess treatment capacity for larger pulses of acidity. The additional costs of construction must be considered.
- **in-built storages** – Water can be stored to allow treatment, effectively spreading the discharge from large flow events (e.g. ponds where large flows are collected and treated over time after a large acidity flushing event).
- **use of rapid response treatment systems** – Design systems that can quickly accommodate changes in water volumes and acidity loads (i.e. active treatment units).
- **redundancy** – Combinations of treatment systems with differing treatment capacities provide overlapping and complementary treatment benefits, affording some capacity to treat both smaller and bigger volumes over many years.

Note that the above list is not exhaustive and may not be appropriate for all situations.

6 In-drain treatment options

This section outlines treatment options suitable for treating acidic water within drainage channels. Important elements of design and construction are listed together with considerations for managing the treatment units.

Guideline information on performance of the treatment options, extrapolated from local trial examples, should help understand what might be achieved using each type of treatment approach. Note that the extent to which different treatment options will neutralise and remove minor elements from drain water can vary and not all of the options will achieve complete treatment. Case studies of pilot treatment units are outlined to give users some insight into their local design and performance.

6.1 Lime-sand beds

6.1.1 Principle

This treatment involves low cost, passive neutralisation of acidic saline water by direct application of lime-sand to drainage channels, achieving treatment at source. Lime-sand beds are a very simple version of an oxic limestone drain (e.g. Skousen et al. 1998; Cravotta & Trahan 1999) but differ in using an existing drainage channel where flow velocities are likely to be much lower.

6.1.2 Design and construction considerations

i. Water depth: The depth of water within drains is likely to influence the effectiveness of lime-sand beds. Application of lime-sand to sections of drains with shallow flows and few pools will promote greater contact between water and lime-sand and minimise the risk of burial by sediments deposited during high flow events.

ii. Lime-sand quality: Treatment performance will be greatest using high-grade, small-grained carbonate materials (assaying as >90% calcium carbonate). Grain size has a greater effect on efficiency than purity (Sherlock 1997) so the poorer performance of lower grade materials may be offset by crushing. Materials with smaller grains (<1 mm) will sustain greater rates of reaction than materials with larger particles (generally, most coarse crushed limestone). However, unless continuously stirred or regularly disturbed, materials with smaller particles may form crusts and begin to seal over, reducing the reactive contact area between water and neutralising materials. With larger particles (e.g. >1 cm diameter), the contact area may be maintained. In leveed drains, horizontal flow velocities are usually limited (due to bed gradients of <0.01%) and there is little risk of finer lime-sand being eroded. In unleveed drains, there is a greater risk of high flow velocities and larger coarser lime-sands or aggregate limestone may need to be considered to avoid erosion within drains.

Crushed dolomite, which contains a mix of magnesium carbonate ($MgCO_3$) and calcium carbonate ($CaCO_3$), is only recommended for use if high assay lime-sand or limestone are not available. Crushed dolomitic limestones dissolve much more slowly than calcium carbonate materials (Cravotta et al. 2008). Trials using this in active treatment situations have found dolomitic limestone materials to be much less effective than calcium carbonate materials (Cravotta et al. 2008; Green et al. 2007). Further information can be obtained from the significant body of literature in Europe and the USA regarding the selection and use of limestone for treatment of acidic waterways (e.g. Sverdrup 1983; Olem 1991; Menendez et al. 2000; Donnelly et al. 2003).

iii. Lime-sand placement: Placement of lime-sand evenly along the length of drains may achieve greatest effectiveness in situations where bank erosion and silting or sedimentation is expected to be minimal. Where silting or sedimentation is likely to be significant (probably in most drains), greatest effectiveness is likely to be achieved by placing lime-sand in shallow-flow sections (i.e. sections of drain with step-changes in gradient) or by forming sand riffles in ponded sections (see Beacon lime-sand bed trial, Section 6.1.5). Riffles are small dams over which water cascades (see lime-sand bed trial). Although lime-sand can be loaded using a front-end loader (Fig. 13) bunds along many drains will make this difficult and it may be best to load sand with a side-casting conveyor belt or sand auger to reach over drain walls.

iv. Capture of precipitates: Neutralisation of acidic drain waters by lime-sand will result in the formation of iron- and aluminium-rich sludges that ideally are retained within the drainage system. Pools within drains may facilitate their retention within a drainage system. However, a settling pond (see Outflow oxidation and settling ponds in Section 7.2.2) may be required at the drain discharge point, particularly for unleveed drains where there is a risk of these sludges being flushed from the drain during high flow events.

Figure 13 Loading lime-sand into a drain using a front-end loader: note the opening of the levee banks to allow access

6.1.3 Operational management

Although lime-sand beds are intended to be passive treatment systems and should not require regular maintenance, many will probably benefit from the following:

i. Crust removal: Periodic physical disruption of the lime-sand bed to break iron oxide/gypsum crusts on the surface of the sand beds can increase effectiveness. This will need to be repeated regularly since crusts tend to reform within days.

ii. Sludge management: As outlined in the design and construction considerations, sludges are best retained within the drains or captured at the end of a drain in a settling pond. Periodic removal may be required. This may be done during normal drain maintenance procedures and must be contained on site (out of the drain) to avoid re-release of adsorbed metals or disposed of safely. Since these are often low density materials, retention is likely to best achieved in drains where flow velocities are kept low.

iii. Redosing: Reapplication of lime-sand is expected to be required infrequently. For example, in the case study at Beacon (see below), the 30 tonnes added to this drain would not need replacing for 3–6 years, given rates of lime-sand reaction in the order of 300–500 g calcium carbonate per lineal metre of drain per day (assuming a lime-sand riffle length of 43 m). If the same 30 tonnes of lime-sand were applied to form a layer no more than 50 mm thick (i.e. lining 328 m of drain), additional lime-sand may need to be reloaded at least every 6 months, although the layer may be buried by silt much faster, decreasing the neutralisation rate of lime-sand.

6.1.4 Effectiveness

The application of lime-sand directly to drains (i.e. with no other materials) is very unlikely to achieve 100% treatment of Wheatbelt acidic drain water over days to weeks or in the long-term (months to years). Neutralisation of water with high concentrations of dissolved iron can result in the formation of iron oxide crusts combined with precipitation of gypsum (i.e. armouring) within weeks (see Beacon drain case study below). The formation of this thin barrier greatly slows the rate of neutralisation by the lime-sand (even though reactive lime-sand remains).

Despite the formation of crusts, the lime-sand remains partially reactive and can have a low-level long-term effect on acidity. This effect may be additive over long sections of treated drains (>1 km) and is likely to contribute to an overall reduction in acidity loading for an entire drainage system. Nonetheless, this treatment approach should not be relied on as a primary method of treatment but instead will require integration with other methods (see Section 8). An unquantified advantage of applying lime-sand directly to aged drains (older than one year) will be the treatment of acidity within the drain sediments. These muds often contain at least ten times the acidity of drain water, though are less reactive. This acidity store may ‘leak’ back into the drain water (Fitzpatrick et al. 2008) or add to the loads from drains during rainfall events.

6.1.5 Case study: Beacon lime-sand bed trial

i. Site: This was a 210 m section of the Beacon Engineering Evaluation Initiative drain on Tim Cashmore's property immediately downstream of the Scotsman Road culvert, Beacon, Western Australia (see location map in Appendix D) that received inflow from 6.5 km of drainage channel.

i. Construction: Thirty tonnes of coastal lime-sand was loaded into the drain using a front-end loader at six points along the 210 m section of drain. Based on the baseflow and acidity concentrations at the drain inflow being a maximum of 100 kg $CaCO_3$ acidity/day, the lime-sand was expected to provide at least 300 days of treatment. Because of the limited reach of the loading bucket the sand had to be spread manually within the drain base. Sand was loaded to form six flat-top riffles of 0.3–0.45 m depth with a collective length of 43 m (individual riffle lengths being 2.5–13 m). The riffles effectively formed small benches over which the drain water flowed as a thin sheet during low flows, maximising water–sand contact. The lime-sand used for the trial was sourced in bulk from the AgLime Australia coastal lime-sand pit at Cervantes and contained 91% calcium carbonate by weight with over 80% of particles 0.125–0.250 mm diameter.

iii. Monitoring outline: Inflow and outflow water were sampled at days 1 and 8 after treatment then fortnightly for several months. Water-quality measurements included on-site pH, electrical conductivity (EC), redox potential, alkalinity and acidity with samples taken to measure concentrations of major ions, metals and other elements (see Appendix A). Net acidity was calculated from metal concentrations, pH and alkalinity (see Section 3.2.1 & Appendix B). Flow through the treatment section, inflow and outflow pH and EC of the drain water were also monitored at 5 minute intervals using automated measurement and data-logging systems for several weeks before treatment and for 12 months after treatment. Inflow was gauged using a 30 degree V-notch in a plastic weir secured to the inlet of a 600-mm culvert (at Scotsman Road) and coupled with a Mindata water-level measurement system. Plastic mesh cages were constructed around the weir inlet and upstream to protect the site from blockage by the large amounts of organic debris that can blow into and wash down the drain.

iv. Performance results: Immediately after the addition of the lime-sand, large quantities of aluminium precipitates formed and were trapped on vegetation blown into the base of drain (mostly *Salsola spp.* or roly-poly weed). One day after addition, water flowing through the treatment section had increased in pH from 3.3 to 4.4, with net acidity decreasing by 35% from 0.46 g to 0.29 g $CaCO_3$/L (Fig. 14) along with similar decreased concentrations of dissolved aluminium and iron. The lime-sand had no effect on the minor metal concentrations, most probably because the pH of the water was not raised enough to remove them.

While some differences between inflow and outflow water quality persisted after eight days, by three weeks the treatment effectiveness had decreased to less than 4% of the acidity load. Extended monitoring over subsequent months found that the effect of the lime-sand on water quality diminished to being indistinguishable after 100 days (Fig. 14).

Reactive lime-sand was still present in the drain although separated from the overlying acidic water by a 1–2 mm dark red iron oxide-rich crust (Fig. 15). This had formed within three weeks of the lime-sand addition despite the lime-sand being formed into riffles over which there was turbulent flow.

Analysis of the crust indicated that it was rich in iron and aluminium though it did not contain significant concentrations of lead, copper, zinc, nickel or selenium. The crust also contained high concentrations of sulfate, most likely due to formation of a calcium sulfate mineral (possibly gypsum) during the neutralising process.

More focused monitoring of water quality in individual riffles 12 months after the commencement of the trial showed that some degree of neutralisation was occurring within each riffle. On-site measurement of acidity over two riffles indicated that as drain water flowed over these there was a slight rise in pH of 0.2–0.3 units and a decrease in acidity of average 0.02 g $CaCO_3$/L for flows of 1.75–3.3 L/s. This was likely to be the maximum neutralisation rate, since, during higher flows after rainfall, thin silt layers which will slow neutralisation rates can be deposited on the lime-sand riffles. Although slight, the neutralisation effect amounts to 3–5 kg lime-sand per day for each riffle and could conceivably equate to the order of tonnes of acidity treatment per year. Similar low-level neutralisation benefits have also been reported in other limestone drains despite significant armouring (Skousen et al. 1998).

Long-term, continuous, low-level acidity reduction might be significant if lime-sand was applied to significant drain lengths and if used in a combined treatment system (Section 8).

Take home message: Lime-sand application offers a low-cost approach to reducing the acidity prior to final treatment by other more expensive measures.

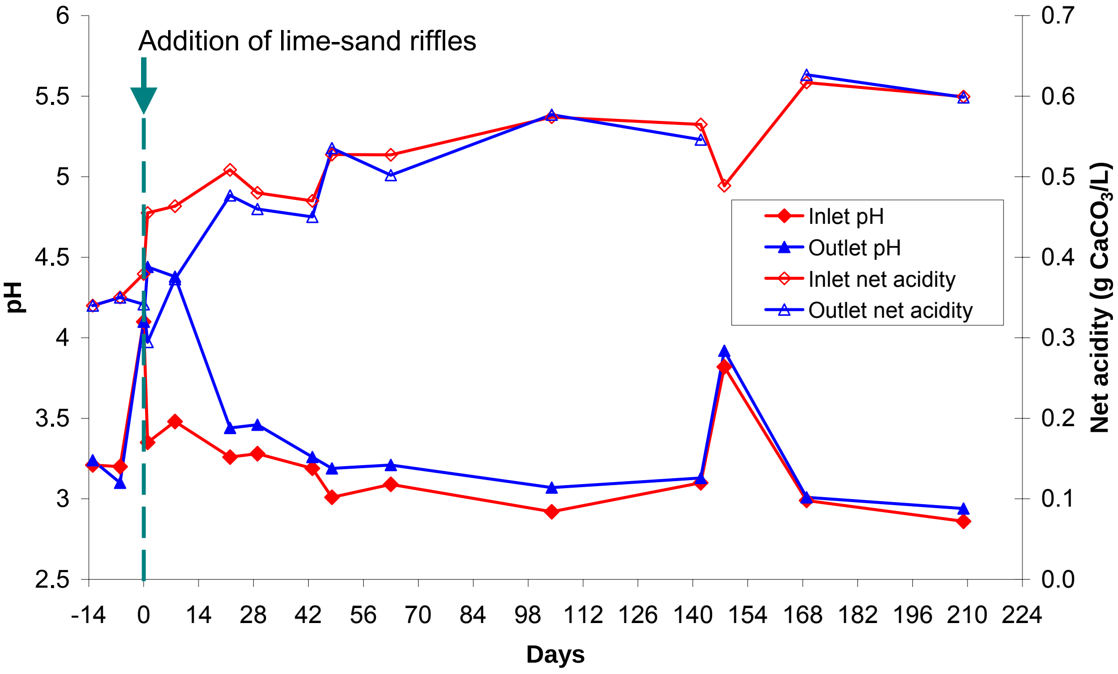


Figure 14 Net acidity and pH in inflow and outflow water of a 210-m drain section treated with lime-sand

Figure 15 Iron oxide crust formation on lime-sand implicated in slowing neutralisation rates

6.2 Subsoil carbonate beds

6.2.1 Principle

Carbonate-rich subsoil horizons removed during drain construction can provide a low-cost source of neutralising materials for use in direct application to drainage channels or bank lining. This option is a variation of lime-sand beds.

Significant concentrations of carbonate materials (see Table 2) can occur in some subsoil horizons in areas with acidic groundwater (Fig. 16). The total amount of carbonates in excavated subsoils is estimated to be between at least 7 and 745 kg per metre of drain length, based on an excavation of a 2.5 m deep drain with a base width of 1.2 m and walls with 2:1 gradient (see Table 2). These carbonate materials may occur in horizons of 0.5–1.8 m thickness, often above the watertable, but vary widely in reactive carbonate concentration. The quantities of neutralising materials within the excavated soil horizons have the potential to offset some of the acidity in groundwater discharge into the base of the drains. Acidity seepage into the base of drains can be as much as 51 g $CaCO_3$ acidity per metre of drain per day (see Table 3; Section 6.3) which requires about 186 kg of carbonate for neutralisation over a 10-year period. Some drains potentially contain this amount in the subsoil horizons excavated during construction. However, these calculations may be confounded by the fact that the carbonate materials are not likely to dissolve and react with drain water at a rate comparable to that at which acidity is generated.

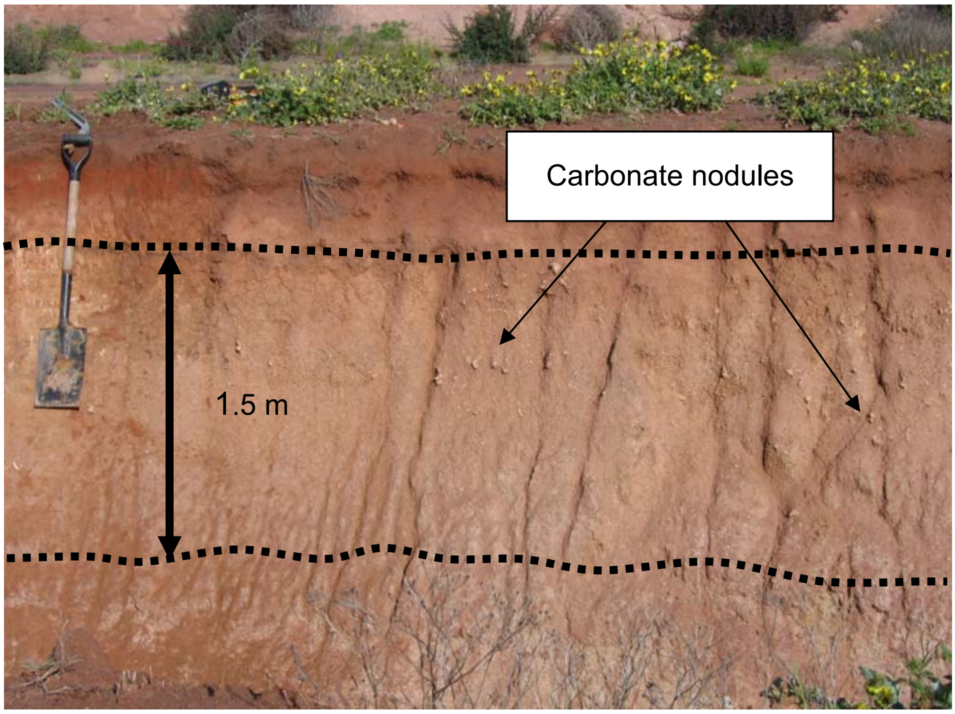


Figure 16 An exposed profile on the edge of a drain showing subsoil horizons rich in fine carbonate materials (<2 mm diameter) and carbonate nodules (2–20 mm diameter)

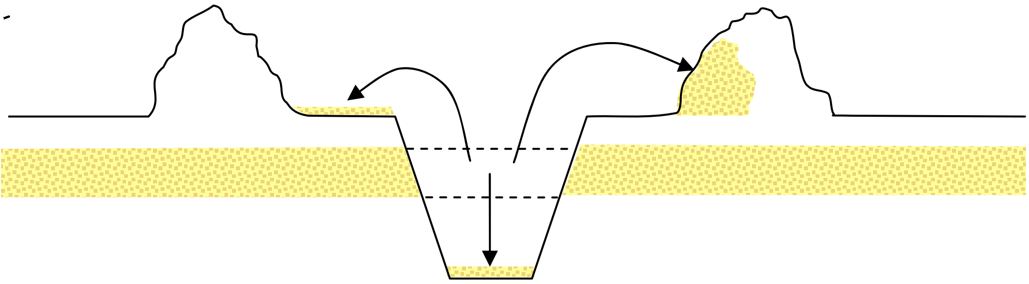


Figure 17 A schematic cross-section showing options for placing excavated alkaline subsoils for use in drain spoil, to enhance alkalinity in runoff from berms or directly in drain base

Table 2 Estimated amounts of neutralising materials (as subsoil carbonates) potentially available per metre of drain in the north-eastern and eastern Wheatbelt

Locality (Road/Landholder)	Carbonate content (% <2 mm)[1]	Total thickness of horizons with carbonates (m)	Estimated kg $CaCO_3$ in excavated soil per metre drain length (<2 mm)[2]
Pithara (Birdwood Rd)	0.8–0.9 0.9		30
Doodlakine (Wallatin Rd)	0.1–3 0.65–0.7	5	7–58
Doodlakine (Wallatin Rd)	5–10* 0.5–1.2		105–470
Beacon (Kirby)	0.4–33 1.5–1.8		270–550
Beacon (Scotsman Rd)	5–20* 0.8–1.1		170–740
Elachbutting 1.3–3.9		1.1	120
Ejanding 1.2–11		1.1	270

[1] Carbonate determined on <2 mm soil fraction by HCl digestion (Rayment & Higginson 1992). The <2 mm soil fraction is considered to be rapidly reactive carbonate though larger carbonate fragments (2–10 mm) may play a role in neutralising acidic water and sediments over the long term.

[2] Estimated as weight of carbonates likely to be deposited in drain spoil per metre of drain calculated using depths and thicknesses of subsoil horizons containing carbonates for each site, a carbonate particle density of 1.6 g/cm^3 (to convert material volume to weight) and based on construction of a 2.5 m deep drain with base width of 1.2 m, side wall gradient of 2:1.

* Carbonate content includes nodules up to 10 mm diameter.

6.2.2 Design and construction considerations

i. Pre-construction tests: Carbonate-rich layers need to be identified while digging test-pits before drain construction. Adding a few drops of hydrochloric acid (or battery acid) can be used to test whether fine-grained carbonates are present. If there is immediate fizzing carbonates are present.

ii. Handling and placement: During construction, remove carbonate-rich layers and selectively place in drain spoil (e.g. all on one side of the drain or on the inside part of levee banks) followed by replacing as a surface lining on berms and in the drainage channel (Fig. 17). There may be some neutralisation benefit by forming the carbonate soils as riffles in the drain base (Section 6.1.5) rather than lining the whole length of the channel. Placing unused carbonate horizons on the top of levee banks will enable them to be used for future encapsulation and neutralisation of any acidic muds cleaned from drains.

iii. Capture of sludges: As with lime-sand beds, retain in the drain aluminium- and iron-rich precipitates formed during neutralisation.

6.2.3 Operational management

Requirements are similar to those needed for lime-sand beds.

6.2.4 Effectiveness

The effectiveness of subsoil carbonate beds is unknown. The materials may at best provide long-term low-level neutralisation similar to that of lime-sand directly applied to drains (Section 6.1). If used to line the area within the levees and the drain channel, the materials may increase the neutralising capacity of runoff into drains and reduce acidity loads during high flows. This already partly occurs in some drains as runoff water erodes carbonate materials from drain walls during rainfall events, resulting in partial neutralisation of drain water (see Section 4.2). Lining the drainage channels should also reduce acidity loads during these events by reducing the acidity of the muds accumulating in the drains.

A major impediment to the effectiveness of this treatment option is the reactivity of the carbonate materials in the subsoils. Compared with other neutralising materials such as lime-sand, their reactivity is likely to be much lower due to larger particle size, lower percentage in soils and natural soil armouring of carbonate concentrations by gypsum and iron oxides.

Take home message: Subsoil carbonate horizons present a very low-cost, readily available resource that can potentially be used to *partially* off-set drainage acidity, though the effectiveness is likely to be low. This treatment approach should not be relied on as a primary method of treatment but instead requires integration with other treatment methods (see Section 8).

6.3 In-drain composting beds

6.3.1 Principle

This option involves constructing composting beds within drains using locally available sources of organic matter that encourage neutralisation of acidic water and trapping of dissolved metals through the microbial-driven process of sulfate reduction.

The treatment approach is dependent on the use of organic matter combined with permanent waterlogging to establish and feed a microbial community within which sulfate-reducing bacteria flourish. These bacteria grow and function best under high pH (5–8) conditions and reducing conditions (i.e. low redox conditions, see Glossary). The bacteria create alkalinity by reduction of sulfate (present in all saline water) to hydrogen sulfide, commonly known as rotten-egg gas (see Neculita et al. 2007 for a subject review). The hydrogen sulfide reacts with iron (dissolved and solid forms) to form iron sulfides visible as black oozes (Fig. 18). These sulfides effectively store acidity *as long as the materials remain in a low oxygen, waterlogged environment and are not flushed downstream*.

Dissolved metals such as copper, lead, cadmium, nickel and zinc readily precipitate from water as sulfide minerals (Gazea et al. 1996; Neculita et al. 2007), commonly occurring within black oozes and sludges in or beneath waterlogged organic layers. Soluble manganese is not generally removed from water by treatment wetlands since this metal rarely forms stable sulfide precipitates.

Figure 18 Black oozes containing iron sulfide minerals formed in waterlogged organic matter as a result of sulfate reduction by bacteria in low oxygen conditions

6.3.2 Design and construction considerations

Key aspects for the design of in-drain composting beds include understanding whether treatment capacity can be matched to acidity load, selection of appropriate organic mixtures, and placement within the base of drains. For best success it is recommended that composting drains are constructed starting at the beginning of drains rather than at the discharge point. Although there are some uncertainties in the design of composting beds, local experience indicates even when the treatment capacity of composting beds is overwhelmed by drain acidity load these can still achieve reduction in acidity discharge from drains (see case study below).

i. Sizing for acidity loading: In-drain composting beds need to be sized (and composed) according to expected rates of acidity loading from water discharging into the composting bed. This can be water entering directly beneath where the organic materials are placed, as well as any acidity expected from upstream sections if composted beds are concentrated in sections (see Section 6.3.2 iv. Organic matter placement).

The treatment capacity of all composting beds is directly dependent on the rate of microbial sulfate reduction, mostly carried out by bacteria. This rate, in turn, is dependent on the rate of organic matter decomposition and the release of sugars and short-chain organic acids (that feed sulfate-reducing microbes) from the organic matter and indirectly as decomposition by-products of other organisms consuming the organic matter.

If the acidity load exceeds the neutralising capacity (i.e. system is overloaded) the treatment capacity can be reduced (Fig. 19). This commonly occurs when the overall pH of the organic matrix falls below approximately pH 5, restricting the zones where sulfate-reducing bacteria can flourish, which results in overall reduced rates of sulfate reduction and reduced capacity to neutralise acidity (Fig. 19).

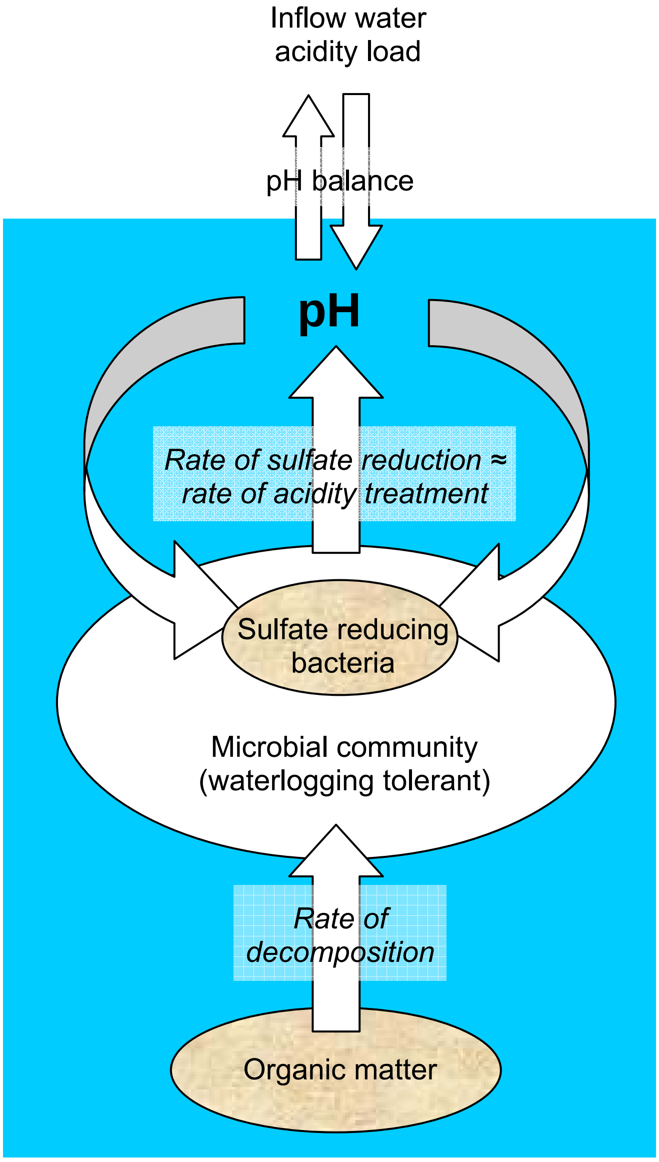


Figure 19 pH feedback loops affecting sulfate-reducing bacterial function (and acidity treatment capacity) and the link between organic matter decomposition and sulfate reducing bacteria activity

Although sulfate-reducing bacteria can survive at low pH, they do not function at a rate sufficient to raise the system pH (Doshi 2006). In a worst-case scenario, a significant fall in pH (to less than 4) can also result in acidity accumulated as iron sulfides (black oozes) and other metal sulfide sludges redissolving in the water and beginning to leak out of the treatment system.

The risk of overloading can be minimised by ensuring that the **size and composition** of the composting beds match the **expected** acidity inflow via groundwater. Over-design is better than under-design. Although the size of composting beds is limited by the area in the base of drains, using good quality organic mixes can offset this limitation to some extent. It may also be possible to reduce the risk of overloading affecting the treatment capacity by designing the system to allow acidic water to bypass or flow over the treatment zone. Some of these elements are discussed in detail below (see Section 6.3.2 iv. Organic matter placement).

Composting beds lining the complete length of drains are likely to achieve full treatment of water in the medium term (i.e. 2–3 years) with average acidities of no more than 0.6 g $CaCO_3$/L and flows less than 0.25 L/s. The longevity of treatment is likely to be less if composting beds are concentrated in sections in the base of drains, where each section treats the acidity load from groundwater discharge within the section and acidity load flowing in from untreated sections. Estimated rates of acidity loading from groundwater discharge per metre of drain are in the range 4–43 g $CaCO_3$ per m^2 of drain base per day assuming average acidity (0.6 g $CaCO_3$/L in the Wheatbelt, see Fig. 4) and a range of groundwater inflows (see Table 3). These figures provide an indication of the acidity loads that composting beds will need to treat to be fully effective. Composting wetlands (using composting beds) have been reported to handle similar ranges of acidity loads: 2.5–48 g $CaCO_3$ per m^2 composting bed area per day (Ziemkiewicz et al. 2003; Younger et al. 2002). However, to ensure that a composting treatment system effectively treats all acidity in drain waters over the long-term (>several years) it is internationally recommended that treatment rates (and therefore acidity loading) should be no more than 3.5 g $CaCO_3$ per m^2 composting bed area per day (Hedin et al. 1994; PIRAMID Consortium 2003). Clearly this is not possible in drains where there is limited area and no control over acidity loads but, as discussed earlier, this may be offset by using higher quality organic mixtures to achieve higher treatment rates per unit area over a shorter period.

A trial at Wallatin Creek (see case study below) found that composting treatment with a minimal mix of organic materials (5% by volume of sheep manure with straw) achieved treatment rates up to 21 g $CaCO_3/m^2$ composting wetland area/day in 3–4 months after adding organic matter, declining with time to no less than 2 g $CaCO_3/m^2$ composting bed area/day after 12 months (see case study below at Wallatin Creek). Adding particular types of non-cellulosic material in the form of waste grain could achieve treatment rates of up to 25 g $CaCO_3/m^2$ composting bed area/day.

Composting beds may function for short periods (<1 year) in drains with acidity loads of more than 20 g $CaCO_3/m^2$ composting bed area/day (see Table 3 for comparison) with high quality organic mixes covering the full length of drain. In any event, where overloading occurs, most will probably treat less than 50% of the acidity load but be less effective in removing dissolved metals and other elements (see Section 6.3.2 v. Incorporating neutralising materials).

Table 3 Estimates of expected acidity loads per metre drain length and per square metre of drain base for a range of groundwater discharge rates with 0.6 g acidity/L

Groundwater discharge rate to drain[1] (L/km/s)	Daily[2] groundwater discharge per m of drain (L/m/day)	Daily acidity load per m of drain[3] (g $CaCO_3$/m/day)	Daily acidity load per m^2 of drain base[4] (g $CaCO_3/m^2$/day)
0.1 8.6		5.2	4.3
0.25 21.6		13.0	10.8
0.5 43.2		25.9	21.6
0.75 64.8		38.9	32.4
1.0 85.4		51.8	43.2

[1] Range of groundwater discharge figures for drains in WA (pers. comm. Nick Cox; Cox & Tetlow in press). Note that drain discharge normally measured in drains can be less than this rate since water is commonly lost through evaporation and sometimes seepage back into the ground. During and after rain, drain flows can be much greater than rates of groundwater discharge.

[2] Assuming no variation in groundwater discharge throughout a day

[3] Assuming a total acidity of 0.6 g $CaCO_3$/L. This is the median total acidity of shallow groundwater sampled at 116 sites across the Wheatbelt (see Fig. 4). To estimate acidity loads for drains with half or double this average acidity, halve or double the calculated acidity load.

[4] Calculated for a drain base width of 1.2 m

ii. Organic matter fuel: A mix of good quality, easily decomposed organic materials (non-cellulosic) and more cellulosic (woody), slower decomposing materials have been reported from international studies to provide the best results for composting treatment systems (Younger et al. 2002; Zagury et al. 2006; Neculita et al. 2007). These mixtures are needed as an energy source for sulfate-reducing bacteria and to support a community of other microbes that help break down the organic matter under waterlogged conditions. This greater community in turn can also become a feed-source for the bacteria that carry out the sulfate reduction; important in powering these treatment systems over the long term.

Mixtures of at least 10% by volume of non-cellulosic (easily decomposed) materials with cellulosic materials appear to perform best (Neculita et al. 2007). A mixture of more than 50% by volume of easily decomposed, non-cellulosic material results in decreased performance, mostly due to poor porosity (Neculita et al. 2007) and a greater likelihood of bacterial methane production. In the Wheatbelt case study discussed below, 5% composted sheep manure with straw resulted in reasonable treatment rates over a 12 month period.

The quantities of organic materials must be calculated on the final volume and not by volume of delivered material. For example, 0.5 tonne straw bales have a compacted volume of approximately 3.5 m^3, but a final expanded volume of approximately 6 m^3. Non-cellulosic materials can also help in holding the organic mix in place since higher quality organic materials are generally less dense and can generate gases causing the mix to float.

Good quality, faster decomposing materials include fresh plant matter such as hay, silage, grass/weed cuttings, composted sheep or cow manure, green wastes (if these are primarily composed of annual plants), food wastes and sewage sludges (Neculita et al. 2007). High starch materials such as malting waste, wet or sprouted grain, pinched grain or waste seed (e.g. lupins, wheat, barley or screenings) are also likely to be good non-cellulosic materials. These contain high concentrations of available sugars and can easily be decomposed but must be incorporated with less decomposable materials to ensure longer periods of operation.

Cellulosic, slower decomposing materials include wood chips, sawdust (Neculita et al. 2007), straw, stubble residues, harvester chaff and shredded tree branches (including oil mallee distillation residues).

Internationally, composted mushroom waste with added gypsum is highly recommended (Hedin et al. 1994; Younger et al. 2002), probably because of the alkalinity remaining after composting or the addition of gypsum. The residual alkalinity contained in composted materials can be also be supplemented by mixing in 2–5% calcium carbonate (as lime-sand, crushed limestone or subsoil carbonates).

Recent investigations with local materials highlight that the choice of slower decomposing, cellulosic organic matter can have a significant effect on the microbial treatment activity that these can support. Bench-top bucket tests using composted sheep manure mixed with locally available organic materials such as wheat stubble, harvest chaff residues, composted lupin stubble or oil mallee distillation residues found that chaff and composted lupin mixtures (with composted sheep manure) supported the highest rates of treatment in the month after set-up (Table 4; Santini et al. 2009). Initial treatment rates with these residues were more than five times that of the oil mallee or wheat stubble mixtures. This highlights there can be some benefits from using better quality, cellulosic organic materials, even if only of marginally better quality.

Acidity treatment rates can be increased by adding dilute ethanol to stubble-composted sheep manure mixes (Table 4) indicating that treatment capacity for composting wetlands could perhaps be boosted by feeding with mixtures of solutions of sugars (e.g. molasses dissolved in water) or alcohols (e.g. low-grade methanol). This is not likely to be practical over the longer term and may be more expensive than simply building a larger wetland, but may provide a way to boost treatment capacity during periods of large acidity loading to wetlands.

A final consideration for selecting organic materials is the hydraulic conductivity of the final mix which determines whether water can flow through the materials. This generally depends on fragment size and compaction during construction.

Table 4 Comparison of the short-term (<30 day) acidity treatment potential for different mixes of locally available organic materials (after Santini et al. 2009)

Organic matter mix[#]	Relative rate of acidity treatment potential over one month[*] (compared with wheat stubble + composted sheep manure)
Wheat stubble + sheep manure	1
Oil mallee distillation residues & sheep manure	1
Harvest trail chaff (wheat) & sheep manure	3–4
Composted lupin stubble & sheep manure	5–9
Wheat stubble & ethanol & sheep manure	10–11

[#] All organic matter mixes consisted of composted sheep manure at 8% (by volume) with the balance made up by the residues, except for the wheat stubble & ethanol mixture where ethanol was fed into inflow water at a rate of 1.1 g/L.

[*] Acidity treatment potential based on estimation of rates of sulfate reduction (expressed as millimoles sulfate reduced per 1000 cm^3 per day; see Santini et al. 2009). An average rate of 0.01 millimoles sulfate reduced per 1000 cm^3 was used for the wheat stubble.

iii. Water level requirements: The efficiency of operation of composting treatment beds relies on flooding and permanent inundation of the organic mix with the choice of site depending on the number and length of pools in drains. Pools provide opportunities to maximise flooding of organic materials with little change in the drain water levels. Step-changes in gradient common in many drains often provide a series of pools interspersed with shallow flowing sections. Flow *over* the organic mix is preferred to flow *through*, although some flow will always occur through the base (due to groundwater discharge) and through shallow layers.

Installing composting beds will require sacrificing up to 0.25 m of effective depth in some drains (with no pools). Alternatively, drains could be constructed 0.25 m deeper (with vertical walls in the bottom 0.25 m) to allow for the treatment beds or sections of the drain constructed deeper with treatment occurring in, for example, 10–20 m sections as discussed below (see iv. Organic matter placement). This may appear to impose additional costs on drain construction but the treatment area gained within the drain base deepened an additional 0.25 m requires less digging than the equivalent area at the end of a drain. For example, deepening by 0.25 m allowing use of composting beds gains approximately 1200 m^2 of treatment area per km drain (for removal of 300 m^3 soil) whereas a similar sized composting wetland at the end of a drain might require excavation to more than 1 m depth requiring removal of at least 1200 m^3 soil (see Section 7.2.2 Design and construction considerations).

The final water levels in the drains should allow for flow over the organic matter during high flow events. Composting beds are not recommended for unleveed drains or drains where there is a high risk of large flows.

iv. Organic matter placement: Organic materials need to be up to 0.25 m deep in the base of the drains, though depth can be as little as 0.1 m interspersed with deeper filled sections. Depending on acidity loading rates and types of organic materials used, the full length of drain channels may need to be filled or organic materials placed in sections (Fig. 20). Using 'core-cells' of organic materials, where better quality organic matter is concentrated in drain sections, is more likely to be successful than better quality organic material spread and diluted along a drain.

Materials are best placed as layers with the more easily decomposed organic materials in the base overlaid by more resistant organic materials (Fig. 21) to promote the greatest fuelling of sulfate reduction in the zone closest to the base and facilitate retention of the sulfides formed in the underlying basement soil materials. The layering also provides a measure of protection from acid overloading in that the growth of the sulfate-reducing bacteria is concentrated in bottom materials allowing the passage of acidic water exceeding the treatment capacity of the system over these. Thicker organic layering might be used to concentrate the treatment in sections, though would need to ensure that the organic materials are flooded and remain waterlogged during treatment.

Organic materials may also be arranged along the channels as alternating sections. For example, materials may be arranged in sections of mainly readily decomposed, non-cellulosic organic matter interspersed with sections of mainly slower decomposing, cellulosic organic matter (e.g. 10–20 m non-cellulosic material every 150 m). This will also make it easier to top up with fresh materials in later years. Although the sections of less decomposable organic materials may not provide organic materials that support sulfate-reducing bacteria, these will nonetheless assist by removing oxygen from the water and maintaining low-oxygen conditions.

Compaction of the organic materials may be needed after placement to ensure low-oxygen zones can form within the organic mix and promote microbial sulfate reduction, particularly if the materials are packed loosely on initial loading.

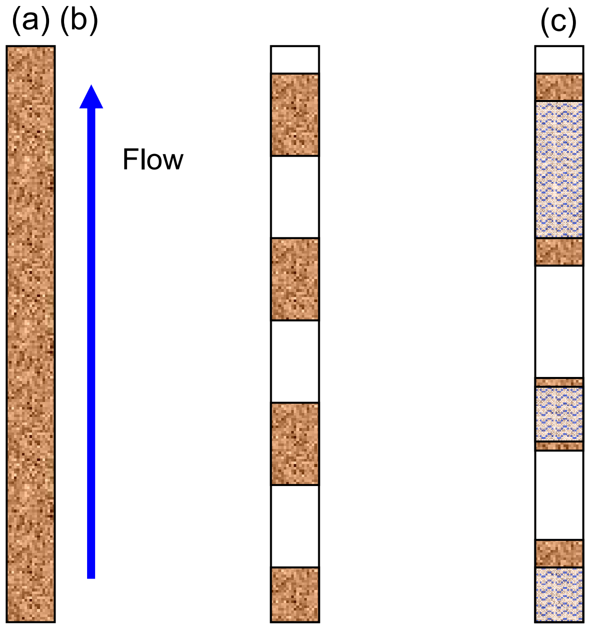


Figure 20 Plan views of organic material placement along drains in composting bed design: (a) full drain length, (b) regularly spaced sections and (c) concentrated and aligned with ponds (blue waves) selected after drain construction

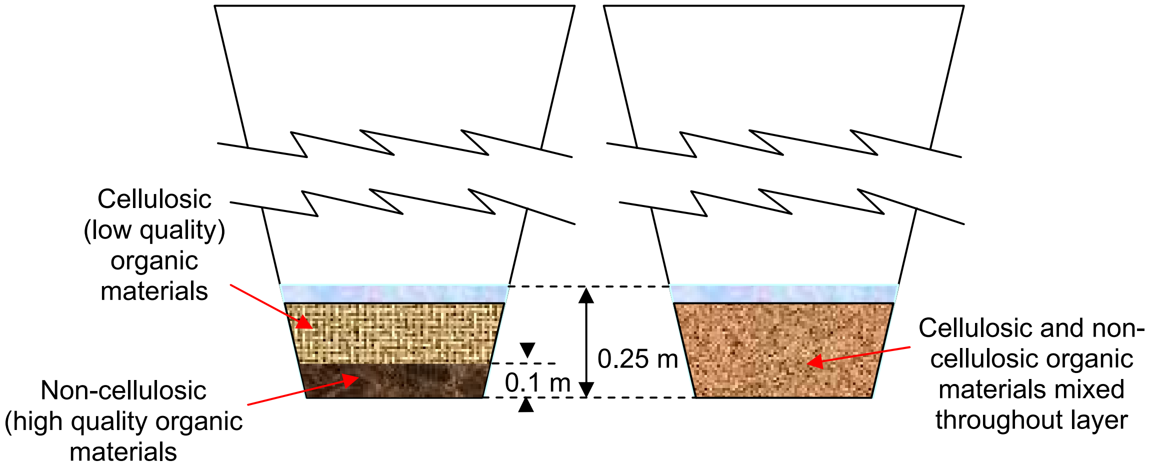


Figure 21 Cross-section views of examples of vertical placement arrangements of organic materials in composting bed design

v. Incorporating neutralising materials: Improved treatment rates may be achieved by incorporating lime-sand, limestone or subsoil carbonates (see Section 6.2 Subsoil carbonate beds) to protect bacterial-sulfate reducing zones from acidification.

This design has the advantage over the in-drain application of lime-sand or soil carbonates of containing the carbonate minerals in conditions that prevent armouring by iron oxides (PIRAMID Consortium 2003; Cravotta & Ward 2008). The effectiveness of the neutralising materials should be much greater (at least initially) and will promote the establishment of the microbes that carry out sulfate reduction.

Long-term effectiveness depends greatly on whether water can continue to flow through the layers of neutralising materials, since elsewhere in the world the formation of aluminium precipitates has been widely reported to cause problems where carbonate neutralising materials are used in low-flow systems (Gusek & Wildeman 2002). Such approaches are not recommended for treating water with more than 2 mg aluminium/L (Younger et al. 2002; PIRAMID Consortium 2003) – commonly encountered in many acidic Wheatbelt drains.

vi. Flow restriction risks: In-drain composting beds must be designed to ensure that organic materials do not become dislodged and move downstream during high flows.

This risk may be offset by ensuring the materials are compressed during construction or a matted straw layer is placed over the surface of finer materials that might be dislodged when flows increase. Ensuring that the organic matter is placed to a depth that allows some ponding will allow flow over the straw, thereby providing some protection to the capacity of the treatment beds. Additional protection can also be achieved by placing plastic mesh at intervals in the drain to trap any organic materials flushed downstream in high flows. Design minimising flow within drains (e.g. use of levee banks) will also greatly reduce the risk of organic matter being flushed down drains.

6.3.3 Operational management

i. Start-up time: There may be delays of weeks to months before microbial growth and activity build up to a level that effectively treats acidity. This must be considered when managing acidic discharge from a drainage system.

ii. Organic sludge and sediment management: Composting bed materials contain acidity that has been removed from drain water (accumulated and stored mostly as iron pyrite minerals). These materials will need to be contained and treated when excavated from the drains during maintenance.

The data in Table 3 indicate how much potential acidity might accumulate in sulfide minerals within in-drain composting beds if 100% successful in treating acidity. Over 10 years, most drains would accumulate at most 19–189 kg of acidity per metre of drain treated (listed in Table 3 as equivalent $CaCO_3$ per metre of drain). If the sediments are removed, these will require 28–284 kg of calcium carbonate per metre of drain treated to neutralise the acidity every 10 years (incorporating an over-liming safety margin of 50%). These materials could be allowed to dry and be slowly neutralised by burial within drain levee banks containing carbonate-rich subsoils (see Section 6.2 Subsoil carbonate beds).

Where composting beds are concentrated in sections, the acidic storage per metre of drain may be greater as acidity generated in non-composting sections is concentrated within the composting sections. Accordingly, these will need to be encapsulated in materials with higher carbonate concentrations.

iii. Organic matter recharging: It will be necessary to periodically recharge the organic materials in the composting beds, depending on treatment performance.

Current experience (see case study below) indicates that this will depend on the mix of organic materials initially loaded into the system; in particular, the easily decomposable (non-cellulosic) component of the organic mix. It will not be necessary to top up the slower decomposing materials such as straw as frequently.

6.3.4 Effectiveness

i. Short term (<6 months): Composting beds in drains have a high likelihood of successful treatment over the short term (if sized according to flows) with up to 100% treatment of acidity and removal of a variety of minor metals (e.g. Pb, Ni, Zn, Cu). The greatest risk of failure arises from using a poor organic mix containing insufficient non-cellulosic materials or acidity loading much greater than expected, resulting in acidification of organic materials in treatment sections.

ii. Long term (>6 months): The long-term performance of composting beds in drains is unknown, though would be expected to decrease over time.

A trial in WA indicated that effectiveness can be expected to fall to less than 50% of the acidity loading after 6 months if insufficient high-quality organic materials are used at construction (see case study below). Poor treatment can also result in limited capacity to trap some minor metals. Despite suboptimal treatment, in-drain composting beds can be expected to keep providing some neutralisation and metal removal beyond 12–18 months. The risk of long-term poor treatment (or treatment failure) can be reduced by using improved quality organic materials at construction or topping up with fresh materials when poor performance is noticed.

Sediments eroded from drain walls over the long term may diminish the treatment capacity of composting beds as the materials bury the organic mix. The silt layer may reduce movement of alkalinity from organic materials into overlying acidic waters, hence slow rates of treatment.

6.3.5 Case study: Wallatin Creek in-drain composting bed

i. Site: Uppermost 200 m section of the Wallatin Creek drain on Ashleigh and Jenny Bonser's property, Doodlakine, Western Australia (see location map in Appendix D).

ii. Construction: *March–April 2007* – A mix of wheat stubble and composted sheep manure was loaded into the uppermost 200 m section of the drain (Fig. 22). The mix consisted of 7 tonnes (48 m^3) of wheat stubble sourced as 0.5 tonne bales from Wellard Feeds, Wongan Hills, and 0.9 m^3 of composted manure from sheep feed-lots sourced from bulk suppliers in Mundaring. This gave an approximate mixture of 1.1% composted sheep manure (by volume of straw packed in the drain, estimated to be 84.7 m^3). The composted sheep manure contained 29% (by weight) carbon, 2.3% nitrogen and 0.7% phosphorus. The stubble contained less than 0.4% nitrogen with a C:N ratio of 90:1.

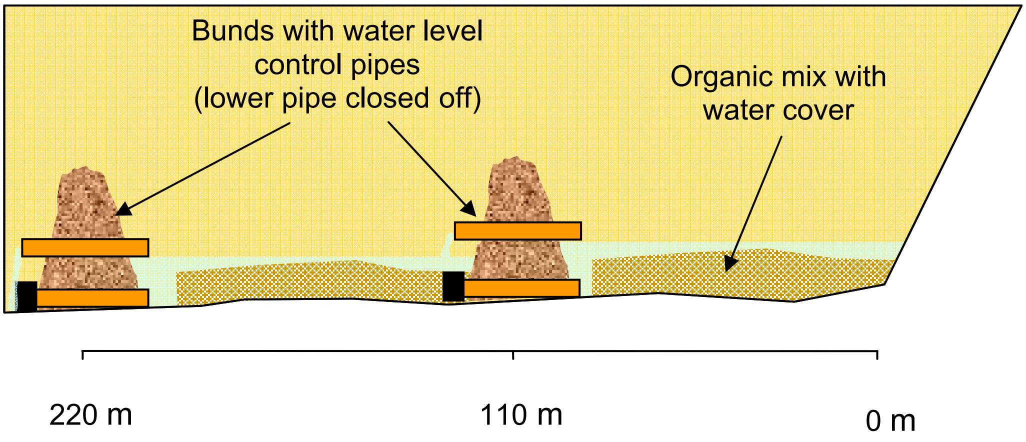


Figure 22 Cross-sectional view of bunding and water-level control pipe layout in the 200 m in-drain composting bed at Wallatin Creek. Note water level adjusted to achieve water cover over organic matter laid on uneven drain base.

Two earth bunds with 90 mm PVC pipes inserted were placed across the base of the drain at 110 m (from the end) and 220 m. These were constructed to enable water levels to be raised and controlled in the treatment system (by end-capping the PVC pipes).

The wheat stubble was layered by hand in the base with the final depth of organic matter 0.3–0.4 m. Composted sheep manure was sandwiched at approximately 0.1–0.13 m above the base. Although some sections of the drain had pools up to 0.2 m deep, it was necessary to use the bunding to raise water levels up to 25 mm above the surface of the organic materials to keep them flooded. Raising water levels also enabled outflows to be measured using an automated flow meter. Water levels were raised by end-capping the outflow pipes through the bund. Although some of the organic mix tended to float after loading (Fig. 23), this settled within a few weeks.

From a practical perspective, loading and placing straw bales in the base of drains (for larger treatment systems) may be more easily achieved by using rolled straw (i.e. enabling the straw to be rolled out in the base) or a loading system involving a side-casting conveyor belt (reaching over the drain bunds) followed by compaction of the material in the drain.

End October 2007 – The organic mix in the in-drain composting bed was recharged after continuing poor performance in the months after establishment. A further 3.5 m^3 of composted sheep manure was incorporated into the organic material in two sections of 6 m and 10 m length. This gave a final mix (including the initial composted sheep manure loading) of 5.0% non-cellulosic material by volume. The additional composted sheep manure was mixed into the existing organic materials to provide a concentrated treatment cell at the mid point and at 10 m from the end of the composting bed. Material was added in a concentrated cell in preference to mixing it along the entire 200 m of the treatment system. The effective composted sheep manure concentration within the recharged sections was 40% in the 10 m long section and 67% in the 6 m long section.

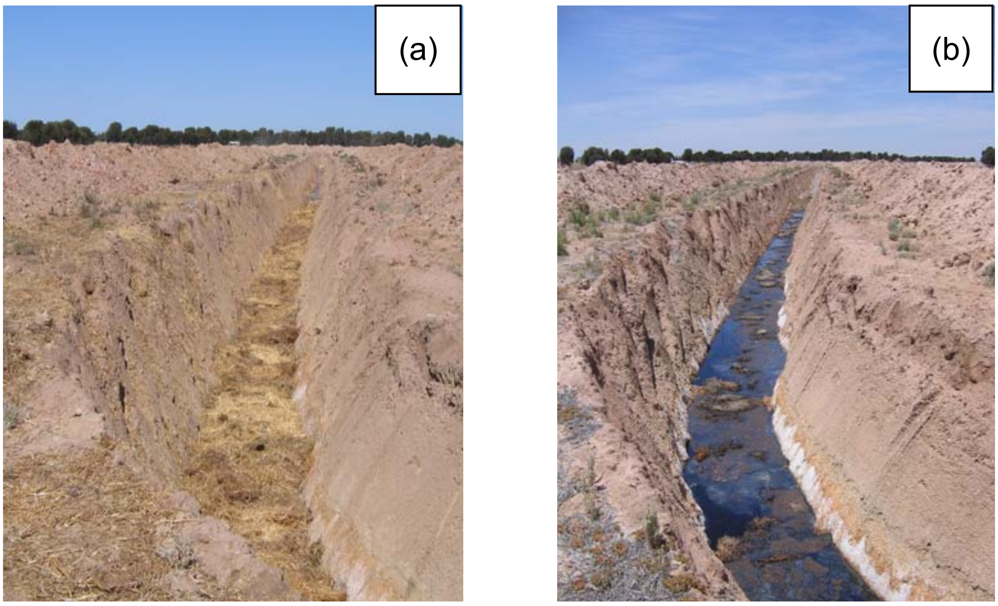


Figure 23 In-drain composting bed (a) after construction and (b) 6 months operation (during Trial period 2)

iii. Monitoring outline: The trial results are reported in two sections – Trial period 1 (April–end October 2007) and Trial period 2 (end October 2007 – end July 2008). Point measurements of water quality and flow were made in the 3 months prior to construction of the treatment system and fortnightly to monthly after construction. Outflows from the treatment section were continuously recorded using an electromagnetic flow meter (EmFlux 2020 electromagnetic flow meter with I300 transmitter). Water quality measurements included regular monitoring of pH, EC, redox potential, alkalinity and acidity with samples taken to measure concentrations of major ions, metals and other elements at the laboratory (see Appendix A). Net acidity was calculated from metal concentrations, pH and alkalinity (see Section 3.2.1 and Appendix B). Twelve 4 m deep groundwater monitoring bores were also installed in two rows of six bores along the levee banks (i.e. six on the east, six on the west) to provide information on changes in water levels adjacent to the drain due to raised water levels in the drain and the quality of groundwater seeping into the drain section. These were sampled for water quality twice during the evaluation period (July 2007 & April 2008).

Estimated rates of acidity treatment were calculated by difference between outflow net acidity loads and estimates of acidity load in groundwater inflows to the 200 m section of drain. Net acidity loads were estimated between water quality sampling times from the product of flow volume and net acidity. Rainfall contributions to flows between water quality sampling times were removed from outflow estimates since most water quality measurements reflected baseflow conditions and not rainfall events. To estimate groundwater flows, inputs from rainfall were removed and losses by evaporation added to total outflows recorded between water quality sampling times. Interpolated rainfall and evaporation data for the site during the trial was provided by the SILO Data Drill (Queensland Department of Natural Resources and Mines 2008). In using this data, it was assumed that rainfall resulted in direct input to the drainage channel and that 10% runoff occurred from the banks and berms for daily rainfall >5 mm. Losses due to evaporation were assumed to occur from the full area of the in-drain composting bed (nominally 200 m long x 1.2 m wide).

Net acidity load to the composting bed via groundwater inflow (between water quality sampling times) was estimated using the product of groundwater flow volumes (discussed above) and average groundwater net acidity. Groundwater net acidity information was obtained from monitoring bores and decreased from an average of 0.43 g $CaCO_3$/L in the first six months to an average of 0.40 g $CaCO_3$/L over the remainder.

iv. Performance : Over the 494 days (16½ months) of trial evaluation the 200 m of in-drain composting bed treated 83% (795 kg) of the 955 kg acidity (as $CaCO_3$) estimated to be discharged to the drain. During the first 7 months (Trial period 1), half of the acidity loading was treated but outflow waters rarely exceeded pH 5 and frequently contained untreated acidity (Fig. 24), often present as dissolved iron. Some of this was partly attributed to microbial reductive dissolution of iron in soil that had washed from the walls of the drain.

During the second trial period, after loading additional organic matter into the drain, 100% treatment was achieved with outflow waters exceeding pH 5.5 (reaching 6.6 on some occasions) and containing no net acidity during the first five months (Fig. 24). Although pH declined slowly over this period, all the acidity discharging to the drain during this time was treated and outflow water contained excess neutralising capacity (equivalent to 153 kg of $CaCO_3$). This treated acidity in drain water for more than 550 m downstream of the trial site for at least 3½ months (as shown by water quality monitoring at a point 550 m downstream of the composting drain). During this period of optimum operation black sulfides were visible just beneath the water surface (Fig. 23b) along with the odour of rotten eggs, indicative of hydrogen sulfide gas.

Six months into Trial period 2, the in-drain composting bed began to show signs of not treating the entire acidity loading from groundwater seepage. Although the pH of outflow water was 4.7–5.5, this frequently contained untreated acidity, often present as dissolved iron. Even when this occurred, the composting bed maintained treatment of 71% of the acidity load. Over the 9 months of Trial period 2 reported here, 406 kg of a total loading of 454 kg of acidity in drain water was treated (or 89% of the acidity loading).

The 200 m of drain discharged 1843 kL of water over the 16½ month period, of which 1701 kL was estimated to be from groundwater discharge. Flows in this period were 1.36–6.74 kL/day (approximately 0.08–0.39 L/s/km).

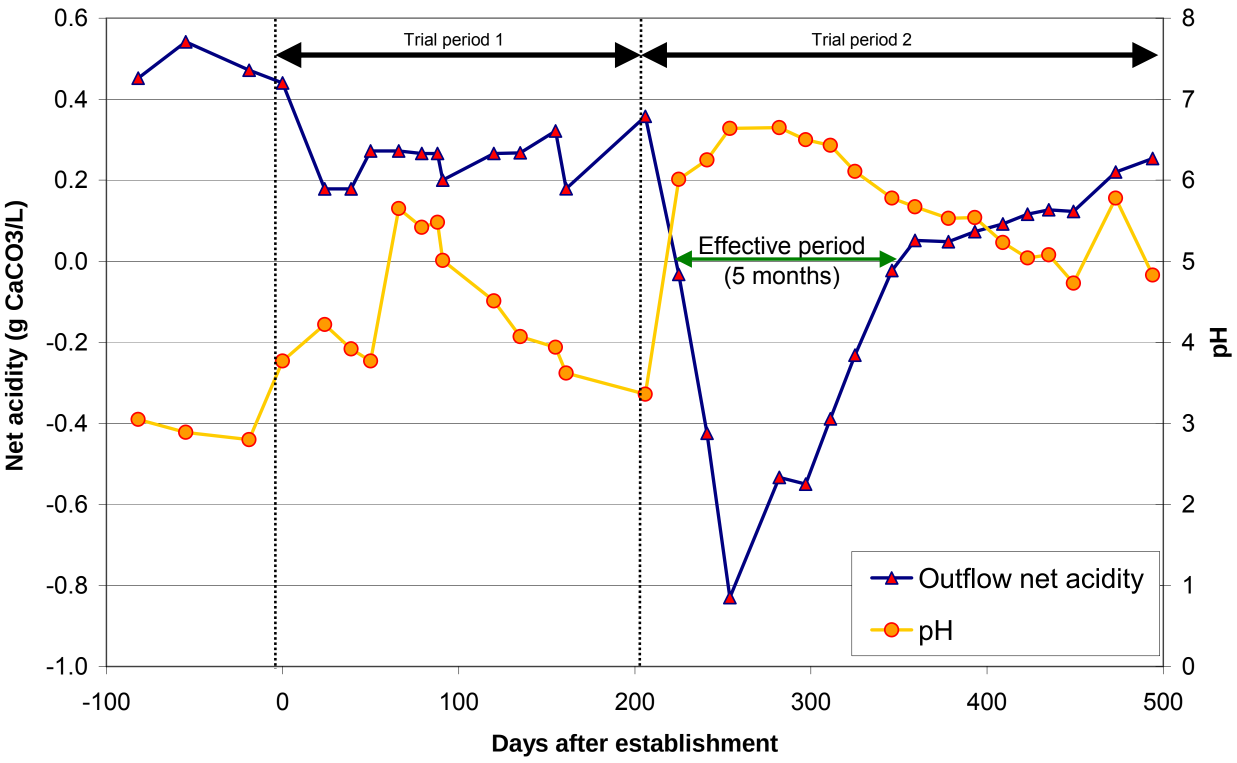


Figure 24 Net acidity (on the right axis) and pH (on the left axis) in outflow waters after installation of an in-drain composting bed (note: negative acidity indicates alkaline water)

Rates of net acidity loading from the groundwater inflow were estimated to vary from 2.4 to 16 g $CaCO_3$ /m^2 drain base/day (Fig. 25) – much greater variation than estimated when designing the treatment system. Net acidity loading generally declined during the trial but recovered sharply at the end of Trial period 2, probably as a result of late autumn–early winter rainfall causing increased groundwater discharge.

For most of Trial period 1, treatment rates were sufficient to neutralise about half of the net acidity load (from groundwater inflows) but during Trial period 2 were sufficient to neutralise up to double the net acidity load (Fig. 25). Maximum treatment rates were estimated to be up to 10 g $CaCO_3$ /m^2 drain base/day during Trial period 1 and 21 g $CaCO_3$ /m^2 drain base/day during Trial period 2 (Fig. 25). These persisted for up to 3 months (in Trial period 2) and declined thereafter to rates as low as 2 g $CaCO_3$ /m^2 drain base/day, although did not remain at these levels (Fig. 25). The composting bed removed a variety of minor elements from the drain water when functioning well, although some minor elements were not removed even when acidity treatment was 100%. When the pH of the outflow water exceeded 5, the concentrations of minor metals such as cobalt, nickel, zinc and lead and of minor elements such as uranium, arsenic, lanthanum and cerium decreased by more than 95% (often to below detection concentrations) compared with concentrations before treatment. This continued for minor elements such as lead and uranium when the pH of outflow water fell to less than 5.

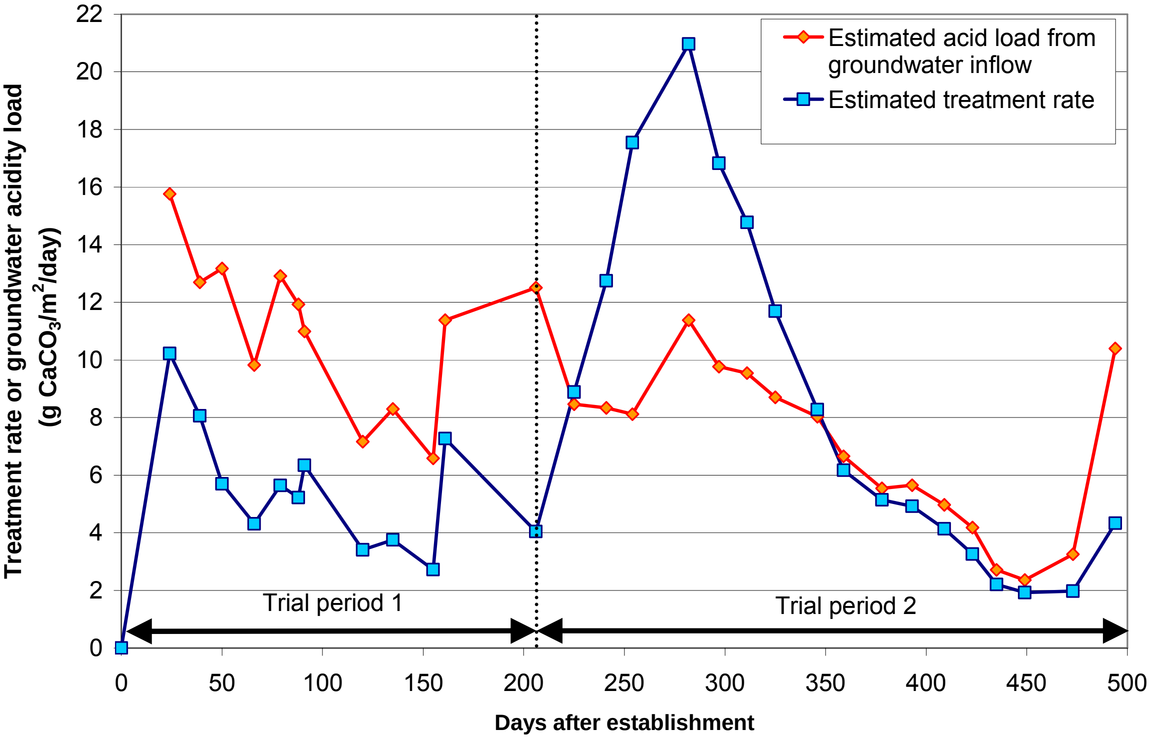


Figure 25 Estimated daily net acidity load attributed to groundwater inflow per m^2 of drain base and net acidity treatment rates in the in-drain composting bed during the two trial periods. Net acidity treatment rates calculated from net acidity in outflows and estimated net acidity in groundwater inflows.

One side effect of the treatment process was that high concentrations of nutrients leaked into drain water. Concentrations of outflow dissolved nitrogen and organic carbon were quite high after initial loading of organic materials into drains (e.g. up to 22 mg total dissolved nitrogen/L and 112 mg dissolved organic carbon/L early in the second trial period). Much of the nitrogen was present as ammonia with no nitrate detected throughout the evaluations. Discharges of dissolved phosphate were also high (up to 3.1 mg/L in Trial period 2) though they decreased after several months to concentrations similar to those of untreated drain water. Most of the nitrogen and phosphorus was probably derived from the composted sheep manure (which was initially 0.7% phosphorus by weight), though some may also have been derived from phosphate adsorbed onto iron oxides dissolved under the waterlogged conditions. High phosphorus concentrations in outflow water can be avoided by using organic materials with low phosphate concentrations.

Where in-drain composting beds are constructed in sections, the dissolved nitrogen and carbon in treated water may be consumed by microbes in the sediments of drain sections without composting beds. Phosphate was probably retained in these sections by adsorption onto sediments since concentrations of phosphate were generally less than 0.2 mg/L at a distance of 550 m downstream of the composting bed despite up to 3.1 mg/L of phosphate in the outflow.

v. **Design implications:** Creating an inexpensive mix of organic materials that can support microbial activity sufficient to cope with the acidity flowing into a drain is the key to successful design of in-drain composting beds.

Partial and full treatment of comparatively high acidity loading was achieved over six months using a mix of as little as 5% composted sheep manure with straw costing approximately $16 per m^3 ($5.76 per metre drain for a 0.3 m bed). The mixture effectively defines the lower limit of non-cellulosic material required within organic mixes suitable for treating acidic saline drain water. Such a minimal mixture is only suitable for treating long-term acidity loading rates of less than 3.5 g $CaCO_3/m^2$/day, similar to the acidity load from a drain discharging approximately 0.1 L/s/km with an average acidity of 0.45 g $CaCO_3$/L.

Concentrating the more decomposable material in short sections in the drain (creating localised mixtures of 40–67% sheep manure in straw) enabled the material to work more effectively than if the material had been evenly mixed with the slower decomposing straw materials (i.e. creating a homogenous mixture of 5% sheep manure in straw).

The treatment effectiveness was probably gradually eroded by acidic water percolating directly through the base of the composting sections. This could perhaps be minimised by using a subsoil carbonate, lime-sand or limestone lining. In many other situations with composting wetland treatment, acidic water generally flows over the surface (see Section 7.2 Composting wetlands). This effectively limits the extent to which the acidic water can interact and affect sulfate-reducing bacterial activity (given that these organisms and the black oozes formed by them are pH sensitive).

The challenge in using in-drain composting beds is maintaining high rates of treatment for periods longer than six months and increasing treatment capacity to cope with drains with high acidity loading. Tripling the sheep manure content (to 15%) would be expected to yield a much higher treatment rate over the initial 6 months and up to 2 years of effective treatment for similar drains with acidity loading of 4–12 g $CaCO_3/m^2$/day (see Table 3 for comparative flows). The use of alternative slower decomposing (cellulosic) material other than straw (e.g. lupin residues or harvester chaff) is also likely to achieve significantly higher and more sustained treatment rates. Recent results (at the time of publication) indicate that recharging sections of the composting drain with 1 tonne of seconds grain (approximately 1% by volume) boosted treatment capacity by more than 500% (reaching 25 g $CaCO_3/m^2$/day) and that this has been maintained for more than 7 months.

The costs of treatment per kL of discharge depended on whether full treatment or partial treatment was achieved and the period of evaluation. A key factor is that, if organic materials can be sourced for minimal cost and treatment can be maintained over several years, costs are likely to be minimal. During the first trial costs were approximately 40 cents per kL but only partial acidity treatment was achieved (excluding sludge management costs). During the 6 months of optimal treatment (Trial period 2) costs were less than $1 per kL, but could be as little as 30 cents per kL if averaged over 18 months and less if cheaper organic materials are used. All of these cost estimates exclude managing the materials when drains are cleaned.

Take home message: In-drain composting beds offer low-cost, easy to construct units for treating and retaining acidity and minor metals within drains.

6.4 Other potential in-drain options – diversion wells

Diversion wells using local crushed carbonate materials or lime-sand may be a viable alternative for partial neutralisation of drain water in some situations. Investigations in Europe and North America indicate that diversion wells can achieve partial treatment for acidic water (Skousen et al. 1998), though these have only been applied to waters with moderate levels of acidity (<0.3 g $CaCO_3$/L). Diversion wells are carbonate-based treatment systems designed to use the head pressure of flows along drains to force water through a well loaded with crushed limestone, though coastal lime-sand could also be used. The flow is intended to maintain the limestone in an abrasive environment where particles vibrate against each other and flush precipitates from the chamber (Skousen et al. 1998). The latter process is intended to maintain performance by reducing sludge or silt build-up.

Achieving sufficient head pressure may limit the application of this option in Wheatbelt drains. Conceptually, flows need to be collected upstream from a pool and piped to the well with sufficient head pressure to force water through the limestone or lime-sand (Fig. 26). This may require more than 500 m of piping (given drain gradients are less than 0.5%) and ponding of waters in some sections. Regular refilling of the well (up to once a week) will also be required as limestone or lime-sand is consumed. There has been no trialling of this treatment approach in Western Australian conditions.

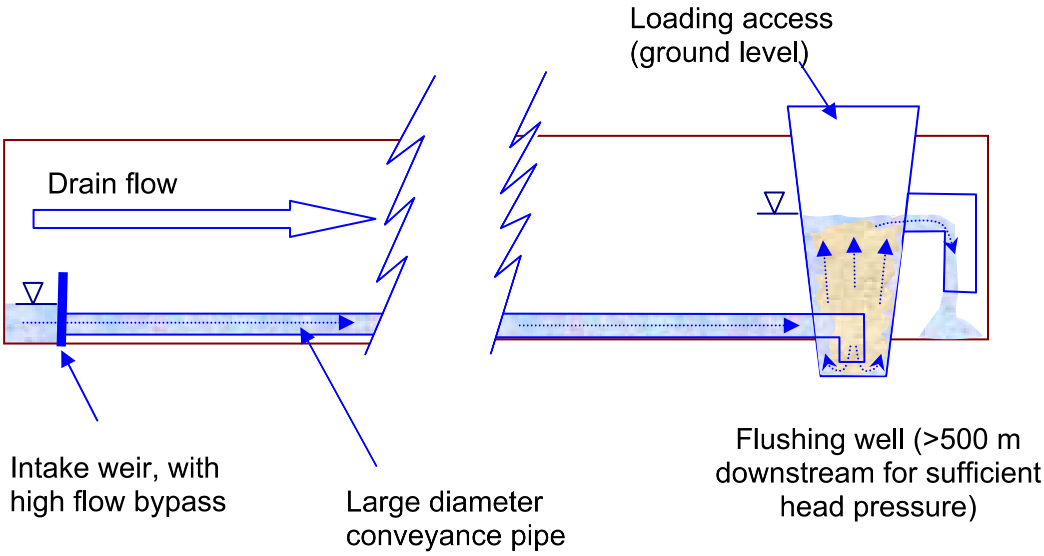


Figure 26 Schematic cross-section of a diversion well illustrating intake and downstream well outlet within a drain

7 End-of-drain treatment options

The following section outlines treatment options suitable for treating acidic water at the end of drainage channels using purpose-built structures.

Important elements of design and construction are discussed along with considerations for managing the treatment units. Guideline information on performance of the treatment options, extrapolated from local case studies, is provided to assist with understanding what might be achieved using each type of treatment approach. Note that the extent to which different treatment options will work may vary, and not all of the options will achieve complete treatment. Case studies of pilot treatment units are provided to give users some insight into local design and performance.

7.1 Lime-sand basins

7.1.1 Principle

Lime-sand basins involve passive neutralisation of acidic water by ponding in a constructed disposal basin lined with finely crushed limestone or lime-sand. An alternative design is the use of limestone leach beds where water percolates through crushed limestone (Black et al. 1999). Lime-sand basins generally operate at constant rates of neutralisation and in some situations may result in significant evaporation of water before treatment is completed.

7.1.2 Design and construction considerations

i. Water pre-treatment: Neutralising acidic drain water is most effective after as much dissolved iron as possible is removed by passage through an oxidation pond. Dissolved iron (as Fe^{2+}) can be removed by oxidation in air to form fine iron precipitates and settling from the water. This is best achieved with net alkaline, iron-rich waters (PIRAMID Consortium 2003; Younger et al. 2002), which are uncommon in the eastern and central Wheatbelt (Shand & Degens 2008).

The primary function of an oxidation pond can be achieved simply by dividing evaporation basins into cells with drain inflow water held in the first cell before overflow via a riffle (to aerate the water) into other cells containing crushed limestone or lime-sand. Experience with evaporation basins in the WA Wheatbelt indicates that ponds must be large enough to hold water for at least several weeks. This holding capacity is estimated by dividing pond volume in litres by inflow rate as L/day. Where water pumped from a drain is poorly aerated, an in-line venturi or fountain/spray outlet can be used to increase aeration and to increase rates of iron oxidation.

As discussed previously (Section 3.1), a major limitation to the success of this pre-treatment is that the oxidation rate of dissolved iron (mostly Fe^{2+}) is very slow at the pH of most acidic drain water in the Wheatbelt. So this reaction may not be complete before overflow into the treatment basins. In summer, this is further exacerbated by the pH of drain water often falling to less than 3.

Further guidelines for sizing iron oxidation ponds (with aeration riffles) is provided in Section 4.1.1.2 – Settlement Lagoons in PIRAMID Consortium (2003).

ii. Depth: The depth of lime-sand basins should be based on the design depth for evaporation, given that this option is a modification of an existing evaporation basin (see guidelines for designing evaporation basins; JDA Consultants & Hauk 2004). Nevertheless, designs with shallow water are desirable as they promote the mixing of acidic water and crushed limestone/lime-sand on the basin floor by wind and wave action, thereby increasing neutralisation rates.

iii. Sizing: There are no international guidelines for sizing lime-sand basins but, based on the case study at Beacon (discussed in Section 7.1.5), acidity can be treated at a rate of at least 0.27 g $CaCO_3$/L per 30 days for an evaporation basin filled to no more than 1-m depth. This figure can be used to calculate the size of evaporation basins needed for treatment (see example calculation in text box below).

Retention time (days) = 30 x net acidity/0.27

An important consideration in using lime-sand basins is concentration by evaporation: salt concentrations may more than double during treatment. This will need to be considered when there are specific uses for the water after treatment or before discharging water to floodways or lakes. This evapoconcentration effect will diminish with less acidic water treated in conditions with low evaporation rates (e.g. water with acidity of 0.25 g $CaCO_3$/L treated in winter).

iv. Side batters: Gently sloping batters (at least 1:5 gradient) are important to avoid wave action eroding silts and clays from banks. Steep banks can increase wind–wave erosion of these fine soil materials from basin walls and their deposition over the lime-sand/crushed limestone bed, effectively creating a barrier that slows neutralisation.

v. Leakage: Leakage from basins is generally considered to enhance water treatment, although it represents a loss of water that cannot be easily recovered (if required for re-use). Rates and pathways of leakage will depend on site soil and hydrogeological conditions, particularly depth-to-groundwater before filling the basin. Siting the basin and placing lime-sand/crushed limestone within the basin can be varied to match these leakage pathways. For example, where groundwater is shallow and leakage likely to be greatest through the side walls, lime-sand/crushed limestone should be placed on the walls to ensure neutralisation.

vi. Inflow point: Maximising the mixing of water flowing into the basin with lime-sand on the bottom can increase initial neutralisation rates, particularly if water is pumped in.

Up to half of the neutralisation of acidic water by lime-sand can occur within the first few minutes of contact if the water is well mixed with neutralising materials. If water is pumped into the basin, mixing can be increased by using a small lime-sand/crushed limestone-filled pit or cell (e.g. using concrete sumps filled with lime-sand) or jetted across the lime-sand/crushed limestone on the floor of the basin. With gravity-fed inflows, initial lime-sand contact and mixing might also be achieved by directing water across the floor of the lime-sand/crushed limestone lined basin using slotted/perforated piping to distribute flow.

Example calculation — sizing a lime-sand basin

Drain flow – average 1 L/s

Net acidity – average 0.5 g $CaCO_3$/L

Assuming that lime-sand basins treat acidity at a maximum rate of 0.27 g $CaCO_3$/L per 30 days, the acidity of one litre of water (at 0.5 g $CaCO_3$/L) needs to be held in the lime-sand basin for 56 days (i.e. 30 x 0.5 ÷ 0.27 = 1.85 x 30 days = 55.6 days)

Over 56 days, the drain generates a flow of 4.8 million litres (i.e. 1 L/s x 86 400 seconds/day x 56 days)

Therefore the lime-sand basin must be large enough to hold at least **4.8 million** litres every 56 days. Allowing a safety margin for treatment of (say) 10%, the basin must hold at least 5.3 million litres (5300 cubic metres).

The area of the water in the basin will need to be at least **0.53 hectares** (i.e. 53 800 m^3 ÷ 1 m depth), based on previous trials using no more than 1 m depth of water. *Note that final volume sizing also needs to allow for sloping walls and allow for 0.3 m of freeboard and can be calculated using information published by the Department of Agriculture (2004).*

A final consideration is the change in salt concentrations. In mid-summer, approximately 0.62 m of water depth will be lost in evaporation over 56 days (assuming average maximum evaporation losses of 11 mm per day; Bureau of Meteorology 2008). Given an initial depth of 1 m and assuming no loss of salt or water by seepage, this will result in salt concentration more than doubling before the treatment is completed.

vii. Lime quality: Treatment performance will be greatest in aerated acidic water using high grade, small-grained carbonate materials (assaying as >90% $CaCO_3$). Grain size has a greater effect on efficiency than purity (Sherlock 1997), therefore the poorer performance of lower grade materials may be offset by crushing. Materials with smaller grains (<1 mm) will sustain higher reaction rates than materials with larger particles (generally most coarse crushed limestone) but, unless continuously stirred or resuspended, materials with smaller particles may form crusts and begin to seal over, reducing the reactive contact area between water and neutralising materials. With larger particles (e.g. >1 cm diameter), the contact area may be maintained.

Crushed dolomite, which is a mix of magnesium carbonate ($MgCO_3$) and $CaCO_3$, is only recommended for use if high assay $CaCO_3$ is not available. $MgCO_3$ dissolves much more slowly than $CaCO_3$ (Cravotta et al. 2008) and trials using this in active treatment situations have found $MgCO_3$ materials far less effective than $CaCO_3$ (Cravotta et al. 2008; Green et al. 2007).

Further information can be obtained from the significant body of literature in Europe and the USA regarding the selection and use of limestone for treatment of acidic waterways (e.g. Sverdrup 1983; Olem 1991; Menendez et al. 2000; Donnelly et al. 2003).

7.1.3 Operational management

Although lime-sand basins are intended to be passive treatment systems and not require regular maintenance, it is likely that these may require:

i. Crust and mud removal: Periodically scrape off the surface of the lime-sand bed to remove accumulated muds and crusts when treatment effectiveness decreases.

Water with high iron concentrations (>10 mg/L) will require more frequent crust removal to maintain effective treatment as such water will speed the formation of rate-reducing crusts on the surface of the lime-sand bed. Some rejuvenation of effectiveness may be achieved by mixing the mud and crusts with underlying unreacted lime-sand, though this will limit the future effectiveness of the neutralising material. Maintaining water cover on the sediments and avoiding high salinity (above approximately 120 000 mg/L total dissolved salts or an electrical conductivity of 14 500 mS/m) will reduce the formation of salt and gypsum crusts.

ii. Sludge management: Periodically remove sludges retained within the basin, particularly to reduce the risk of metals accumulated during previous treatment periods re-dissolving in water subsequently passing through the basin. The sludges should be disposed of to avoid erosion into waterways that may surround the evaporation basin. There are likely to be significant quantities of unreacted lime-sand in any material removed from the floor of lime-sand basins and this will reduce the risks of any metals trapped in the sludges being remobilised into treated waters.

iii. Redosing: Reapply lime-sand every few years. This will depend on the ongoing acidity loading to the basin and the concentrations of iron in the acidic waters. For example, acidic water 1 m deep with an average acidity of 0.5 g $CaCO_3$/L will consume a maximum of 0.5 kg of $CaCO_3$ for every square metre of basin floor (or about 0.56 kg lime-sand, assuming 90% $CaCO_3$ content). If this water is replenished six times a year, 3.3 kg of lime-sand/m^2 will be consumed by neutralisation reactions representing an approximately 3 mm thick layer of lime-sand on the basin floor (assuming a lime-sand density of 1.1 kg/L). Note that if there is armouring of lime-sand (due to high iron concentrations in inflow water or siltation), more frequent redosing of lime-sand will be needed, irrespective of treatment rates as not all of the lime-sand will be available for treatment.

7.1.4 Effectiveness

Short term (<6 months): Lime-sand basins have a high likelihood of successful treatment over the short term. In some cases, 100% treatment of acidity may be achieved though it can require several months to achieve this. There can be mixed effectiveness in removing minor metals (see case study).

Long term (>6 months): Long-term performance of lime-sand basins may be similar to that over the short term if maintenance is carried out to prevent mud accumulating and a crust forming. Effectiveness will also depend on the concentration of iron in treatment water, where higher concentrations will reduce the effectiveness of the lime-sand due to armouring. This does not result in a complete loss of neutralising capacity, although it may result in slowed neutralisation rates, as has been reported for lime-sand in reactor columns (Santomartino & Webb 2007). The longevity of effectiveness is likely to be greater with acidic water containing high concentrations of aluminium which does not form hard coatings on lime-sand grains but a fine gelatinous white sludge.

The efficiency of $CaCO_3$ use is likely to be low in lime-sand basins mostly because burial by mud washed off the banks of the basin, formation of aluminium and iron precipitates and growth of salt and gypsum crusts will reduce the effective working life of the materials before all the neutralising capacity is consumed. Despite this, it is reasonable to expect that an efficiency of up to 15% might be achieved over several years of operation (see case study below); that is, a maximum of 15% of the lime-sand is used in achieving neutralisation before it needs to be replaced.

7.1.5 Case study: Beacon lime-sand basin

i. **Site:** A pilot-scale evaporation basin on the margin of an evaporation basin on Gerry and Ty Kirby's property, off Bimbijy Rd, Beacon, Western Australia (see location map in Appendix D). This evaporation basin was constructed as part of the Engineering Evaluation Initiative drain and water management investigations at Beacon.

ii. **Construction:** The trial was undertaken in an evaporation basin (20 m x 25 m base area) constructed within the perimeter of a larger evaporation basin of approximately 15 hectares. Three walls on the inside of this basin had a 1:1 gradient with the wall on the external side having a 1:5 gradient. All walls were constructed to 2 m height above the basin floor.

Ninety tonnes of lime-sand were spread on the evaporation basin floor giving an average depth of 13 cm (Fig. 27). It was initially intended that 20 tonnes be spread to give an average depth of 3 cm but it was necessary to add more lime-sand to achieve a level lime-sand pad because the basin floor had been rutted by the 20-tonne loader spreading the sand. Rutting occurred because of boggy conditions created by groundwater seepage from the surrounding evaporation basin. The lime-sand used for the trial was sourced in bulk from the AgLime Australia coastal lime-sand pit at Lancelin (90% calcium carbonate by weight with 86% of particles 0.125–0.250 mm diameter and 10% less than 0.125 mm diameter).

iii. **Monitoring**: The lime-sand basin treatment trial was carried out through a series of five batch treatments. For each, at least 400 kL of water was pumped over several days from the surrounding larger evaporation basin to fill the pilot basin (Table 5; Fig. 28) with water jetted across the floor to maximise initial mixing with lime-sand during filling. Water was monitored at regular intervals over subsequent weeks and months after filling until either treatment was considered to be complete or all the water had evaporated. Any remaining water was pumped out of the basin before refilling with the next batch.

Figure 27 Levelling the lime-sand bed in the floor of the pilot evaporation basin

Regular measurements of water quality were made on inflow water during filling and in the basin after filling. These included on-site measurement of pH, EC, redox potential, alkalinity and acidity with samples taken for laboratory analysis of all major ions, metals and other elements (Appendix C). Net acidity was calculated from metal concentrations, pH and alkalinity (see Section 3.2.1 & Appendix B). The volume of water pumped into the basin was measured using a helical vane flow meter (50 mm, Elster H4000 Woltman Helix helical vane meter, Elster Metering Pty Ltd.). Volumes in the basin were estimated from a depth gauge and a depth-by-volume relationship established using the pumped volumes. Leakage was observed through the earth walls of the pilot evaporation basin, which occurred at variable rates between trials due to varying water levels in the surrouding larger evaporation basin.

iv. Performance results: Five batch trials were performed over an 11 month period (Table 5) with complete treatment regarded as when acidity of water in the basin was reduced to less than 0.05 g $CaCO_3$/L. Treatment was completely effective for all but Batch 3, although the time to reach this state varied (Table 5), mainly due to the various concentrations of acidity in the water pumped into the basin (Fig. 29).

Partial neutralisation occurred for the third batch (Table 5), at the end of which the water had evaporated prior to being treated. Failure of treatment in this batch was attributed to a gradual build up of mud (Fig. 30) and a salt–gypsum crust on the surface of the lime-sand at the end of the second batch (Fig. 30). The crust was scraped and removed prior to the start of the fourth batch.

Although treatment times varied between batches, the rates of acidity treatment were generally similar, despite different initial acidities and different weather conditions between batches (Table 5, Fig. 29). Treatment rates generally varied from 0.27 to 0.37 g $CaCO_3$/L per 30 days for all except Batch 3, where treatment was very slow. These treatment rates indicate that treatment is more dependent on the water volume present than the contact area with lime-sand (which increases as depth decreases). The loss of water by seepage and evaporation from the basin varied from batch to batch resulting in variable water depths, but did not appear to influence the neutralisation rate, indicated by the linear declines in the concentration of acidity with time (Fig. 29). For example, 30 days after filling, in Batch 1, the water was 0.39 m deep whereas for Batches 4 and 5 the depths were 0.2 m and 0.15 m respectively (with all batches starting at depths of at least 0.9 m; Table 5).

Figure 28 Pilot lime-sand basin nearing the point of maximum filling (Batch 1)

The efficiency of $CaCO_3$ use for all five batches was estimated to be at least 3.8% and would continue to increase marginally with further trials given that the neutralising capacity was not exhausted. This was based on assuming that 50% of the estimated 1500 kg of acidity loading in the five batches of acidic water was directly neutralised by the lime-sand bed and was compared with the 20 tonnes of lime-sand on the basin floor that would have been within the surface 3 cm. This layer was assumed to be interacting most with the basin water, whereas deeper sand would be less likely to play a part in neutralisation. It was assumed that half of the water seeped through the side walls and was neutralised by materials other than the lime-sand on the basis of the rate of decline in water depth being more than double that estimated to be due to evaporation.

The efficiency of $CaCO_3$ use for the full 90 tonnes of lime-sand spread in the basin floor would be at least 0.6%, but as described earlier much of this would not have played a role in neutralising the water. Although 90 tonnes of lime-sand were spread, this was not by design but because more sand was required to ensure an even spread as a result of the rutted basin floor.

Table 5 Summary results from batch treatment trials with the pilot lime-sand basin (Note: The double line break between 3 and 4 indicates that crust and mud were removed from sand surface between batches 3 and 4)

Batch	1 2		3 4		5
Volume pumped into basin (kL)	590 (1.15 m)#	502 (1.03 m)#	702 (1.29 m)#	460 (1.00 m)#	415 (0.9 m)#
Inflow pH range	3.5–3.7 3.2–3.3		3.0–3.1 3.1–3.2		3.6–3.8
Inflow net acidity range (g $CaCO_2$/L)	0.28–0.36 0.40–0.4	9	0.78–0.85 0.61–0.6	6	0.38–0.47
Weeks to reach pH 6.0	3.5* 6–7*		Did not reach pH 6.0 (evaporated)	6-6.5* Estimated	6
Rate of acidity treatment (g/L/30 days)	0.32 0.27		0.16 0.37		0.27
Seasonal conditions during treatment	Warm, intermittent cool NW to W winds	Hot, intermittent warm mostly N to E winds	Hot, intermittent hot mostly N to E winds	Warm, intermittent cool NW to W winds	Cool, intermittent cool NW to W winds
Sand surface condition after treatment	Thin red mud cover	Thick mud cover with gypsum crust	Thick mud layer with gypsum + salt crust	Thin mud cover	Thick mud cover

* Approximate value estimated between measurement dates

\# Depth of filling after pumping (m above basin floor)

The salinity of water does not appear to influence treatment rates. Treatment rates and the salinity of basin water during Batches 1, 2 and 5 were similar: salinity for these batches started at 40 000–60 000 mg TDS/L and increased to 57 000–107 000 mg TDS/L during treatment (until acidity was less than 0.05 $CaCO_3$/L). This observation is supported by similar results in laboratory situations, where rates of pure $CaCO_3$ dissolution are generally only slightly depressed as salt concentrations (exceeding 60 000 mg TDS/L) increase, although the effect can be increased if carbon dioxide gas builds up in the water (Gledhill & Morse 2006). This might occur on still days or in deep ponds where mixing by wind is poor. It was not clear why treatment rates in Batch 4 after scraping and removing the crusting were greater than during other batches (even greater than fresh lime-sand in Batch 1). The pH of the waters during treatment may have played a role since these tended to remain lower during Batch 4 (compared with Batch 1), although were similar to the pH in Batch 2 (Fig. 29).

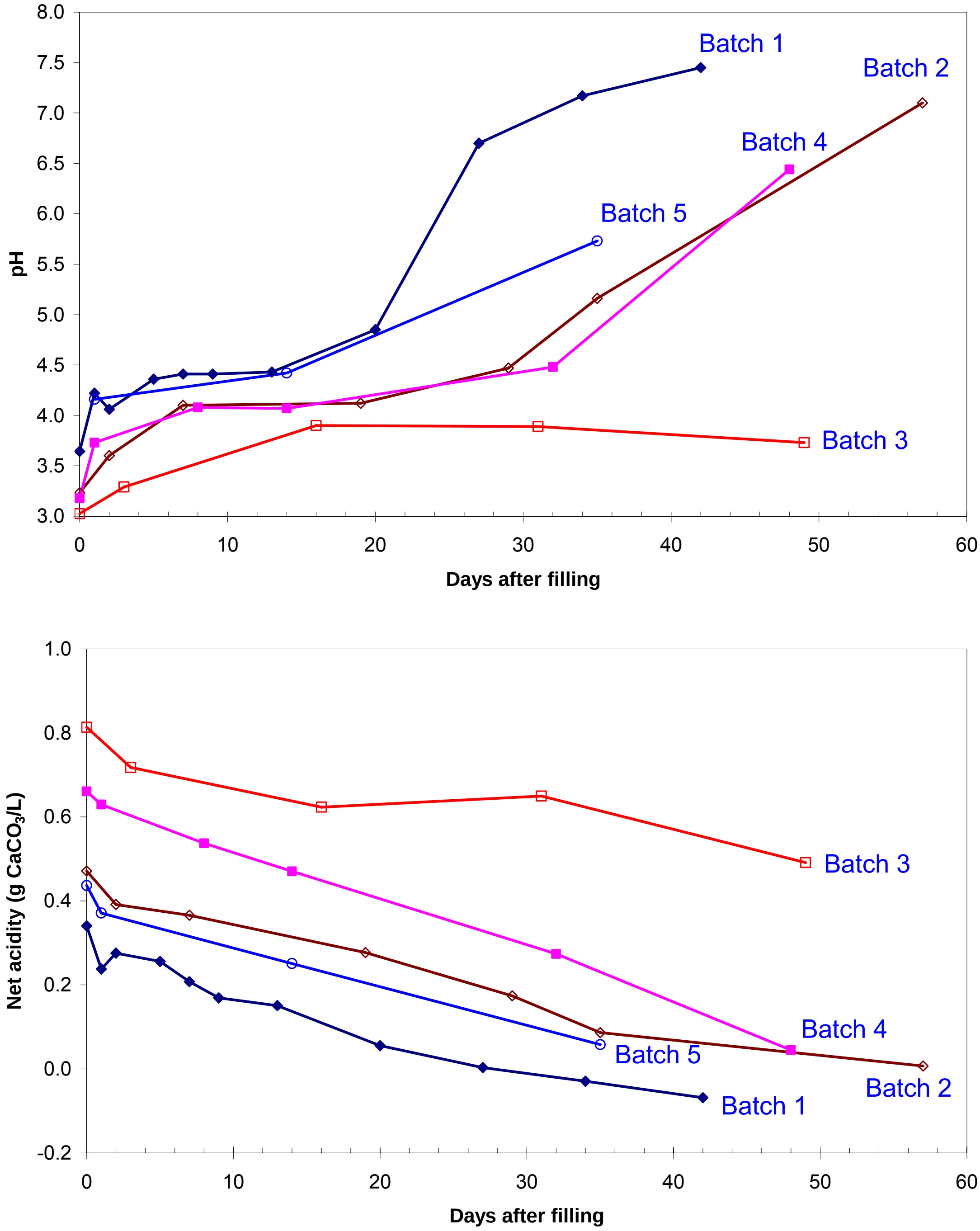


Figure 29 pH and net acidity (g $CaCO_3$/L) for successive batches of acidic saline water in a pilot evaporation basin lined with lime-sand (Note: negative net acidity indicates water containing alkalinity; in this case, bicarbonate; Day 0 = flow averaged pH and acidity of inflow water)

One factor contributing to the success of treatment using the lime-sand basin was the comparatively low concentrations of iron. Flow-weighted iron concentrations were 6 in Batch 1 to 15 mg/L in Batches 3 and 5. These are low compared with 10–36 mg/L in the Beacon drain and 10–100 mg/L in other drains (Degens et al. 2008b). The reduced iron concentrations in inflow water were fortuitously achieved because the water had been pumped from the Beacon drain into the larger evaporation basin and aerated by wind-mixing (aided by the shallowness of the basin) over some weeks before being pumped into the pilot lime-sand basin. The lower concentrations of iron in inflow water into the lime-sand basin would have limited the amount of armouring. Armouring of lime-sand was found to occur rapidly when lime-sand was used to treat acidic water with more than 15 mg iron/L in a drain (see case study in Section 6.1.5) and a tank reactor (see case study in Section 7.3.5).

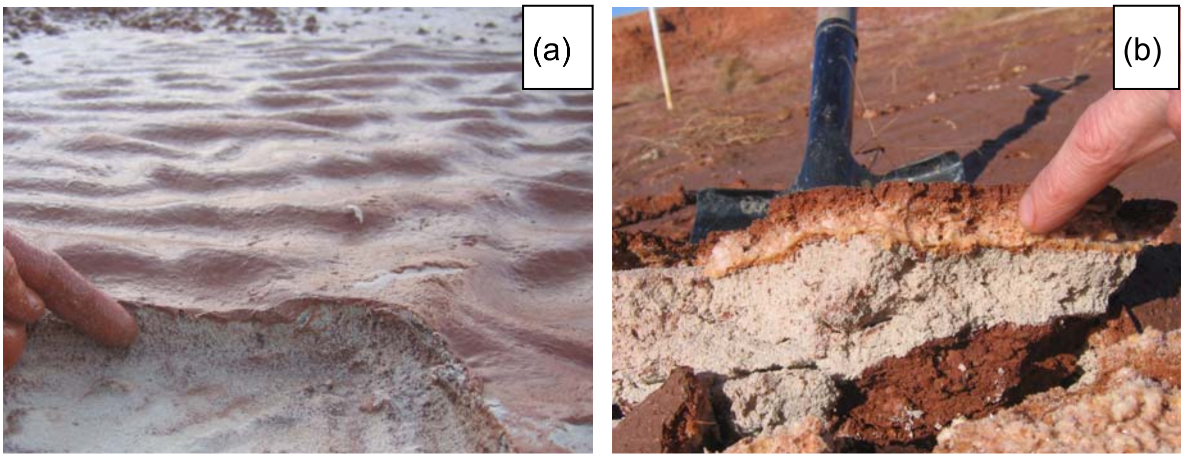


Figure 30 Thin mud layer (<5 mm) covering lime-sand after the first batch (a) and the mud and salt crust layer (up to 20 mm deep) covering the lime-sand after Batch 3

Although the water left in the basin was neutralised, a significant volume of water was also treated as it seeped through the basin wall. The basin was filled to at least 0.88 m depth for each trial but, within 30 days, typically less than 20% of the water remained. The seepage rate would have been controlled by the water level in the larger surrounding basin. This decreased from 0.41 m above the lime-sand basin floor during Batches 2 and 3 to 0.3 m below the basin floor at the end of Batch 5. No acidic water was found in groundwater <1 m beneath the basin floor (in shallow monitoring bores) or in the shallow groundwater beneath the outside walls. The raised groundwater salinity was immediately beneath the basin during the trials indicating vertical leakage.

Neutralisation of the acidic water enabled selective removal of some of the minor elements. Concentrations of lead, copper and arsenic were reduced by 40–99% when pH exceeded 5.5 in most trials, except for Batch 3 where concentrations increased (probably as the water evaporated). Initial concentrations of lead were 96–412 µg/L, but concentrations of arsenic and copper were much lower (2–12 µg/L and 6–23 µg/L respectively). Concentrations of some minor metals and elements were not reduced; for example, selenium (initially 0.3–5.4 µg/L), nickel (initially 68–171 µg/L) and cadmium (initially 0.3–1.1 µg/L). During most batches, the concentrations of other elements such as uranium, cerium and lanthanum consistently decreased during treatment, with the greatest reductions generally after pH exceeded 5.

Minor metals were removed from the water, most likely by adsorption onto clays in the muds on the floor and walls of the basin. These were likely to become strongly adsorbing as pH was raised during neutralisation. Removal of metals as hydroxides was unlikely since the pH was not raised sufficiently (Langmuir et al. 2004). Although removal of some minor metals from water by adsorption can be effective while pH remains high, the metals can be readily desorbed if the muds re-acidify. This may have occurred to some extent in the first few weeks after refilling the basin with acidic saline water.

v. Design implications: These results indicate that acidic drain water can be treated using lime-sand lined evaporation basins when iron concentrations have been reduced. Treatment rates indicate that most water can be neutralised at a rate of at least 0.27 g $CaCO_3$/L per 30 days for basins starting with water no deeper than 1 m and containing concentrations of dissolved iron not exceeding 15 mg/L.

7.2 Composting wetlands

7.2.1 Principle

Constructed composting wetlands use locally available sources of organic matter to encourage generation of alkalinity (neutralising acidic waters) and to trap minor elements as minerals by microbial reduction of sulfate to sulfide (see Section 6.3.1).

Acidity and minor elements are stored mainly as sulfide minerals which will remain immobile in the wetland *as long as the materials remain waterlogged.* The capacity of composting treatment to remove a broad spectrum of minor elements can be useful in situations where the minor element composition varies or is not well known.

Established designs for this type of treatment generally have water flowing over the top of the organic materials, where treatment occurs by downward diffusion of acidity (as H^+), dissolved metals and sulfate into the waterlogged organic materials, with alkalinity and hydrogen sulfide (rotten-egg gas) diffusing upwards to overlying water. Outflow from composting wetlands generally requires final 'polishing' in an oxidation and settling pond before discharge (Younger et al. 2002; Skousen et al. 1998).

7.2.2 Design and construction considerations

i. Sizing for acidity loading: Treatment wetlands need to be carefully sized to handle expected acidity loads from drains.

Under-sizing the wetland area for the acidity loads can result in the gradual failure of the treatment system and with minimal decrease in acidity. In this event, additional cells can be added later.

Sizing is based on the amount of acidity that can be treated per m^2 of wetland per day. Internationally, a minimum treatment rate of 3.5 g of acidity (as equivalent $CaCO_3$) per m^2 wetland per day is used in designing treatment wetlands (Hedin et al. 1994; Younger et al. 2002; PIRAMID Consortium 2003). Results from a pilot wetland in the north-eastern Wheatbelt indicate that this figure is reasonable providing that an organic mix containing at least 10% non-cellulosic material is used. Further detail on this subject is covered in Section 6.3.2 on in-drain composting beds.

The size of a wetland needed to treat acidic water may be calculated as:

$$\text{Size (m}^2\text{) = daily acidity load (as g CaCO}_3\text{)/ treatment rate}$$

For example, a drain flowing at 1 L/s with an average acidity of 0.6 g $CaCO_3$/L has a daily load of 51 840 g per day (1 L/s x 86 400 seconds/day x 0.6 g acidity/L). This requires a wetland treatment area of 51 840/3.5 = 14 811m^2 or **approximately 1.5 hectares**.

Wetland size can be reduced by using better quality organic materials as higher treatment rates require smaller areas to treat the same acidity load. For example, higher treatment rates are likely with organic mixes such as composted lupin residues and sheep manure or harvester chaff residues and sheep manure than straw and sheep manure (Santini et al. 2009; see Section 6.3.2).

The treatment rate in composting wetlands can be high during the initial months of operation (see case study below) and will generally decline over time, most likely due to decreased decomposition rates as the more readily decomposable organic residues are depleted.

ii. Wetland siting: It is best practice to select the optimum site for placing a treatment wetland and constructing drains to deliver to this point rather than prescribe the siting of treatment wetlands after constructing a drain discharge point.

Composting wetlands require large, flat areas with a long history of shallow groundwater (within 1.5 m of the ground surface) preferably on the margins of floodways or in groundwater discharge zones, although siting within degraded playas and clay pans might be considered.

Shallow groundwater is required to ensure that the materials in the wetland will at least remain moist during summer (by base contact with the watertable) over the long term if drain inflows cease. Staying flooded or at least moist (after decommissioning) is important in retaining the acidity stored within the composting wetland over the long term (Younger et al. 2002). If a wetland is to be sited in an area where the watertable is deeper than 1.5 m, the design will need to consider leakage of water and possible mounding of the watertable in the area around the wetland.

Construction at a site with subsoil carbonate materials would also be worthwhile, since these can be used to cap and be part of a dry cover to prevent infiltration if the wetland is decommissioned (see PIRAMID Consortium 2003).

iii. Earthworks: A single treatment system is best designed as a series of smaller wetland cells (Fig. 31) to allow flow management and maintenance (for example, being able to shut down treatment cells for restoration work).

In its simplest form, individual wetland cells can consist of a linear pit with length at least 20 times the width (e.g. cells of 100 m length x 3 m width). The intention is to ensure that water flows the longest possible path through organic matter in each treatment cell. Using meandering pools (as treatment cells) and gently sculpted banks might be necessary if aesthetics are important (Younger et al. 2002). Banks with gradients of 1:2 with top-slopes draining away may be more appropriate to reduce the erosion of clay into the wetland cells. Eroded clays tend to settle and form thin layers that may slow treatment rates over the long term.

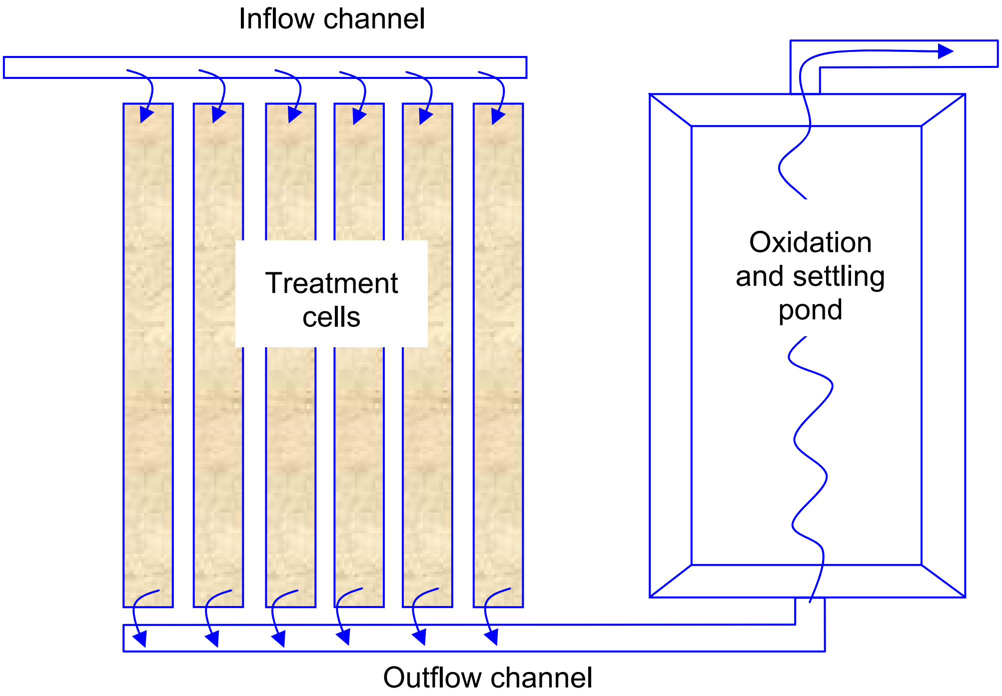


Figure 31 Outline plan diagram of a simple treatment wetland with a series of linear treatment cells and outflow to an oxidation and settling pond prior to discharge

The cells need to be narrow enough to allow access from the side for maintenance (wider cells may be difficult to refill with organic matter) and reduce the effort required to load organic matter into the wetland. For example, it may be easier to confine the width to the length of 0.5 tonne straw bales (2.4 m) than trying to spread the straw from these across a 4 m pit.

The depth of the wetland pits is not critical but needs to be not more than 1 m, allowing room for 0.3–0.4 m of organic materials and a maximum of 0.3 m freeboard for water overlying this layer (Younger et al. 2002). Construction with compost deeper than 0.5 m offers little additional benefit for the greater expense. Deeper cells may be needed if the minimum depth to the regional groundwater is greater than 1.5 m (in winter–spring). Shallow or above-ground wetland designs would only be suitable if leakage was controlled by using liners or other impervious barriers. This may be necessary to reduce the mounding of the watertable beneath the wetland which is less likely to be evident if the wetland base is cut into the local watertable.

Where wetlands are sited within floodways, erosion risk can be minimised by minimising the depth of the surface level below the surrounding landscape (to < 0.5 m) or using levees.

iv. Organic matter fuel: A mix of faster (non-cellulosic) and slower (cellulosic or woody) decomposing organic matter is best for optimum operation of composting wetlands. The details are covered in the section on choice of organic materials for in-drain composting beds (Section 6.3.2). It is important to keep in mind that a balance of materials is needed to provide sustained fuelling of microbial activity over many years and that ratios of cellulosic to non-cellulosic organic materials are based on volumes when packed into a wetland and not compacted or baled volumes.

v. Organic matter arrangement: Organic materials need to be at least 0.3 m deep in the base of the wetland cells and need to be no more than 0.5 m deep (Younger et al. 2002). The simplest arrangement involves mixing materials and placing them as a single layer with water flowing over the surface (Fig. 32a). Sulfate reduction is encouraged throughout the organic mix with treatment limited by rate of acidity seeping down (red arrows; Fig. 32a) and/or alkalinity seeping up (green arrows; Fig. 32a) across the water–organic matter interface. To achieve adequate mixing of some organic materials it may be necessary to screen or mill the materials.

Placing the higher quality material at depth may be more practical in some cases (Fig. 32b). This layered design promotes the greatest fuelling of bacterial sulfate reduction in the zone closest to the wetland base and ensures that the sulfidic black oozes (where acidity is stored) form near the base. This may better protect the treatment zones and materials forming in these from acidity-flushing events since water flows over the top.

A number of alternative, but untested, layering options are outlined in Figure 32, mostly differing in being more applicable to water flowing through rather than over the organic materials. Faster decomposing organic matter that readily forms chunks or aggregates could be roughly incorporated with slower decomposing material (Fig. 32c), which often has higher hydraulic conductivity. This might also occur if materials are inadequately mixed for the simple mixed design (Fig. 32a). Alternatively, the material could be placed as intermittent layers (Fig. 32d) or concentrated into cells (Fig. 32e). These configurations potentially allow more area for exchange between pockets of bacterial sulfate reduction and surrounding water and may thereby enable higher treatment rates than single layer configurations.

For all organic matter loading configurations, water can flow over the organic matter and does not need to be forced through the material. Vertical flow (rather than horizontal) through composting organic layers has been trialled with some success in the USA for treatment of acid mine drainage (Gusek 2002; Doshi 2006). This flow path was achieved using slotted pipes to drain water through often deeper composting layers, with riser outlets on the outside wall to control water level.

Some compaction may be needed in placing the organic materials into the treatment cells, though compaction using heavy machinery is best avoided to maintain hydraulic conductivity at no less than 0.01–1 m/day (Younger et al. 2002).

Finishing the surface layer with a series of ridges perpendicular to the flow path will reduce flow short-circuiting across the top of the organic materials and encourage water to flow through the shallowest materials.

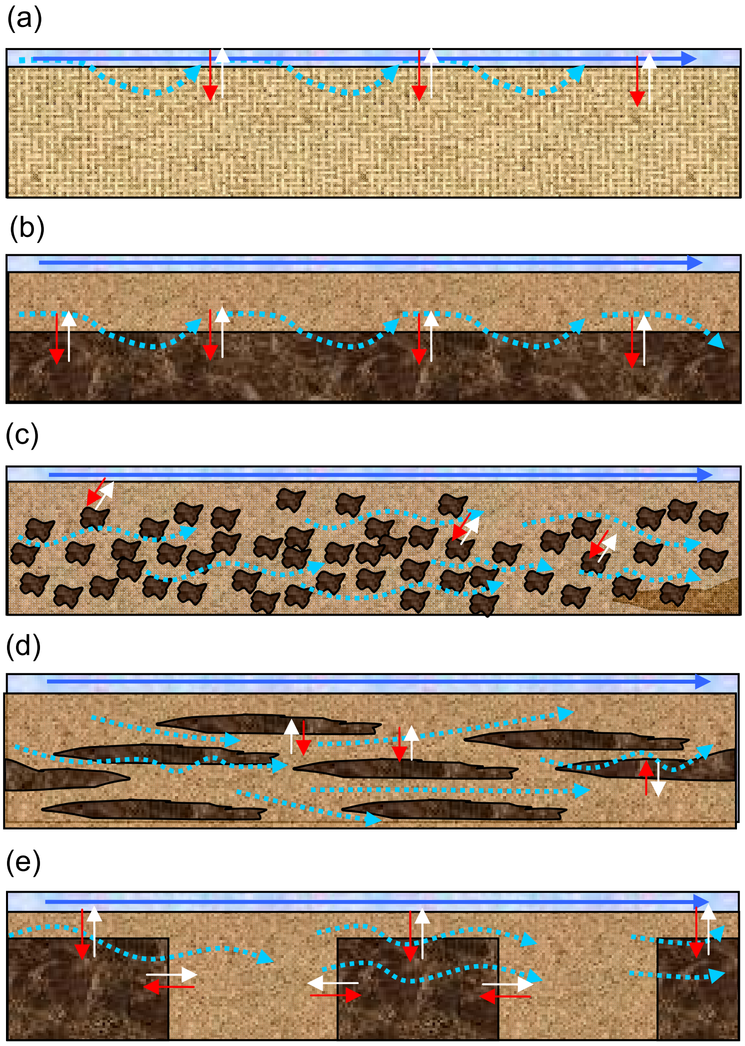


Figure 32 Examples of arranging faster decomposing (non-cellulosic) organic materials (dark hatching) in relation to slower decomposing (cellulosic) materials (lighter hatching) in composting wetland cells as (a) a mixed single layer (b) with higher quality materials as a layer at depth or (c) with higher quality materials as chunks (d) overlapping lenses or (e) cells within slower decomposing material. Light blue arrows indicate flow paths and red and white arrows indicate acidity and alkalinity seepage paths, respectively.

vi. Water level requirements: The operation of the treatment wetland relies on flooding and permanent inundation of the organic mix. Common composting wetland designs used in the mining industry (Hedin et al. 1994; PIRAMID Consortium 2003) involve flow over the waterlogged organic materials with treatment limited to diffusion across the organic matter–water boundary. Construction with 0.2–0.3 m freeboard above the wetland will allow for pooling of water and some capacity to store and treat water from drains, perhaps after high flows (see Section 8).

Maintaining water levels below the surface of the organic layer will force more flow through the organic materials and may achieve higher treatment rates per m^2 wetland area. Forcing water through the organic matrix allows for greater contact between acidic water and bacterial sulfate-reducing zones, so increasing treatment per unit area. But this approach may result in less protection against acidity overloading should treatment rates decline with time relative to acidity loading. Flows through the organic materials may also gradually decrease as metal sludges (particularly black oozes) accumulate over the long term, although this has been shown not to occur with aluminium-rich waters (Gusek & Wildeman 2002) common in many acidic Wheatbelt drains.

vii. Inlet and outlet structures: The inlet and outlet of a composting wetland need to be durable and sized to handle the expected flows through the system. When choosing pipe materials, exposure to sunlight, the effects of elevated temperatures and corrosive untreated water need to be considered. Experience with polypropylene piping in the Eastern Wheatbelt where it is widely used in smaller culverts has found that it can be sufficiently durable. Inlet design should aim for even flow distribution into the wetland and can be achieved using an inlet pool with a broad overflow weir or a pipe with multiple outlets (see PIRAMID Consortium 2003 & Fig. 33).

Outlet structures can consist of weirs or pipes, provided they are low maintenance and not prone to clogging. Microbiological films and metal and organic sludges can sometimes build up in pipe outlets, though these can easily be cleaned out providing that large diameter outlets with short flow paths are used.

viii. Incorporating lime-sand/carbonates: Improved treatment may be achieved by incorporating subsoil carbonate materials sourced on-site during construction or lime-sand/crushed limestone within or beneath the organic materials. This design should not be confused with the basic reducing and alkalinity-producing system (RAPS) where acidic water is forced directly through the neutralising materials under waterlogged conditions.

Using these neutralising materials in the inlet part of a composting wetland may provide additional treatment capacity with the benefit of buffering against sudden influxes of acidity from drains. In general, unless using large enough quantities of neutralising minerals to persist for the expected operating period of the wetland, the main benefit of this design feature is in promoting the establishment of the sulfate-reducing microbes.

ix. Outflow oxidation and settling ponds: Outflow water from composting wetlands often has low concentrations of dissolved oxygen and sometimes high concentrations of dissolved iron that require removal before discharge or re-use. This ‘final polishing’ can be achieved using a combined oxidation and settling pond (Figs 31 & 33) which can be sited adjacent to the wetland or downgradient with water flowing via a shallow channel. The latter option might reduce the need for excavation (as in Fig. 33).

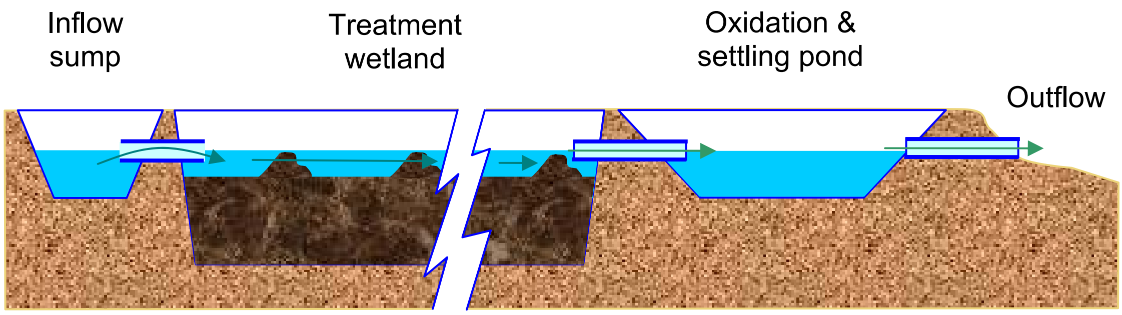


Figure 33 Conceptual layout of composting treatment wetland with inflow sump and polishing oxidation/settling pond

The design of oxidation ponds is based on the amount of iron that can be oxidised per area of pond per day, providing that it is shallow (i.e. no more than 0.5 m depth). This rate is nominally 10–20 g iron/m^2/day (Younger et al. 2002; PIRAMID Consortium 2003), based on wetlands containing wetland plants. Translated as a basic rule of thumb, outflow water with 10 mg iron/L flowing at 1 L/s (all day) requires a minimum of 86 m^2 of pond area at 0.5 m depth to oxidise the iron, providing that water pH is greater than 4.5 (Hedin et al. 1994; PIRAMID Consortium 2003). This means that the oxidation pond area required is directly related to the flow rate or iron concentration. For example, doubling flow or iron concentration requires doubling the pond area. Active aeration (for example, a solar-powered fountain) may also be needed in some cases to help the precipitates form where the surface area is too small or if the pond is too deep.

The basic design of settling ponds is also based on treatment per unit area, where 100 m^2 is required for every L/s inflow to settle out most solids (Younger et al. 2002). Recent recommendations are that the design should include a holding time of 48 hours (PIRAMID Consortium 2003). Effective settling also requires minimising turbulence; for example, wide ponds with baffles or submerged walls to spread flows or deep ponds (up to 3 m) with mesh barriers parallel to the pond's main flow direction. The pond (inlet to outlet) should also be at least twice as long as wide to reduce the likelihood of flows short-circuiting the settling pond (PIRAMID Consortium 2003). Sludge captured in the settling pond may need to be removed every few years and, in the case of water from treatment wetlands, is much less likely to contain metals and other elements at concentrations that may require careful disposal. More information on settling pond design can be obtained from Younger et al. (2002) and PIRAMID Consortium (2003).

7.2.3 Operational management

Although composting wetlands are intended to be passive treatment systems not requiring regular (monthly) maintenance, it is likely many will benefit from the following considerations:

i. **Start-up time:** Allow some time (up to several months) for the composting wetlands to establish before accepting inflow. This period is necessary to allow growth of the microbial community (particularly sulfate-reducing bacteria) central to treatment. This delay needs to be considered when managing acidic discharge from a drainage system.

ii. Managing acidity overloading: Overloading the treatment capacity (i.e. acidity overloading) can result in ongoing decline in performance. Reasons include death of the microbes carrying out sulfate reduction, aeration of the treatment system or insufficient organic matter. If overloading is due to sudden acidity influx, this may be addressed by stopping inflow and allowing the wetland to recover. Where overloading has occurred more gradually, possibly indicating exhaustion of organic matter, it may be necessary to construct additional treatment wetlands.

iii. Organic matter recharging: It will be necessary to periodically top up the organic materials. The frequency depends on the continuing performance of the wetland and is generally expected to be a minimum of every two years. The frequency will also depend on the mix of organic materials initially loaded into the system and mainly applies to the easily decomposable (non-cellulosic) component of the organic mix. It will not be necessary to top up the slower decomposing materials such as straw as frequently.

iv. Decommissioning: Should this be needed the materials can be retained in place and should remain waterlogged (i.e. in contact with the regional watertable). Additional protective measures can be used, such as clay capping to prevent water seeping into the organic materials and potentially oxidising and flushing accumulated solutes or capping with limestone or local carbonate-rich subsoils that might have been excavated during construction of the wetland pit (see Section 7.2.2 ii. Wetland siting). In the event that neither of these approaches is suitable, the materials can easily be excavated, mixed with limestone as per Department of Environment and Conservation Guidelines for dealing with potential acid sulfate soil materials (Department of Environment and Conservation 2009) and disposed of at an approved land-fill site.

7.2.4 Effectiveness

Short term (<6 months): Composting wetlands can be very effective in treating acidity and removing various minor metals from acidic drain water over the short term. It is widely acknowledged that these systems can often reduce concentrations of some minor metals that are difficult to remove using carbonate-based neutralisation techniques (Cohen 1996; Younger et al. 2002). So, composting wetlands are likely to be suited to situations where the minor element composition is not known or varies.

Composting treatment systems are likely to perform best with reasonably stable flow rates (i.e. varying by less than one order of magnitude), though can potentially be used in conjunction with holding basins to spread the loading from more variable flows.

Long term (>6 months): There is limited information on the long-term performance of composting wetlands under Wheatbelt conditions, though elsewhere in the world they have been reported to keep functioning for more than a decade with minimal maintenance (Ziemkiewicz et al. 2003; Younger et al. 2002; Doshi 2006). Performance over the long term is likely to be generally dependent on the composition and volumes of organic matter used at establishment (Doshi 2006). Treatment effectiveness can decline slowly over time as the organic matter progressively decomposes and soluble organic carbon is exhausted. This can be overcome by periodically recharging with organic materials (probably no more than every 2–5 years, as required).

7.2.5 Case study: Beacon composting wetland

i. **Site:** the southern margin of the main evaporation basin situated on Harold 'Flash' Beagley's property, Bimbijy Road, Beacon, Western Australia (see map in Appendix D).

ii. **Construction:** A shallow treatment cell (100 m long, 3 m wide, 1 m deep) was excavated between the wall of the main evaporation basin and sump drain surrounding the basin (Figs 34 & 35a). The cell was at right angles to the surface gradient so that water flowed along the floor of the cell. For the trial, the pit was lined with 0.5 mm thick black polyethylene plastic to minimise leakage along the length of the treatment area. Gravel was placed on top of the plastic to hold it in place, to provide a conductive flow path along the cell base and to provide a source of reactive iron (later found to be unnecessary).

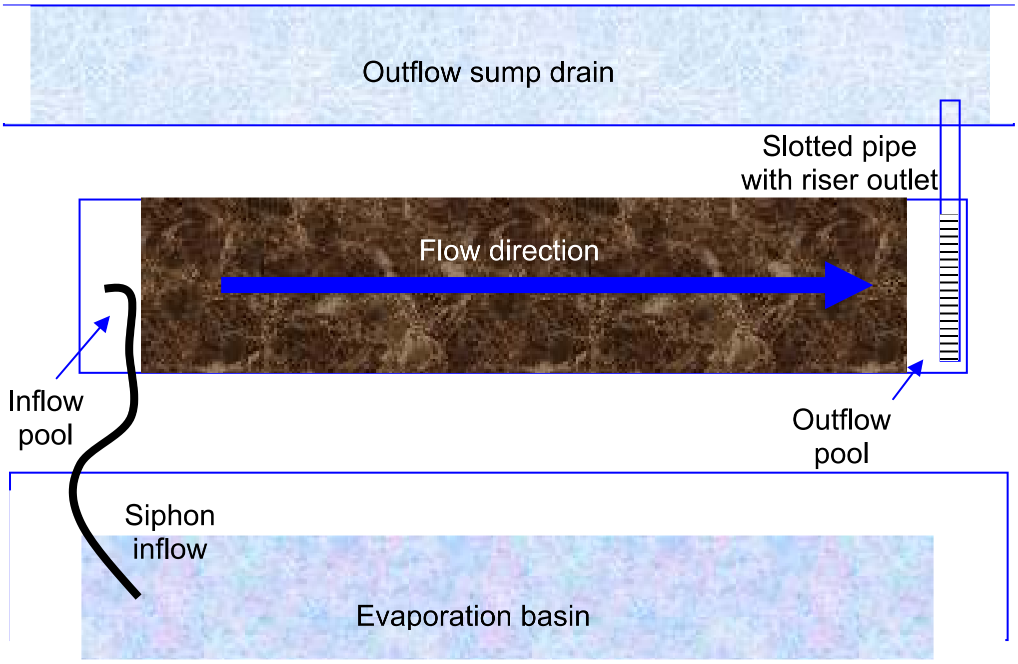


Figure 34 Plan view diagram for the pilot composting wetland constructed at Beacon

Composted sheep manure containing some sawdust (originally from sheep feed-lots and supplied by bulk suppliers in Mundaring) was loaded in 5–20 m sections along the length of the cell, with longer sections towards the wetland outlet (Fig. 35b). Approximately 6.8 m^3 of composted sheep manure was loaded and served as the non-cellulosic component of the composting mix. This was overlaid by 12 t of wheat stubble straw to an average final depth of 0.5 m, forming the cellulosic component of the composting mix. The straw was brought to the site as 0.5 t bales (Fig. 35c) sourced from Wellard Feeds, Wongan Hills. The sheep manure contained 29% (by weight) carbon, 2.3% nitrogen, and 0.7% phosphorus. The wheat stubble contained less than 0.4% nitrogen with a carbon to nitrogen ratio of at least 90:1.

An outlet pipe was buried in the ground at one end of the 100 m cell to allow water to flow into an adjoining cut-off drain. The pipe was 100 mm diameter PVC pipe placed 0.2 m below the fill level of the organic matter with a riser connected to a slotted pipe (Fig. 34). The riser was used to control the outflow level (by lengthening or shortening the riser pipe) and therefore the water level in the wetland. The slotted pipe provided a broad outflow point to reduce the risk of clogging.

The 0.5 t bales were pushed into the base and spread by hand as 0.1–0.15 m ‘biscuits’ (Fig. 35c). The final length of wetland cell loaded with straw was 97 m, which allowed for 1.5 m wide pools at either end for inflow and outflow water (Fig. 34). From a practical perspective, loading and placing straw bales in the bases of multiple treatment cells may be more easily achieved by using rolled straw (so the straw can be rolled out in the bases), or a loading system involving a side-casting conveyor belt. A skid-steer loader could perhaps be used to tease the bales apart, although this would have disrupted the liner in this trial. Alternatively, cells could have been constructed to 2.6 m width and 2.4 m depth. This would allow the 0.5 t bales to be placed without spreading, with the cell width just larger than the width to which the bales expand when the strings are cut and the bale is allowed to fall into the wetland.

After loading, the wetland cell was filled with water from the adjacent evaporation basin (Fig. 35d) and stagnated for nine weeks. This filling and stagnation was required to allow an anaerobic microbial community including sulfate reducing bacteria to establish and grow (see Section 7.2.3).

The volume of organic material in the wetland was calculated to be 145 m^3 with the final proportion of composted sheep manure estimated to be 4.7%. Note that it is important to compile organic mixtures on the basis of final mix volumes (i.e. when loaded) rather than compacted volumes (for example, when supplied as compressed straw bales). In reality, the manure could only be considered to be in contact or mixed with the bottom 0.1 m of straw (i.e. 29 m^3 straw, at most), in which case the effective mix was closer to approximately 23% by volume. The *flooded* volume, excluding inflow and outflow pools was calculated to be 131 m^3. The water volume within the wetland was not measured but was probably close to 60 m^3 (assuming a porosity of 45%).

Water levels during the evaluation were kept just at the top of the composting layer to limit surface oxidation by water, to reduce water loss by evaporation and prevent flow short-circuiting through the wetland (i.e. directly flowing from inlet to outlet) thus forcing water through the organic mix.

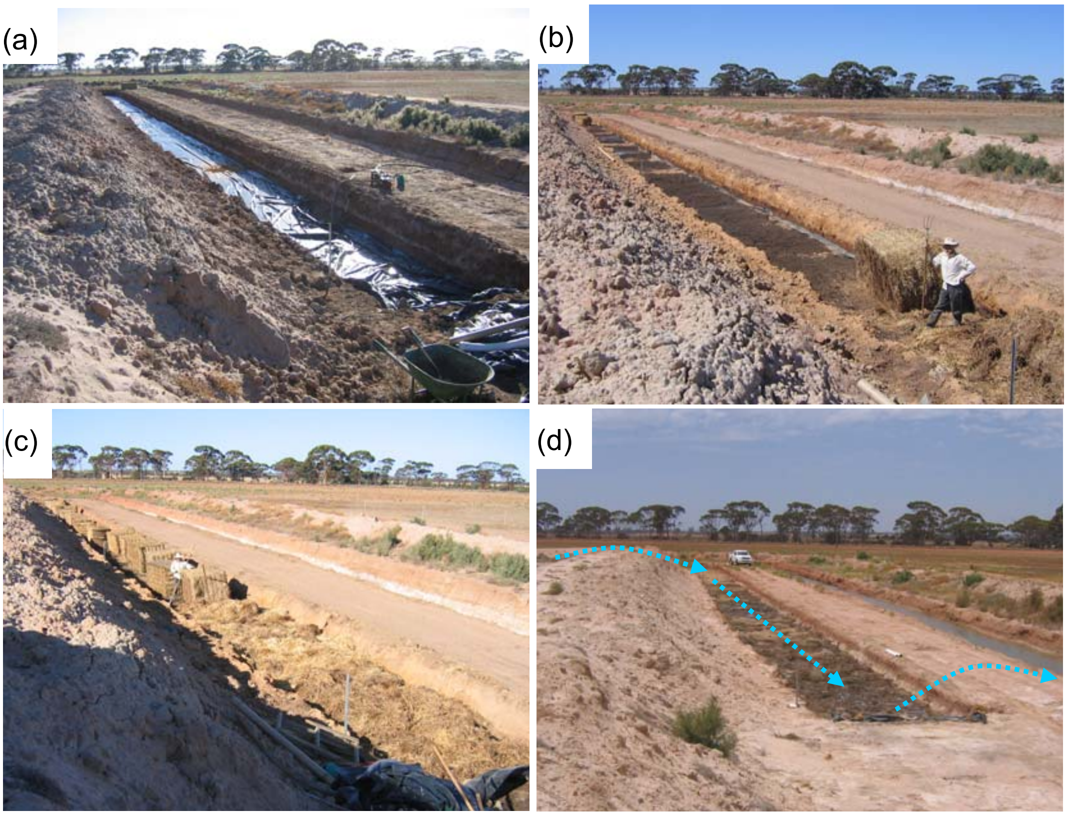


Figure 35 Composting wetland cell construction stages: (a) empty pit (b) after loading sheep manure (c) loading straw bales and (d) after filling with water. The blue arrows indicate the flow path of the water through the wetland from the evaporation basin to sump drain.

iii. Monitoring outline: The performance was estimated by regular sampling of inflow and outflow water quality combined with continuous recording of inflow using an electromagnetic flow meter (EmFlux 2020 electromagnetic flow meter with I300 transmitter). The outflow volumes were not measured because flows were frequently less than half the of inflow volumes (due to loss by evaporation) and there were no available instruments with sufficient accuracy to measure such flows (often <0.02 L/s). Instead, instantaneous outflow rates were recorded when sampled for water quality. These measurements were used to estimate acidity loads in outflows by assuming that the flow recorded at the time of visiting was the same as the flow in the period since the last visit. Water quality measurements on inflow and outflow water included on-site pH, EC, redox, alkalinity and acidity with samples taken to measure concentrations of all major ions, metals and other elements (see Appendix A). Net acidity of inflow and outflow waters was calculated as: $\text{acidity}_{\text{calculated}}$ – alkalinity (see discussion in Section 3.2.1) where: $\text{Acid}_{\text{calculated}} = 1.79 \times \text{Fe}^{2+} + 2.68 \times \text{Fe}^{3+} + 5.56 \times \text{Al} + 1.82 \times \text{Mn} + 50000 \times 10^{-\text{pH}}$ (after Watzlaf et al. 2004) and Fe, Al and Mn are concentrations of dissolved elements in mg/L (see Appendix B). All dissolved iron in inflow water was assumed to be present as Fe^{3+} given the low pH and high redox potential of the waters and all iron in outflow water was assumed to be present as Fe^{2+} as favoured by high pH and low redox potentials.

iv. Performance results: Full treatment of acidic water was achieved during the first 124 days, although effectiveness decreased after this, as illustrated by the outflow pH falling below 5 (Fig. 36) and net acidity exceeding 0.1 g $CaCO_3$/L (Fig. 37). Full treatment was achieved for the initial 432 kL, which was more than 7 times the volume of water held in the wetland. The decrease in effectiveness occurred during a period when inflows increased from an initial average of 2.5 kL/day (approximately 0.03 L/s) to more than 5.3 kL/day (approximately 0.06 L/s) and evidently overloaded the wetland with acidity. The inflow rate was decreased to try and reduce the overloading and allow the wetland to recover.

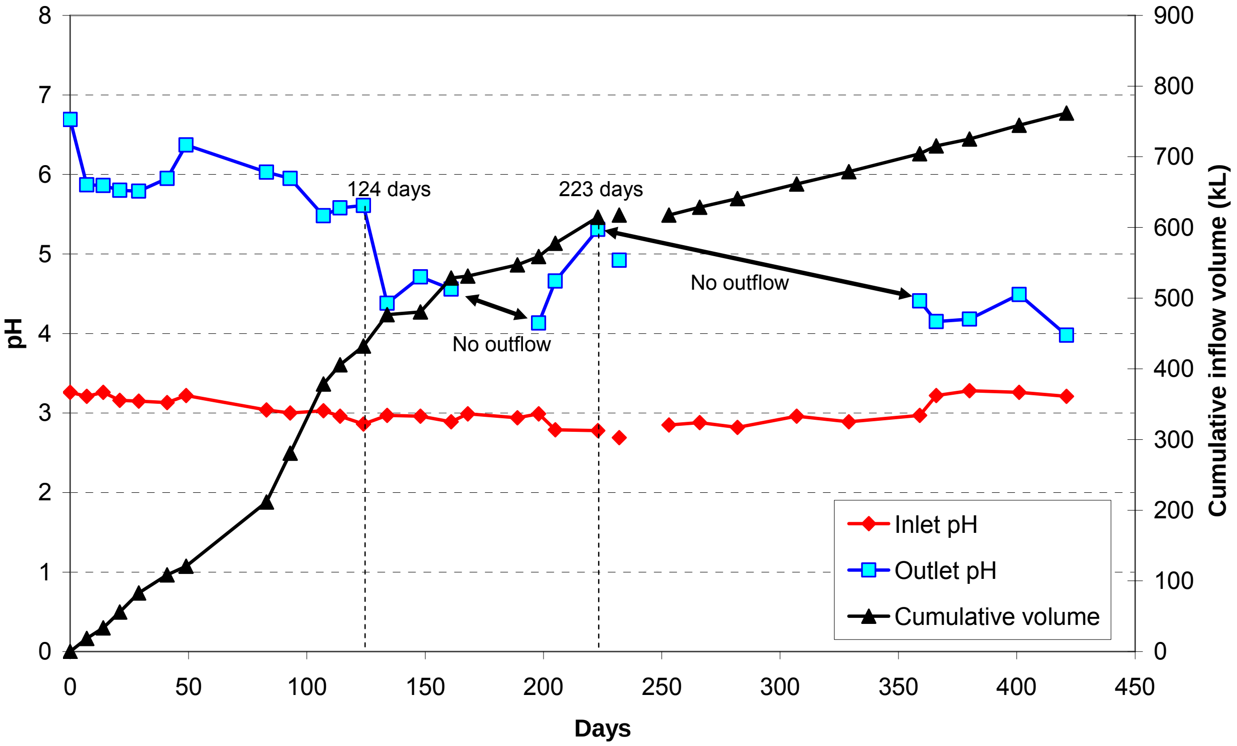


Figure 36 Inflow and outflow pH and cumulative inflow volume for the pilot composting wetland at Beacon. The absence of data points and breaks in plotted lines indicate when inflow was shutdown or there was no outflow.

Some treatment recovery occurred between 198 and 223 days, although acidity loads were difficult to control and after 223 days outflow stopped (Figs 36 & 37). Outflow stopped during mid-summer (150–250 days) when the water loss by evaporation in the wetland exceeded inflow (Fig. 36). Over the first 200 days, the net acidity of the inflow also steadily increased from 0.5–0.6 g $CaCO_3$/L during winter–spring (0–120 days) to 0.8 g $CaCO_3$/L in summer-autumn (Fig. 37). There were also difficulties in maintaining constant inflow to the wetland, resulting in variable short-term acidity loading rates (Fig. 38). Inflow was stopped for four weeks to allow recolonisation of the sulfate-reducing bacteria but only partial treatment recovery was achieved. Even though outflow water pH was less than 4, the system still reduced the acidity of the water by at least 50%.

Sluggish flows and shallow ponded water during the summer–autumn may have contributed to lowered treatment rates by enhancing oxidising conditions in the wetland. Although higher decomposition rates would be expected during warmer summer–autumn temperatures, the effects of oxidation in the surface water and its effects on the underlying organic materials could have hindered the wetland's capacity to maintain conditions suitable for bacterial sulfate reduction. Similar problems have been reported in wetlands in the UK and USA where highly oxygenated acidic water overwhelmed treatment wetlands or ongoing poor maintenance of sufficiently waterlogged conditions reduced microbial sulfate-reduction rates and treatment capacity (Doshi 2006).

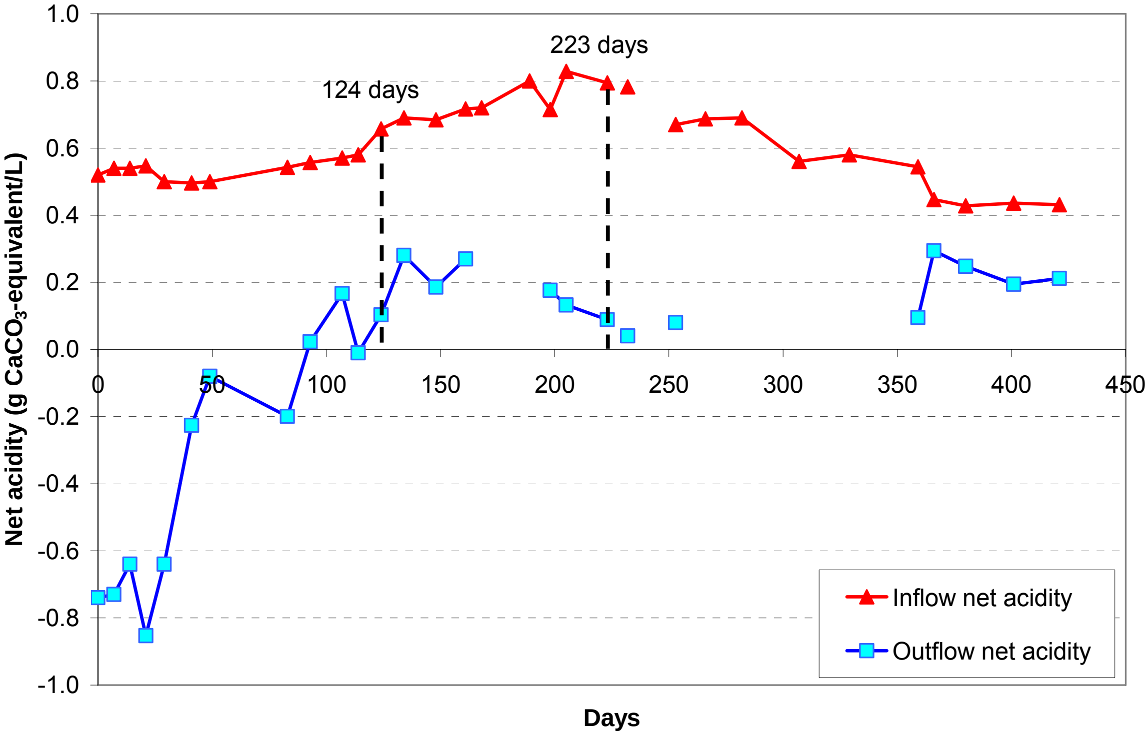


Figure 37 Inflow and outflow net acidity for the pilot composting wetland at Beacon. The absence of data points and breaks in plotted lines indicate when inflow was shut down or there was no outflow. Instances when outflow water containing alkalinity as bicarbonate are reported as negative net acidity values.

Initial wetland treatment rates were estimated to be up to 6 g $CaCO_3/m^2$ wetland/day during the first 106 days (Fig. 38), but spikes in net acidity above 0.1 g $CaCO_3$/L in outflows indicated that this was unsustainable (Fig. 37). The estimates of treatment rates during this period were similar to inflow loading rates because water from the wetland was, for the most part, fully treated. When outflows began to show that full treatment was not occurring, estimates of treatment rates required both inflow and outflow loads (but outflow volumes could not be measured continuously). So, treatment rates after 106 days had to be estimated though they were only marginally less than the acidity loading over this time, reflecting the low rates of acidity loads in outflows. The results indicated that treatment rates decreased from 6 g to 3.2 g $CaCO_3/m^2$ wetland/day from 106 to 380 days (orange line in Fig. 38). It was likely that the long-term average treatment rate was less than 3 g $CaCO_3/m^2$ wetland/day.

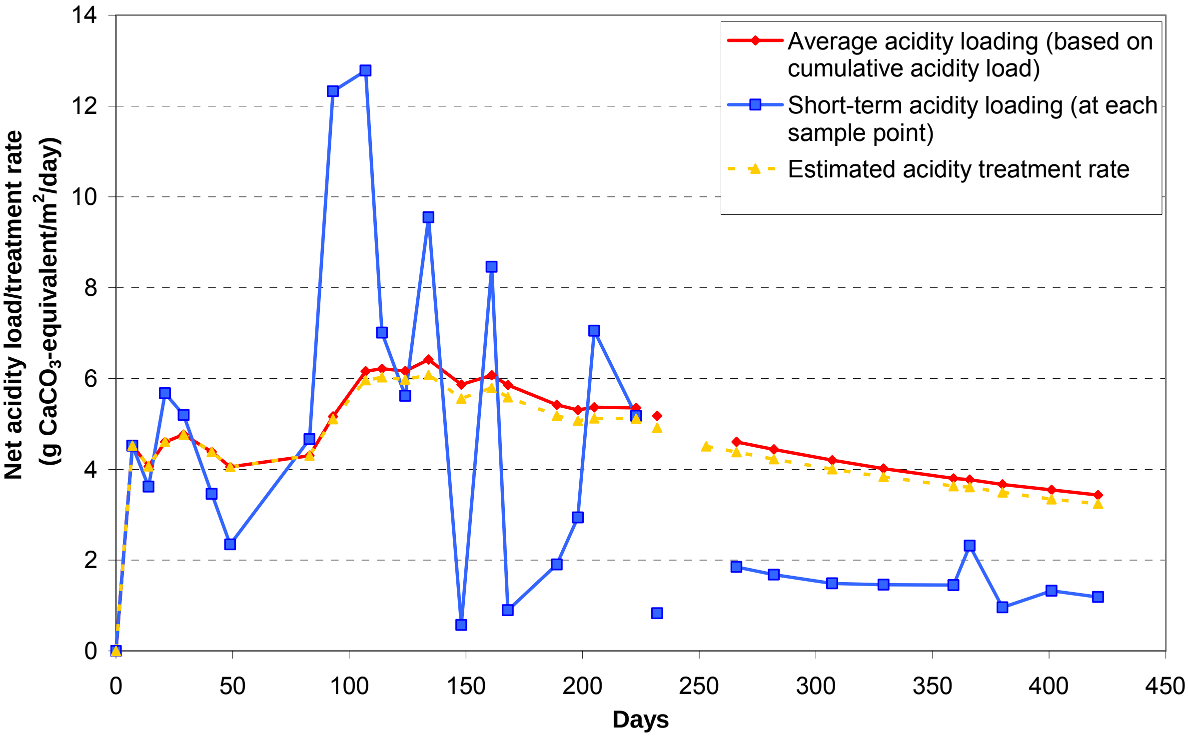


Figure 38 Average acidity loading, estimated treatment rates (correcting for estimated untreated acidity in outflows) and short-term variations in net acidity loading for the pilot composting wetland at Beacon. Absence of points and breaks in plotted lines indicate when inflow was shut down or there was no outflow.

While the wetland was achieving full water treatment, the outflow concentrations of major metals like aluminium were frequently lower than in inflows but iron concentrations were sometimes higher in the outflow (Table 6). These occasional increases in outflow concentrations (by up to 1.5 times) were partially attributed to increasing evapoconcentration as water flowed through the wetland. There was generally no reduction in manganese concentrations attributable to treatment.

The treatment wetland reduced the inflow concentrations of many minor elements (particularly lead, copper, nickel, zinc and uranium) by more than 90% while pH remained above 5.5 (Table 6). This effect persisted for elements such as copper and uranium even when outflow pH fell below 5.5. There were smaller reductions in concentrations of elements such as arsenic, cadmium, lanthanum and cerium while outflow pH was higher than 5.5 (Table 6). The outflow concentrations of most elements did not rise above inflow concentrations when outflow pH was 4–5.5 after four months of operation. This indicated that there was no flushing of accumulated metals despite the wetland not maintaining high treatment rates.

Table 6 Metal and minor element concentrations in water flowing in and out of the composting wetland treatment cell at Beacon

Grouping	Element	Inflow concentration range[1]	Outflow concentration range when pH>5.5[1]	Outflow concentration range when pH<5.5[1]
Major metals[1]		(mg/L)		
	Aluminium	76–127	0–11	6–32
	Iron	10–20	0–50	5–70
	Manganese	2.5–7	5–10	5–15
Minor elements[1]		(µg/L)		
	Arsenic 5–80		<0.4*–7 5–40	
	Cadmium <0.3	*–4.6 <0.3	* <3	*
	Cerium 1310	–4836	3–1124	563–4181
	Copper 2–34		<0.5* <0.	5*
	Lanthanum	507–1861 3–76	1 288–	1936
	Lead 474–	2097	<0.1*–2 3–28	
	Nickel 70–2	17	<0.8*–29 6–13	2
	Uranium 216–	1062	<0.3*–16 <3	*
	Zinc <3	*–50 <0.4	*–6 10–5	0

[1] The concentrations of major elements (aluminium, iron and manganese) are in units of mg/L (parts per million) while the concentrations of minor elements are in µg/L (parts per billion).

* Below analytical measurement limits. Note that the measurement limits vary; with reduced capacity to detect low element concentrations in more saline waters.

v. Design implications: The case study highlights the importance of matching inflow acidity load to the treatment capacity of a composting wetland and that simple mixtures can provide some short-term treatment. This was likely to be extended to the long term by using different organic mixtures. Other aspects of design, including sizing and flow paths, that improve function are also discussed.

Partial and full treatment of very acidic drain water (0.5–0.8 g $CaCO_3$/L) at high loading rates (4–6 g/m^2/day) was achieved for at least four months using a mix with as little as 4.7%

composted sheep manure with straw in a simple layered pit configuration, but this treatment rate was not sustained. Reducing acidity loading to less than 3 g $CaCO_3/m^2$ wetland/day should enable sustained treatment for at least 12 months. The acidity load could be reduced by reducing inflow rates (in this case), although in other situations the same effect can be achieved by using water with lower acidity, where the acidity has been reduced by some other pre-treatment.

The minimal rates of treatment (2–3 g $CaCO_3/m^2$ wetland/day) define a lower threshold for wetland design. These rates stipulate that drain flows of 1 L/s with an average acidity of 0.6 g $CaCO_3$/L will require a wetland with a surface area of up to 2.6 ha (with half the area if the water acidity is half). Trials of in-drain composting beds (see Section 6.3.5) highlight that increasing the proportion of more decomposable (non-cellulosic) organic materials with the organic mix will achieve treatment rates above those found in this case study. Altering the cellulosic component (straw) can have the same effect, at least initially (Santini et al. 2009). Increased treatment rates will correspondingly reduce the required area, but, more importantly, sustain treatment potentially beyond two years (before another addition of organic matter is needed).

A consideration for some wetland sizes and treatment rates will be the evaporation rate and whether the wetland behaves more like an evaporation basin than a flow-through system. In the example above, a 2.6 ha wetland would be expected to lose at least 260 kL per day in mid-summer, in which case outflow would stop, assuming a daily inflow of 1 L/s = 86 kL/day and average evaporation loss of 10 mm in summer. An inflow of 260 kL/day to enable outflow would require a treatment rate of at least 6 g $CaCO_3/m^2$ wetland/day and even then the outflows would besignificantly more saline than inflows. These issues would need to be accounted for where water is treated for further use.

Flow forced through, rather than over, the organic mix may have reduced the wetland’s capacity to withstand overloading and limited the rate of recovery. Monitoring pH and redox profiles during the trial indicated that acidic water permeated throughout the straw layer along most of the length of the wetland cell. This monitoring also indicated that much of the straw was probably not contributing to sulfate-reduction processes since redox levels in the organic bed were generally favouring iron reduction but not low enough to favour bacterial sulfate reduction. In many other situations with composting wetland treatment, acidic water generally flows over the surface (see Section 7.2.2), which effectively limits the extent to which the acidic water can interact and affect sulfate-reducing bacterial activity (given that these organisms and the black oozes formed by them are pH sensitive).

Concentrating the more easily decomposable material in the base of the system was probably the main factor enabling the overall low proportion of faster decomposing material to work effectively. If the composted sheep manure had been mixed throughout the full straw layer the treatment rates may have been lower.

Take home message: Composting wetlands constructed using locally available materials have the potential to fully treat acidity and remove a range of minor elements from acidic drain water over long periods of time with minimal maintenance.

7.3 Lime-sand reactors

7.3.1 Principle

Lime-sand reactors are a form of active neutralisation at the point of pumping, using lime-sand in an up-flow tank system that can be constructed from widely available materials.

The technology is based on lime-sand based treatment systems developed in the USA for waters with acidity up to 1 g $CaCO_3$/L and containing more than 100 mg Fe/L (Hammarstrom et al. 2003; Sibrell et al. 2003; Watten et al. 2005). These more advanced systems use a combination of pulsed flow and CO_2 gas carbonation systems to achieve sustained neutralisation. The design outline here is for a simple pulsed flow system used as an end-of-drain solution, integrated with holding and/or settling ponds.

7.3.2 Design and construction considerations

The best treatment for water containing soluble iron (>10 mg/L) is achieved by intermittent pulsing of water through a bed of lime-sand in an up-flow configuration (outlined below). Carbonation of inflow water with CO_2 gas recovery from outflow water (to re-use in inflows) can greatly contribute to treatment effectiveness (Sibrell et al. 2003; Watten et al. 2005) but requires specialised engineering for larger capacity systems. The design of carbonation and CO_2 recirculation systems will not be covered here, but is broadly outlined in the work by Sibrell et al. 2003 and Watten et al. 2005 (currently the subject of a US patent application).

i. Tank reactor column: A cylindrical reactor column can be suitable although conical systems can be more effective in preventing sand loss in continuous flow conditions (Evans et al. 2007; Vuthaluru et al. 2009). Local trials (see case study in Section 7.3.5) have had some success using a simple narrow (<50 cm) diameter polypropylene tank with water pumped through a diffuser plate to encourage laminar vertical up-flow (Fig. 39). It is important to ensure that the cross-sectional area of the holes in the diffuser plate is sufficient to avoid excessive back pressure on the pump while avoiding sand bridging on top of the plate (5 times pipe inlet cross-sectional area was effective in the case study).

ii. Water pre-treatment: Treatment performance is likely to be improved if the dissolved iron concentrations in drain water are reduced before pumping through a reactor. Although lime-sand reactor design and operation can be adapted to treat water with high concentrations of soluble iron (>100 mg/L; e.g. Sibrell et al. 2003), storage, allowing aeration of drain water and oxidation and settling of iron before treatment, will reduce the pumping and mixing time required for treatment (see Section 7.1.2 i. Water pre-treatment).

iii. Lime-sand specifications: Treatment performance will be greatest using high-grade, small-grained carbonate materials (assaying as >90% $CaCO_3$). With regard to performance efficiency, grain size is more important than purity (Sherlock 1997), so the disadvantages of lower grade materials may be offset by crushing (see Section 6.1.2 ii. for further information). Higher performance might also be achieved using neutralising materials with mixtures of crushed limestone and hydrated lime (calcium hydroxide), although these must be formulated to ensure that the hydrated lime content does not result in over-treatment (i.e. pH reaching 9 or 10 as can occur if excess hydrated lime is used).

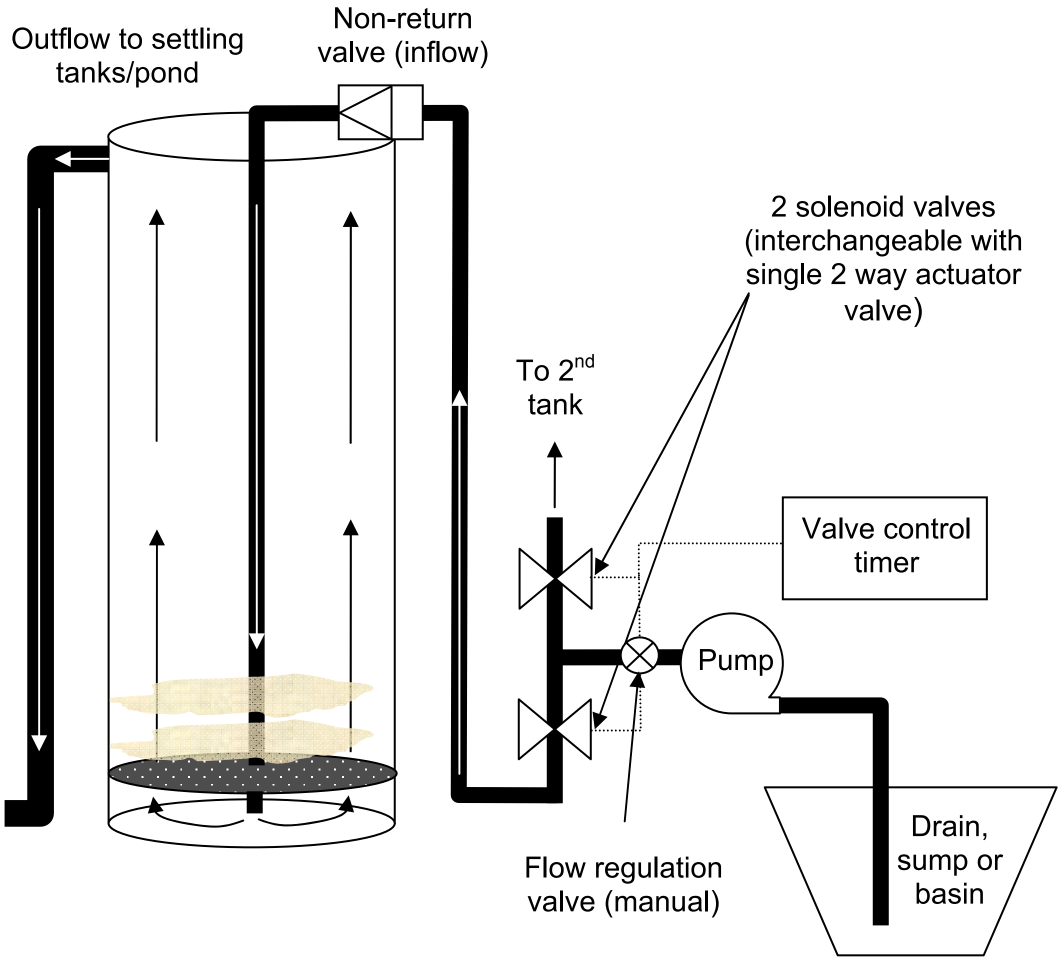


Figure 39 Conceptual schematic of a pulsed up-flow lime-sand reactor

iv. Power supply: This treatment requires a continuous power supply to run the pump. It might be possible to drive the system intermittently with a direct feed solar-powered pump (i.e. no battery storage) if sized to ensure treatment of sufficient water volume. Water could be pumped intermittently (during daylight hours) from one holding cell to another with treatment through a pulsed-flow fluidised tank reactor.

v. Pump: Continuous flow is required, with at least 10 m of head pressure to provide start-up pressure at the beginning of each pulse. The pump and impeller may need to be glass-reinforced polyester or stainless steel to ensure an operating life of at least several years (pumps for transfer of liquid fertiliser nitrogen may be suitable).

vi. Lime-sand feed: The lime-sand in the tank will need to be intermittently replaced/topped up as the material is consumed during neutralisation. This will require addition of fresh lime-sand every few days, depending on the rate of consumption through the tank. It will also be necessary to completely replace the lime-sand every few months as quartz sand and other insoluble contaminants build up in the reactor. Regular lime-sand topping up might be achieved using a sand-screw feed with hopper and auto-timed operation to deliver lime-sand through the top of the reactor.

vii. Oxidation and settling tanks or ponds: After neutralising the water, precipitates will begin to form (mostly iron- and aluminium-rich sludges). Aeration may be required to facilitate oxidation of dissolved iron and the formation of iron-based precipitates, followed by settling of sludges before water is discharged to creeks or other uses. This is an important part of the treatment process because, in many cases, the sludges that form trap minor metals after neutralisation of the water (see Section 3.1 on the basic chemistry of acidic water treatment). Oxidation of dissolved iron can be maximised by aerating water before discharging into a settling pond (e.g. using a fountain outlet) with the pond designed to hold the water long enough to allow oxidation. Details on the designs of outflow oxidation and settling ponds have been discussed previously (see Section 7.2.2 ix.).

Where space is limited or precipitates forming in the water after neutralisation do not settle well, it may be necessary to use more complex water treatment systems; for example, clarifiers, lamellar plate thickeners, adding coagulants and flocculators etc. (see Metcalf & Eddy Inc. et al. 2003).

viii. Pumping rates and hydraulic residence time requirements: The optimal pumping rate is when the upward velocity of water flow in the tank vigorously mixes the sand during each pulse without ‘blowing’ it out of the column. In effect, the upward flow velocity is governed by the rise in water level with each pulse. This must be less than the settling time of the finer sand between pulses or the sand will remain continuously fluidised which will increase the rate of armouring by iron oxides. For the lime-sand from Cervantes (80% by weight 125–250 μm diameter) used in the field evaluations at Beacon (see case study Section 7.3.5), it was necessary to pump at rates resulting in no more than 0.55 cm/s vertical rise in water for each pulse. Higher pulsed flow velocities are probably possible with coarser graded lime-sand or crushed limestone and with conical reactors (e.g. Evans et al. 2007) without loss of the neutralising materials.

ix. Flow conditions: Sufficient turbulent flow velocity is required to minimise armouring of lime-sand by iron oxides. This is best achieved by pulsing (e.g. Sibrell et al. 2003; Watten et al. 2005), since continuous flow and fluidisation of lime-sand generally results in armouring (Santomartino & Webb 2007). Concentrations of soluble iron as low as 10 mg/L in drain water can significantly limit the effectiveness of the lime-sand by armouring within several hours of operation (see case studies). A pulsing frequency of one to two minutes up-flow with several minutes without flow can be successful, but individual systems may need variations on this to achieve good treatment.

A number of factors need to be considered: If pulsing is too infrequent, the dissolved iron may armour the lime-sand too much, thereby reducing treatment effectiveness. If pulsing is too frequent, the sand in the column may not settle between pulses and will stay continuously suspended with little abrasive action between sand grains during pulses. Using two or more tank reactors with flow alternating between these allows a pump to run continuously (i.e. pumping into one tank, then switching to other tank(s) while the first settles).

x. Recirculation: Additional treatment may be achieved by recirculating treated water through the tank several times before discharging. This option requires installation of additional solenoid control valves, piping and associated timer controls (or programmable integrated circuit controllers) to enable outflow from the tank to be returned through the pump intake. Recirculation allows more time for the lime-sand to dissolve and react with the water and, if the system is sealed from the atmosphere, results in greater efficiency of lime-sand neutralisation rates as carbon dioxide gas levels build up in the recirculating water (Hammarstrom et al. 2003; Watten et al. 2005). Similar pulsed up-flow tank reactors in the USA (e.g. Sibrell et al. 2003; Watten et al. 2005) using recirculation combined with carbon dioxide gas injection have achieved full neutralisation of water similar to those encountered in acidic deep drains in the WA Wheatbelt.

xi. Pump–reactor–pond configurations: Overall success in treatment (if used as a sole option) may depend on staged treatment since the single passage of water through a lime-sand reactor may not consistently neutralise all acidity and enable removal of metals.

Staged treatment conceptually involves using several holding ponds with several lime-sand reactors potentially connected in series (or one reactor series with alternating water feed from drain or oxidation pond). This approach may be needed to allow dissolved iron to oxidise in water after initial treatment through a reactor, in particular for drain water containing more than 20 mg iron/L. Water might be pumped from a drain through a reactor into one pond, allowed to oxidise (react with the air) and sludges to settle, before being pumped through a second reactor (or back through the first) into a second pond to allow remaining iron sludges and minor metals to settle out after the final treatment (Fig. 40). Oxidation and settling ponds can also be integrated with lime-sand basins to achieve further treatment and maximise the use of the area, though sludges accumulating on any lime-sand in the bases may limit the benefits.

7.3.3 Operational management

i. Maintenance and servicing: Regular (at least fortnightly) checking of treatment performance and unit servicing will be required. It is possible that operation of the unit (pH of outflow water, pump operation) be automatically monitored and reported by telemetry (transmitted automatically by mobile phone hook-up) to an office. Every few months it may also be necessary to completely drain the reactor to flush out non-reactive sand that is likely to build up with continuous operation. Local lime-sands can contain up to 10% silica sand which will accumulate in the reactor as calcium carbonate sands are consumed. Depending on the quality of the solenoid valves used in construction, these may also need checking and replacing as required.

ii. Neutralising agents: Regular reloading of lime-sand will be required, depending on the acidity load being treated.

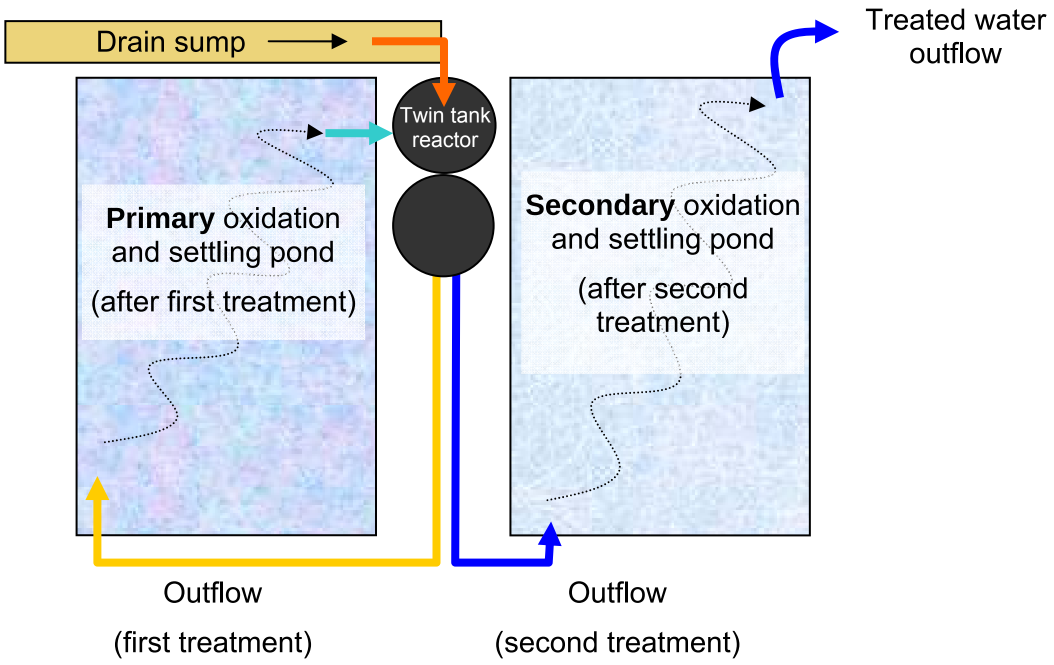


Figure 40 Schematic diagram of a two-staged treatment system using a single twin-tank lime-sand tank reactor with water initially pumped from the drain sump into the primary pond and treated a second time with discharge to the secondary pond.

iii. Sludge removal and disposal: Sludges formed during neutralisation will need to be removed periodically from settling tanks and ponds. Sludge removal is particularly critical where treated water is discharged to creeks, floodways or lakes to avoid flushing the materials and possibly transporting into the environment any trapped metals. The frequency of sludge removal will depend on the acidity load being treated, reflected by the rate of lime-sand consumption: if tonnes of lime-sand are consumed per week it is likely that tonnes of sludge are accumulating per week.

How the sludge is disposed of will depend on the concentrations of metals in the water being treated. Sludges formed from acidic drain water with high concentrations of minor metals such as lead, arsenic, nickel, cadmium, copper and zinc may need to be treated as contaminated wastes and disposed of with due care. For further information on this subject see the Department of Environment and Conservation guidelines for waste classification and management (Department of Environment 2004).

Conservative estimates of minor elements likely to occur in sludges indicate that most could contain at least one element that will require care in management and disposal, particularly from an ecological perspective. Concentrations of minor elements that might occur in sludges were estimated from analyses of acidic drain water across the eastern Wheatbelt surveyed between 2004 and 2008 (Shand & Degens 2008). It was assumed that, for every kL of water, all the iron and aluminium would form sludges on which the dissolved minor elements would be adsorbed (Table 7). This assumption provides a worst-case scenario estimate of sludge minor element concentration because most sludges would contain sand, other clays and salts which would dilute the final minor element concentration.

Concentrations of lead, nickel and zinc in sludges formed from most drain water might pose a risk to some ecosystems, on the basis of Department of Environment and Conservation sediment investigation levels (Table 7). Selenium and, to a lesser extent, arsenic and cadmium can also potentially be contaminants of concern in sludges though it is noted that many lake sediments naturally contain high concentrations of selenium (Shand & Degens 2008). The best practice is to prevent sludges from entering creeks, floodways and lakes since the materials often have very low density, do not settle well and are very prone to being transported by low velocity flows.

7.3.4 Effectiveness

Short term (<6 months): Lime-sand reactors can be very effective over the short term, although dependent on setting pumping rates to achieve maximum treatment and maintenance to ensure continuous supply of sand and removal of unreacted or iron-oxide armoured sand from the column as required. The effectiveness of minor element removal by this treatment will depend greatly on how high pH is raised (at least above pH 5.5) and whether the iron and aluminium sludges remove minor elements by adsorption or co-precipitation in the settling ponds after neutralisation. Other local investigations (see case studies in Sections 7.1.5 & 7.3.5) have found that carbonate-based treatment approaches can be effective in reducing concentrations of some minor elements (e.g. lead, uranium, cerium and lanthanum) but not others (e.g. nickel, cadmium).

As with any carbonate-based treatment approach, armouring of the sand with iron oxides presents the most significant limitation to effectiveness. Despite this, simple pulsed up-flow treatment (see case study below in Section 7.3.5) can sustain treatment of at least 0.3 g $CaCO_3$/L (at 2–2.3 kL per hour) while minimising the effects of armouring on lime-sand reactivity. This would be expected to also maximise the efficiency of lime-sand use, which is the percentage of lime-sand available for the neutralisation process. Although lime-sand consists of up to 96% calcium carbonate, this may not remain completely reactive if heavy armouring occurs. Even when significant armouring occurs, some treatment (at least 0.2 g $CaCO_3$/L) can still occur during a single pass through a reactor column, although metals and minor elements, except for dissolved aluminium, are not removed under such conditions.

Overall success in treatment (if used as a sole approach) will depend on staged water treatment to progressively achieve neutralisation. This will probably be needed in many cases with many simple reactors because a single passage of water through a lime-sand reactor is unlikely to consistently neutralise all acidity and remove metals.

Progressive neutralisation might involve treating water in batches or continuous treatment and flow through ponds. The effectiveness will depend on the pond design to allow oxidation of dissolved iron and settling of precipitates between successive treatments.

Long term (>6 months): The long-term performance is expected to depend more on continuity of power supply, lime-sand supply and maintenance than the treatment effectiveness. If continuous treatment is essential for treatment before further use or discharge to creeks or lakes these factors will be critical for design. The design outlined here is best used in the long term for treating water at the end of a drain in combination with temporary holding ponds.

This approach, although requiring investment in pond design and construction, provides some protection and buffering against gradual or sudden failure in the treatment system.

Table 7 Estimated worst-case scenario concentrations of minor elements in drain sludges and the number of drains with sludges requiring further ecological investigations for safe management of specific trace metals and if the materials would be classified as hazardous waste

		No. of drains where expected concentration in sludge may:		
Trace element (number of drains)	Range of expected concentrations in sludge[1] (mg/kg dry weight)	Exceed **sediment ecological** investigation level[2]	Exceed **soil ecological** investigation level[2]	Exceed levels for Class II landfill disposal[3]
Arsenic	0–53 0	*	2*	2*
(70)		(70)[2]	(20)[2]	(14)[3]
Cadmium	0–30 1	*	2*	3*
(41)		(20)[2]	(3)[2]	(0.4)[3]
Copper	0–2198 4		26	0
(75)		(270)[2]	(60)[2]	(5%=50000)[3]
Lead	0–4902 49		37	75
(75)		(220)[2]	(30)[2]	(2)[3]
Nickel	0–3639 72		70	72
(72)		(52)[2]	(60)[2]	(4)[3]
Selenium	0–194 14		N/A 14	
(70)		(2)[4]		(2)[3]
Zinc	0–534 20		37	0
(74)		(410)[2]	(200)[2]	(5%=50000)[3]

[1] Concentrations estimated from survey results of 75 drains across the eastern Wheatbelt (Shand & Degens 2008) by calculating dry weight of aluminium and iron precipitates (solely as $Al(OH)_3$ and FeOOH, respectively) precipitated per litre of water and assuming direct adsorption of minor metals and elements onto these precipitates.

[2] Department of Environment and Conservation upper ecological investigation levels (mg/kg dry weight) for minor element in soils and/or sediments (Department of Environment 2003).

[3] Department of Environment and Conservation – Contaminant threshold concentration (mg/kg dry weight) for Class I and II waste facilities not requiring leachate tests (Department of Environment 2004). Note: many local governments manage Class I and II landfill sites.

[4] Using suggested safe selenium concentration of 2 mg/kg in sediments (Lemly 2002).

* Most samples had concentrations of minor elements below detection limits, so potential sludge concentrations could not be assessed.

7.3.5 Case study: Beacon lime-sand reactor

i. Site: The main evaporation basin of the Engineering Evaluation Initiative drain evaluation site was situated on Harold ‘Flash’ Beagley’s property, Bimbijy Rd, Beacon (see location map in Appendix D). For the evaluation, treatment trials used acidic water held in the evaporation basin rather than drain water collected in an adjacent sump, where water quality varied over 24–48 hours due to pumping and refilling of the sump.

ii. Construction: Two tank reactor configurations were consecutively constructed and trialled. They consisted of a percolation tank reactor intended to contain a large volume of sand (needing less frequent refilling) and an up-flow reactor with continuous and pulsed flow regimes. Water was pumped using a 0.7 kW aquaculture pump (ONGA Seabass 9), selected for being able to pump saline water efficiently for long periods. The source water for all trials was taken from the Beacon EEI evaporation basin, principally because the chemistry of this water body was more stable over trial periods than the adjacent drain waters. Lime-sand for all trials was sourced in bulk from the AgLime Australia coastal lime-sand pit at Cervantes (91% $CaCO_3$ by weight with over 80% of particles 0.125–0.250 mm diameter).

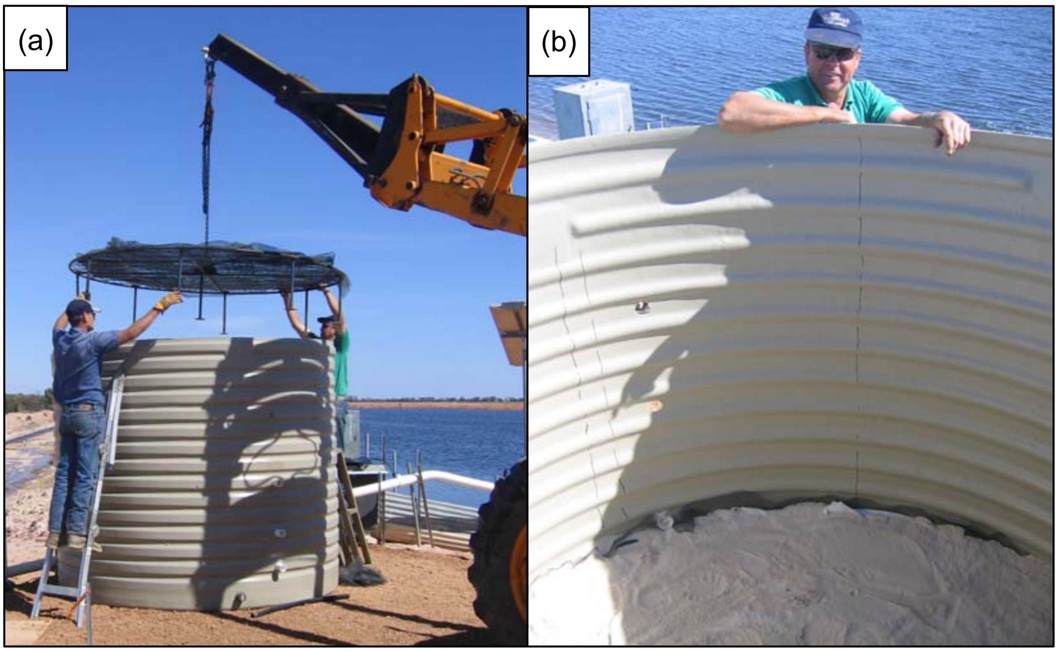


Figure 41 Construction of the percolation tank reactor illustrating (a) lowering the percolation tables through the top of the tank and (b) placing lime-sand on fine nylon mesh on each table

Percolation tank reactor: A 9000 L tank (2.4 m internal diameter x 1.8 m height) had four circular tables on which lime-sand was suspended on a layer of fine nylon mesh overlying flywire and weldmesh (Fig. 41). Each table was later fitted with 150 mm rims to prevent water flowing over the edges and bypassing the lime-sand. Water was pumped into the top of the tank via 50 mm diameter polyethylene piping and sprayed across the lime-sand at a rate allowing percolation through the lime-sand on each table before flowing (under gravity) out of the base of the tank through 90 mm diameter PVC piping (Fig. 42). The tank was loaded with

1.5 t lime-sand distributed between the tables and the base and water pumped into the system at 0.5–1 L/s.

Continuous up-flow tank reactor: A 2800 L tank (0.59 m internal diameter x 3.0 m height) was constructed from a polypropylene externally corrugated culvert and held in place with a galvanised steel support structure (Fig. 43a). Water was pumped through the centre of the tank through 50 mm diameter PVC piping, directed against the base and distributed through a diffuser (25 mm polyethylene disc with 70 x 16 mm holes; Fig. 43b). Water flowed upwards in the tank and exited via 90 mm diameter stormwater PVC piping at the top. The tank was loaded with 450 kg lime-sand and flow pumped through continuously at 0.62–1.32 L/s.

Figure 42 Flow path through four tables on which lime-sand was suspended in the percolation tank reactor

Pulsed up-flow tank reactor: This model was the up-flow tank reactor (Fig. 43) configured to achieve pulsed flow using two 24 V, 50 mm diameter solenoid valves controlled by an interval timer with 24 V relay switch (Jaycar Electronics Kit KA1732). The valves worked concurrently to bypass pump flow to the evaporation basin (i.e. one valve opened concurrently with the other valve closing). This enabled continuous pulsing of the lime-sand in the reactor with one minute of up-flow followed by one minute of settling. Evaluations consisted of filling the tank with varying amounts of lime-sand (360–450 kg) and varying pump flows (from 1.5 to 3.3 L/s during pulsing). Pulsing flow at 1.6 L/s at two minute intervals was also trialled with an extended settling time of 15 minutes every hour to allow complete settling of the sand bed.

Settling tanks: Two 3.13 m x 1.33 m rectangular tanks (1.2 m deep, with 5 baffled sections in each) were connected in series to facilitate settling and degassing of water before discharge back to the evaporation basin. The nominal water volume held in these tanks during operation varied from 3900 L (for 0.5 L/s pumping rate) to 4188 L (for 3.5 L/s).

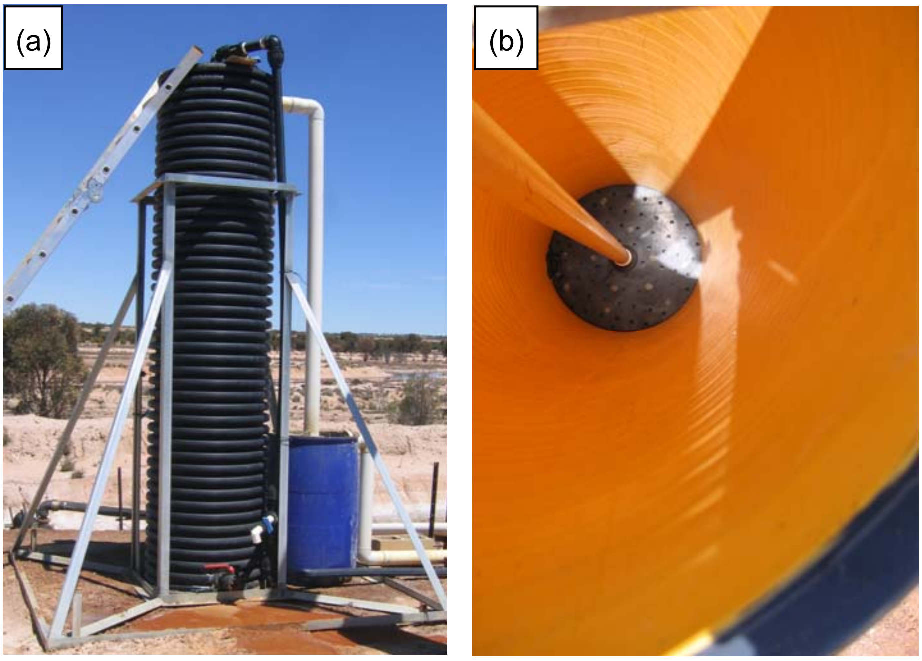


Figure 43 The up-flow tank reactor (a) after assembly of the column and the support tower (with the 50 mm diameter polythene inflow through the top and outflow via the 90 mm PVC)and (b) a view of the diffuser plate inside the reactor after installation

iii. Monitoring: Flow into the tank reactors was continuously monitored using a helical vane flow meter (50 mm, Elster H4000 Woltman Helix helical vane meter, Elster Metering Pty Ltd.) and logged with Unidata loggers. pH and EC on inflow water and pH on outflow water (in the settling tanks) were also continuously logged. Grab samples of inflow and outflow water were taken for on-site analysis of pH, EC, temperature, redox potential (using a Ag/AgCl probe) and laboratory analysis of acidity, alkalinity, all major ions, metals and a range of minor elements (see Appendix A for details). Net acidity was calculated from metal concentrations, pH and alkalinity (see Section 3.2.1 & Appendix B). Samples of lime-sand left in the reactors after treatment trials were also taken for analysis of accumulated iron and aluminium minerals. The water used for all trials averaged 69 800–79 700 mg TDS/L with pH of 2.7–3.2, acidities of 0.5–0.95 g $CaCO_3$/L (highest in the pulsed up-flow trial with settling) and major metals consisting of 11–24 mg iron/L and 82–139 mg aluminium/L.

iv. Performance results: The treatment efficiency of the percolation tank reactor in relation to the volume pumped through the system was limited and rapidly fell to less than 0.15 g $CaCO_3$/L for water with an initial inflow acidity of 0.51–0.55 g $CaCO_3$/L despite containing more than 1.5 t of lime-sand (Fig. 44). There was no improvement in treatment even when flows were reduced to 0.3 L/s though effectiveness appeared to be reaching a steady state at around 0.05 g $CaCO_3$/L with extended treatment times (i.e. more than several weeks). At low efficiencies, iron and aluminium concentrations were reduced slightly with no effect on minor metal concentrations.

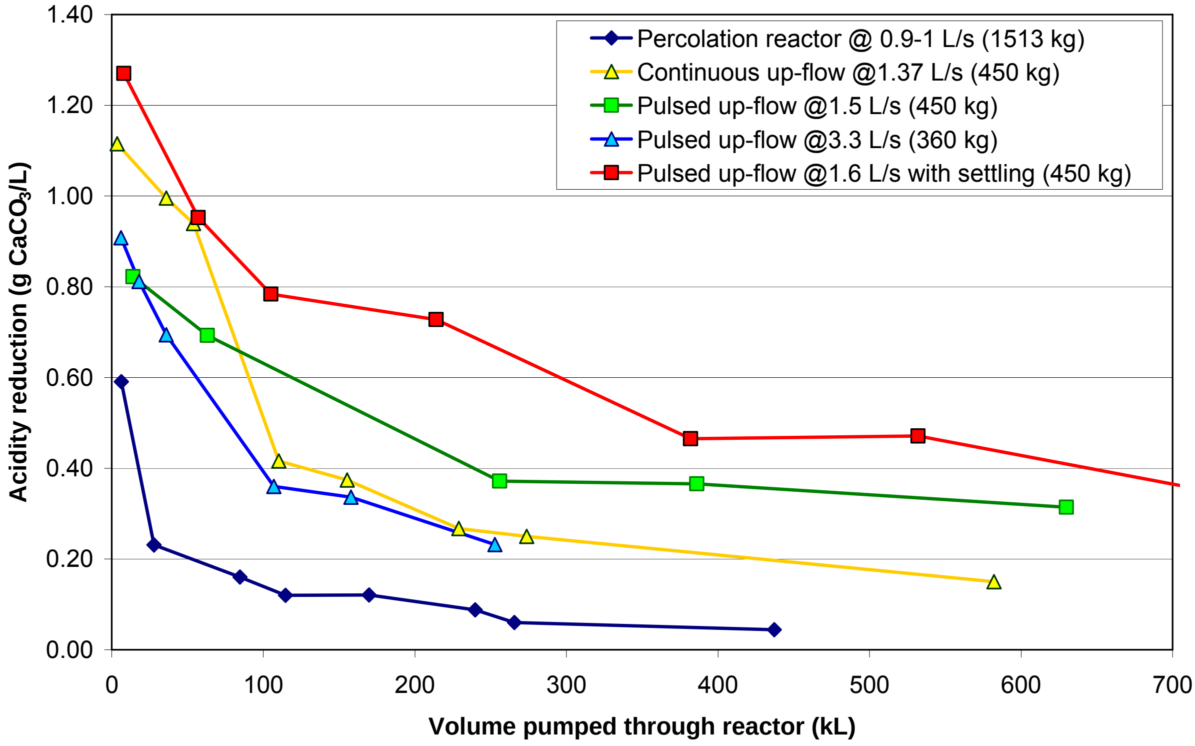


Figure 44 Efficiency of acidity treatment (reduction of outflow acidity compared with inflow acidity) normalised to volumes of water pumped through different configurations of lime-sand tank reactors

Iron crusts forming on each lime-sand layer in the percolation reactor were the main factor limiting treatment efficiency. The crusts were evident within hours of operation starting and on the top percolation table developed to the extent of completely preventing infiltration of acidic water (Fig. 45).

The reduced infiltration caused water to flow unevenly through the sand beds on each table as well as some flow to spill over the edges of the tables, thus bypassing some tables. Each factor would also have reduced treatment effectiveness by reducing the contact time between the lime-sand and water flowing through the reactor. This was estimated to be a maximum of 37 minutes given pumping at 0.5L/s through 4 layers of lime-sand of 150 mm thickness (ignoring the effects of sand in the base) with 40% porosity.

Continuous pumping of water as up-flow through lime-sand significantly extends the effectiveness of the lime-sand (Fig. 44). Full acidity treatment was achieved for water with 0.85 g $CaCO_3$/L during the first 60 kL of pumping (after about 12 hours pumping), but steadily decreased, reaching less than 0.4 g $CaCO_3$/L after treating 110 kL of water (at around 23 hours pumping). After pumping 580 kL, neutralisation declined to less than 0.15 g $CaCO_3$/L but appeared to be steadying (Fig. 44). Inflow acidity during the trial varied from 0.75 to 0.87 g $CaCO_3$/L for water of pH 2.8–3.0. The decrease in neutralisation effectiveness was attributed to iron coatings forming on individual lime-sand grains (Fig. 46); essentially a micro-scale version of the crust that had formed in the percolation reactor. This occurred despite water remaining in the reactor for less than 6 minutes.

Pulsing the water flow upwards through lime-sand achieved significantly greater effectiveness of neutralisation than continuous flow. Treatment rates were sustained above 0.3 g $CaCO_3$/L (50% of inflow acidity) for the duration of a trial involving pumping 630 kL over 11 days (shown by green line and squares in Fig. 44) and were consistently higher than treatment with continuous flow (yellow line and triangles in Fig. 44). Slow deterioration of treatment rates was observed, although the decline in effectiveness was less than in the continuous flow system (Fig. 44). Inflow acidity during the trial varied from 0.58 to 0.62 g $CaCO_3$/L for water of pH 2.9–3.0. Higher treatment rates were achieved using an extended settling time (red line and squares in Fig. 44) and sustained at above 0.25 g $CaCO_3$/L for more than 3 weeks during which 1000 kL was pumped (data not shown in Fig. 44). It was likely that higher neutralisation rates could have been maintained, particularly since the decrease observed after pumping 214 kL corresponded with more than half of the lime-sand initially added to the reactor having been consumed.

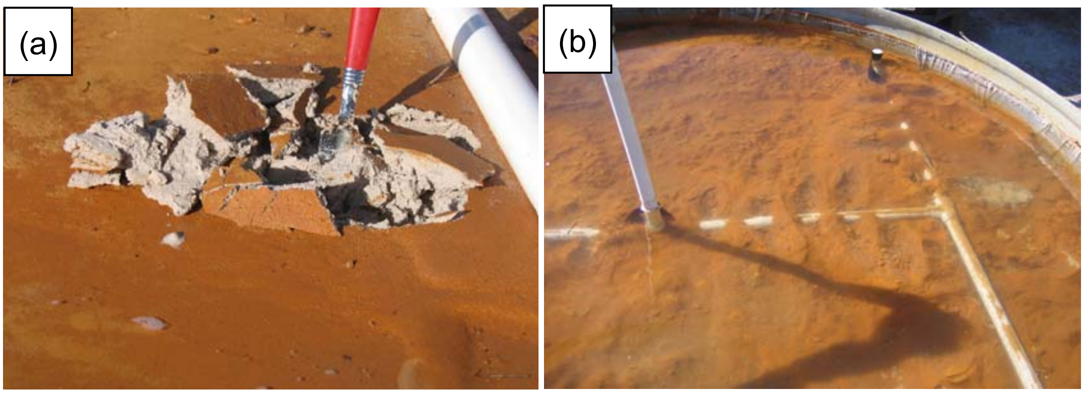


Figure 45 Iron crusting (a) formed after several days on the surface of the lime-sand and (b) gradually reduced the infiltration of water through the sand and caused water to pond to 10 cm depth

The decline in treatment effectiveness with the pulsed up-flows coincided with gradual armouring of the lime-sand by iron oxides, despite the expectation that pulsed flow through the sand should minimise, if not prevent, this. It was likely that expansion of the sand bed within 24 hours of pulsed up-flow reduced the abrasive collisions between sand grains during pulses and so enabled the armouring. Observations during the trials indicated that there was a gradual rise in the height to which the lime-sand was suspended in the tank until reaching a height of 2.7 m from the base of the reactor after three days (this was the fluidisation level). Fluidisation to this level meant that the lime-sand was effectively ‘diluted’ in 546 L of water (a ratio of approximately 1 L solids: 2.8 L water using initial sand loading rates), with the sand ‘concentration’ increasing slightly during the non-pulse time as the fluid level falls. This ‘dilution’ would have limited the abrasive action at the beginning of each pulse and was therefore less effective in preventing iron armouring on the surface of lime-sand grains.

Pulse duration (1 minute on and 1 minute settling time) and flow rate (1.5 L/s) conditions were set to ensure that sand rise (no more than 0.34 m from settled bed sand) during pumping would be counteracted by sand settling back to starting levels in the non-pulse time. The maximum fluidisation level in the reactor loaded with 450 kg lime-sand was expected to have been 1.54 m. The gradual ‘creep’ of the fluidisation level to 2.7 m might have occurred because pulse flow rates were too high or because the settling rate of sand slowly decreased as treatment progressed. The latter cause cannot be addressed by design other than to use larger grained lime-sand or programming flow shut-downs at regular intervals to allow complete settling. Subsequent trialling of one such pulsed flow regime consisting of two minutes pumping with two minutes settling time and 15 minutes of flow shut-down every hour (allowing extended settling time) indicated that this could sustain treatment rates for much longer (shown by red line and squares in Fig. 44).

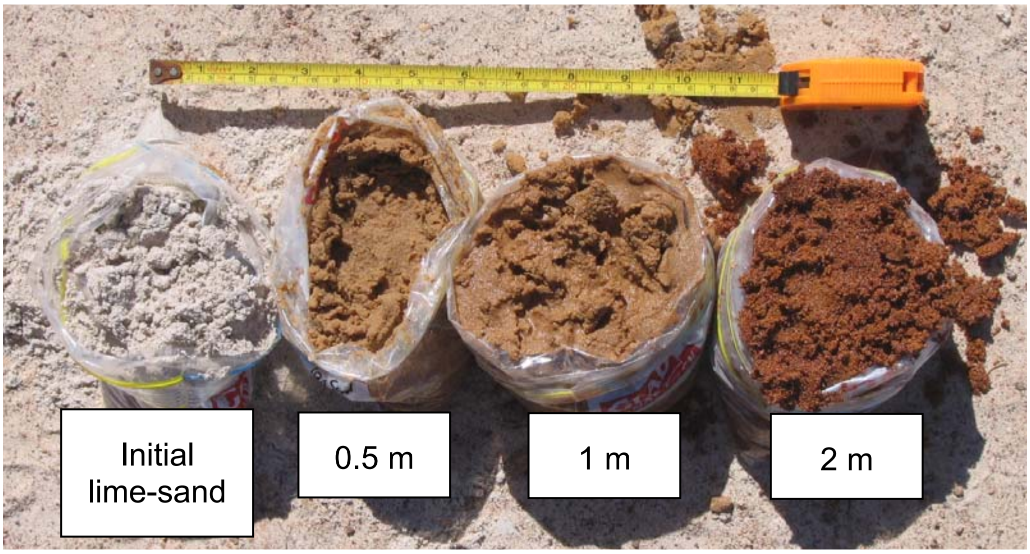


Figure 46 Iron armouring of lime-sand sampled at a range of depths (below outflow) from a pulsed up-flow reactor, evident as red staining after neutralising 580 kL acidic water over 8 days

The trials indicated that high flow rates during pulsing can almost off-set any benefits of pulsing on effectiveness. Effectiveness during up-flow pulsing at 3.3 L/s was not significantly better than that achieved with a continuous up-flow reactor (Fig. 44). This diminished effectiveness might have been due to the initial loss of sand from the reactor observed during the first few hours and the lower weight of sand initially loaded into the reactor pulsed at the higher flows. Pulsing at 3.3 L/s resulted in sand almost immediately suspended in the full volume in the reactor (i.e. initially 360 kg lime-sand with 254 L solids volume in 492 L water resulting in an initial ratio solids:water of 1:2). This was evidently too dilute for pulsing to prevent sand armouring compared with no pulsing.

The reduction in acidity associated with a single pass through the pulsed up-flow reactor (at 1.5 L/s) resulted in more than 40% decrease in soluble aluminium concentrations and more than 20% of soluble iron when acidity treatment rates were greater than 50%. While pH exceeded 5.5, the reductions in iron concentrations were generally smaller, rarely exceeding 50% while reductions in aluminium concentration could exceed 90%. None of the trials reduced manganese concentrations.

For all trials with lime-sand reactors, the extent to which treatment lowered minor element concentrations largely depended on the pH of outflow waters and whether solids settled in the settling tanks. The concentrations of some dissolved minor elements such as lead, uranium and rare earths such as cerium and lanthanum were commonly reduced by more than 50% when acidity treatment rates were high (and pH was generally above 4.5). At best, lead, cerium and lanthanum could be reduced by up to 98% during the first hour of treatment when outflow water pH was between 6 and 7. Other minor elements behaved less predictably, with concentrations of metals such as nickel, copper, zinc and cobalt generally not reduced at all and others such as chromium not generally reduced by more than 35% even during high rates of acidity treatment.

Minor metals and other elements are generally removed by co-precipitating with sludges formed after water is neutralised. The rate of sludge formation was limited in the settling tanks which retained water for a maximum of only 90 minutes. Hardly any iron-dominated sludge was observed to form (and little iron was removed from the water during treatment) which probably further limited removal of most minor metals and other elements. If a soil-lined holding/settling basin was used, the iron and aluminium minerals present are likely to contribute to adsorption of minor elements provided that pH exceeds 6.

v. Design implications: This case study highlights the importance of maintaining abrasive mixing of lime-sand to minimise the formation of iron coating on sand grains and thereby maintain high treatment rates. The trials showed that it was possible to sustain treatment rates of at least 0.3 g $CaCO_3$/L with a single passage of water through a lime-sand reactor over several weeks for waters of 0.58–0.62 g $CaCO_3$/L. Optimisation of flow rates is critical to ensuring that lime-sand does not remain fully fluidised with the duration of up-flow pulsing and intervening settling time likely to be important in reducing lime-sand armouring. While rates of minor metal removal were not high, these can potentially be optimised when the reactor is used in conjunction with larger, earth-lined oxidation and settling basins.

These results also indicate that lime-sand reactors provide some of the lowest cost acidity treatment. For water with an initial acidity of 0.6 g $CaCO_3$/L, which is the average acidity of Wheatbelt drain waters (see Section 3.2), neutralisation costs are expected to be as little as 10 cents per kL in situations where electric pumping is possible. This estimate is based on:

- Neutralisation of at least 0.3 g $CaCO_3$/L of the acidity for each pass through a pulsed up-flow reactor (at 1.5 L/s during pulsing), therefore requiring water to be pumped through a maximum three times to achieve complete neutralisation
- Lime-sand costs estimated to be 3.2 cents per kL based on treatment of 0.6 g $CaCO_3$/L of water and lime-sand (90% $CaCO_3$) cost and delivery of $50 per tonne
- Pumping costs estimated to be 6.8 cents per kL assuming water is pumped through three times and a cost of 2.3 cents per kL pumped based on a 0.7 kW pump (as used in the case study) costing 12.3 cents per hour in electricity costs to run (assuming electricity cost of 17.61 cents per kWh) and pumping 5400 L per hour
- A reactor configuration consisting of a pump switching between several reactors (allowing continuous pumping, but intermittent reactor pulsing)

These cost estimates do not include time involved in checking that the system is working, periodic flushing of the tanks and removal of sludges from settling ponds. The costs will also be greater if the efficiency of lime-sand use is less than 100%. For example, if only half of the $CaCO_3$ in the lime-sand could be used effectively in the tank reactor, this would result in a doubling of the lime-sand cost to 6.8 cents per kL in the above example contributing to a treatment cost of 13.6 cents per kL (excluding capital depreciation and maintenance).

Take home message: It is possible to use cheap sources of lime-sand in a simple up-flow reactor to neutralise acidity in drain waters containing dissolved iron. The trials provide sufficient evidence of stable neutralisation rates being achieved to support further development of the treatment system to neutralise large volumes of water quickly.

7.4 Hydrated lime dosing

7.4.1 Principle

This technique involves the treatment of water using fast-reacting hydrated lime (chemical name: calcium hydroxide $Ca(OH)_2$) through pumping and controlled dosing to achieve target pH levels within a drain, prior to point of discharge from a drain, prior to discharge to a containment basin or within a holding cell/receiving environment.

7.4.2 Design and construction considerations

i. Power supply: This treatment requires a continuous electrical power supply to run the pump and dosing unit. Current locally available treatment units generally require a generator (up to 6 kVA) or single phase power although these could perhaps be configured for operation using solar power. The costs of power supply for remote operation of temporarily deployed treatment units can add significantly to treatment costs but this might be reduced by solar power supply or auto-start control systems for generators.

ii. Treatment units: A range of portable, self-contained treatment units is available for hire locally and interstate. These include the Neutra-Mill, Calibrated Reagent Applicating Blender (CRAB) and Hydro-Active Limestone Treatment (HALT) systems (Taylor et al. 2005). All systems involve a unit for storing dry reagent (as fine powder) or slurry, a pH monitoring and control unit and metered dosing system (Fig. 47). Use of slurries of hydrated lime supplied as a 20% or 30–40% suspension in water may ensure more consistent dosing and operation with limited maintenance. Slurries can also be significantly safer to handle than dry powders. Treatment units can be sized according to the acidity load being treated (as discussed below in treatment approaches). To ensure effective operation all units require supervision of monitoring systems and regular (weekly) servicing which can be arranged under contract to the companies from which the units are hired. Hydrated lime can quickly raise the pH to 12 if overdosing occurs. High alkalinity in drain water poses risks to aquatic environments that are somewhat similar to those posed by acidic drain water. For example, at higher pH, dissolved aluminium concentrations can rise due to the formation of aluminium hydroxide species.

iii. Treatment approaches: Dosing units can be applied to continuous or batch treatment of acidic water. *Continuous treatment* of discharged water (e.g. prior to discharge to creeks, floodways or lakes) requires pumping and treating all water through the dosing units. Dosing all the water achieves complete, reliable treatment. Pumping and treating a portion of the discharge by hyper-dosing at a rate that offsets acidity in the remaining, unpumped portion of the flows risks over- or under-dosing.

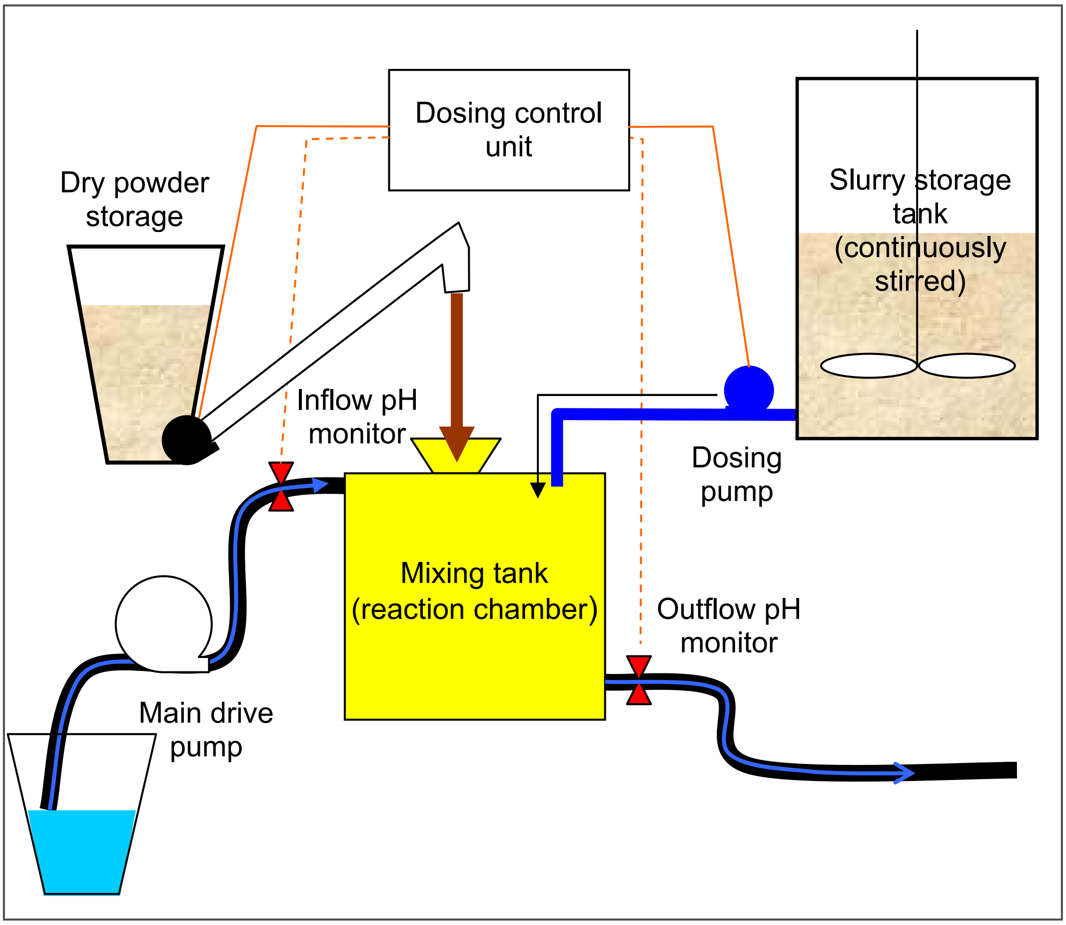


Figure 47 Elements of a hydrated lime dosing unit (with slurry dosing or dry power addition)

Water within evaporation basins, detention basins or lakes can be *batch treated* by direct dosing of the waterbody at a rate dictated by the expected acidity and volume held. This requires continuous monitoring of pH to avoid over-dosing. Batch treatment is commonly used for treating acidic water in old open-cut mine pits (see, for example, Jones et al. 2003) providing significant gains in the efficiency of power and capital use since treatment costs generally decrease if large volumes are treated quickly. Batch treatment is often suited to using larger capacity treatment units dosed at rates of tonnes of reagent per day. Critical to the success of this approach is the settling and retention of precipitates in the waterbody before release or further use of the water.

iv. Settling tanks/ponds: Precipitates from the neutralisation process (mostly iron- and aluminium-rich sludges) must be removed from neutralised waters by settling prior to discharge to creeks or use for desalination or aquaculture (for example). As discussed previously, these sludges may trap minor metals and other elements after neutralisation; hence managing these is an important part of the treatment process (see Section 3.1).

The design of settling tanks and ponds is discussed in previous sections (see Section 7.2.2 ix. Outflow oxidation and settling ponds). Where space is limited or precipitates forming in the water after neutralisation do not settle well, it may be necessary to use more complex water treatment systems, for example, clarifiers, lamellar plate thickeners, adding coagulants and flocculators (see Metcalf & Eddy Inc. et al. 2003).

7.4.3 Operational management

i. Maintenance and servicing: Regular (weekly) monitoring of treatment performance and servicing of the unit will be required. In particular, the operation of the units depends on cleaning and checking the performance of the pH probes that directly influence the rate of dosing. Deterioration of these probes can result in over- or under-treating acidic drain waters and can only be verified by on-site checks. Monitoring data from the treatment units can be sent by telemetry (transmitted automatically by mobile phone hook-up) to help monitor performance.

ii. Storage and handling of neutralising agents: Regular reloading of neutralising materials will also be required, depending on the acidity load being treated. The caustic nature of materials such hydrated lime slurries (often pH> 12) requires storage tanks and handling equipment designed to Australian Standard AS 3780 for the storage and handling of corrosive substances. The hazardous nature of hydrated lime also requires that appropriate protective safety equipment be used when handling the material (see example Materials Safety Data Sheet for Hydrated Lime – Appendix E).

iii. Power supply management: The reliability of hydrated lime dosing units depends on a continuous power supply for metering reagents and, often, continuous water pumping. Experiences with pumping systems in the eastern Wheatbelt show that power supply from the Wheatbelt grid can be erratic, particularly during summer and autumn when thunderstorms are most frequent. The reliability of the power supply must be considered when establishing and managing hydrated lime dosing units in remote locations, particularly since power failure results in failing to treat drain water. Some of these risks could perhaps be overcome by a simple telemetry system alerting an operator that the treatment unit has stopped or installing battery back-up systems.

Sludge removal and disposal: Sludge will need to be regularly cleaned out of settling tanks and ponds where these are used to trap sludges formed during neutralisation. Details on this are covered in earlier discussion (see Section 7.3.3 iii. Sludge removal and disposal).

7.4.4 Effectiveness

Short term (months): Hydrated lime dosing can be very effective in treating acidic saline drain water provided there is continuous power supply. Local trials indicate that hydrated lime dosing can completely treat acidity and remove a range of minor elements, most likely through co-precipitation with or by adsorption onto sludges in settling tanks.

Dosing units can be configured to achieve a particular pH in outflow water – which is not easy to achieve using slower reacting neutralising materials such as limestone or lime-sand. While there may be some variation in the pH control between dosing units depending on the form of reagent used (dry powder or slurry suspension), trials of a local dosing unit indicate that pH variation is minimised by using pre-prepared slurries of hydrated lime. The target pH for treatment can be controlled and tailored to remove specific minor metals as required.

Although minor elements can be efficiently removed, the continued effectiveness of this process depends on whether precipitates are retained in settling tanks/ponds. Loss of fine iron- and aluminium-dominated sludges from settling tanks/ponds to creeks increases the risk of minor metals being released into waterways should re-acidification, changes in salinity or anaerobic conditions occur. This can potentially be problematic even though the sludges might be benign with regards to minor metal concentrations.

Long-term (>6 months): Hydrated lime dosing units are likely to remain very effective over the long term, with limitations related to the regularity of maintenance, servicing, response to and frequency of break downs or power outages. The potential disruption of power supplies in many parts of the Wheatbelt is one of the greatest threats. Anecdotes indicate that power supply outside towns can regularly be disrupted during summer and autumn as a result of thunderstorm activity. This risk could perhaps be lowered by using holding basins with enough capacity to store untreated water for at least a day in case power to a dosing unit fails.

7.4.5 Case study: Wallatin Creek hydrated lime dosing (Water Treatment Systems Australia)

i. Site: The treatment site, situated on Andrew and Ian McNeil's property, Wallatin Rd, Doodlakine, was the discharge point of a valley floor drainage project undertaken as part of the Wallatin-O'Brien Catchment Demonstration Initiative (see location map in Appendix D).

ii. Construction and monitoring: A hydrated lime dosing unit was installed and evaluated on treating all discharge water flowing between April and September from 9.6 km of leveed deep drains in the Wallatin Creek catchment.

Dosing plant: Water Treatment Systems Australia (a subsidiary of Lime Industries) was contracted to supply, install and trial an 'off-the-shelf' hydrated lime dosing unit. This was installed to treat acidity in all water discharged from the leveed drain before discharge to the creek. Locating the treatment unit at the end of the drain 20 m downstream of the leveed section, where the drain was shallowest, reduced the energy needed to lift the water, provided access to the unit (plus service vehicles) and met the needs of treating all water before discharge. The plant was located beyond where flooding from the adjacent creek would be expected.

The dosing unit was a self-contained pH-controlled system through which all drain water was pumped and dosed from a 1000 L tank containing 40% hydrated lime slurry (Fig. 48). The slurry was periodically reloaded into the tank from pre-prepared 25 kg bags (supplied by

Lime Industries, see Appendix E) designed for ease of handling and minimising direct contract with the slurry. Continuous power was required for the monitoring systems, the pump and to maintain mixing of the slurry. The system was powered by a portable diesel-fuelled generator (6 kVa) for this trial, although the unit can readily be powered using mains electricity.

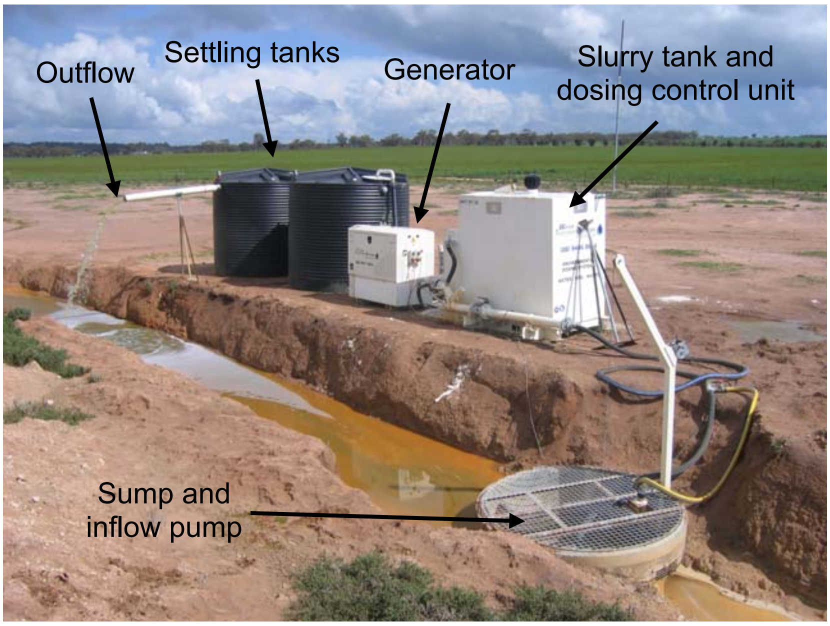


Figure 48 The hydrated lime dosing unit treating water at the end of the Wallatin Creek drain

A sump was constructed in the drain to provide a point from which water could be pumped using a submersible pump controlled by a level switch. The sump consisted of a 1.5 m diameter concrete pipe with inflow level set at the drain base and the outflow level set about 100 mm higher, providing a high-flow bypass option if the pumping and treatment systems failed. A steel grill placed over the inflow reduced the amount of organic debris that could flush into the sump and was complemented by a larger trash retention barrier protecting the Department of Water flow gauging site 700 m upstream. For this trial, the probe monitoring the inflow pH was in the sump and the probe monitoring outflow was placed at the entry to the settling tanks.

Settling tanks: Sludges formed after neutralising water were captured by settling in two 4500 L tanks connected in series. Inflow to the first tank was confined to a 400 mm diameter inflow forcing water to flow to the base of the tank before rising and exiting via a 150 mm diameter outflow into the second settling tank and outflow to the drain (Fig. 48).

Monitoring: The pH of inflow and outflow water and the total volume pumped were monitored continuously and recorded on data loggers in the dosing unit. The unit was equipped with an electromagnetic flow meter. Water flow, pH, EC and temperature were also measured and logged continuously at the Wallatin Road culvert crossing upstream of the dosing unit. Flow measurement involved gauging water levels flowing over a 30 degree V-notch plastic weir secured to the inlet of the 600 mm culvert. Water levels were measured to the nearest millimetre using a Mindata water level measurement system. Plastic mesh cages were constructed around the weir inlet and upstream of this to protect the site from blockage by the large amounts of organic debris that blew into and washed down the drain.

Grab samples of inflow and outflow water were also taken for on-site analysis of pH, EC, temperature, redox potential (using a Ag/AgCl probe) and laboratory analysis of acidity, alkalinity, all major ions, metals and a range of minor elements (see Appendix A for details).

iii. Performance results: The dosing unit achieved consistently effective treatment of acidic drain water ranging in acidity 0.07–0.43 g $CaCO_3$/L and in dissolved iron 14–72 mg/L (Table 9). The outflow pH was consistently above 7, although the water contained little additional alkalinity, which did not exceed 30 mg equivalent $CaCO_3$/L. Inflow pH during the trial period was 4.4–5.5.

Average flows during the treatment trial increased from 0.7 L/s in mid-autumn to 2.3 L/s in winter and early spring (Table 8) probably due to increased flows from rainfall and lower evaporation rates. The increased average flows corresponded to reduced average dosing rates from an equivalent 0.63–0.36 g $CaCO_3$/L (Table 8). Lower average dosing rates during winter–spring reflected the higher volumes of drain discharge with lower acidities, particularly during and immediately after rainfall events. Samples from four high flow events indicated that acidity can fall to less than 0.07 g $CaCO_3$/L during these events.

Table 8 Summary of Wallatin Creek trial monthly averages of treated flows, hydrated lime dosing rates and main operating costs

Month	Average flow rate (L/s)	Average dosing rate (equivalent g $CaCO_3$ /L)	Range of measured acidity[1] (equivalent g $CaCO_3$ /L)	Bagged reagent cost[2] (cents/ kL)	Bulk reagent cost[2] (cents/ kL)	Diesel cost (cents/ kL)
April[3] 0.7		0.63	0.05–0.30	80	39	50
May 0.8		0.46	0.31–0.43	59	29	45
June 0.7		0.45	0.07–0.37	58	28	50
July 1.6		0.37	0.07	47	23	20
August 2.6		0.36	0.35–0.44	45	22	14
September 2.3		0.36	0.38–0.42	45	22	13

[1] Analysis of total acidity from 1 to 3 grab samples, mostly during baseflow. Note only one sample taken in July.

[2] Reagent costs do not include freight to site. This was expected to increase per kL treatment price by 7% for bagged reagent and 14% for bulk reagent at a nominal price of $40 per tonne slurry. Bagged reagent costs were based on $516 per tonne of 40% slurry as 25 kg bags and bulk reagent costs were based on $250 per tonne of 40% slurry as 1000 L pallecons.

[3] Treatment from 16–30 April 2008

Table 9 Inlet and outflow concentrations of major metals and minor elements in the Wallatin Creek trial

Grouping	Element	Inlet concentration range[2]	Settling tank outlet concentration range [2]
		(mg/L)	
Major metals[2]	Aluminium	2–34	0.05–1.3
	Iron	23–72	0.3–0.8
	Manganese	5–8	1.5–5
		(µg/L)	
Minor elements[2]	Arsenic <3	*–33	<3*–12 (mostly below detection)
	Cadmium <0.3	*–21 <3	*
	Cerium 242–	842	3–49
	Copper 40–5	2	<10*
	Lanthanum 126–	634	1–29
	Lead 4–30		<2*
	Nickel 23–4	7	2–15
	Uranium 3–42		<3*
	Zinc 10–4	0	<10*

[1] Range based on grab samples of drain waters and outflow from dosing unit settling tanks over the 2½ month period (midApril–June).

[2] Concentrations of minor elements in µg/L (parts per billion), except for concentrations of aluminium, iron and manganese which are in units of mg/L (parts per million).

* Below analytical measurement limits. Note that the measurement limits vary; with reduced capacity to detect low element concentrations in more saline waters.

Note: Selenium concentrations were consistently below analytical detection limits (0.5 µg/L).

Although complete treatment during low flows was achieved, there were some difficulties in treating all flow after rainfall when some acidity was still present in drain water. For example, a rainfall event in May 2008 caused drain flows that exceeded the submersible pump capacity by at least two-fold and resulted in untreated overflow from the sump (although a sample after the peak flows contained less than 0.15 g $CaCO_3$/L). This probably occurred after other larger flow events as well. Although the maximum pumping rate was set to 15 L/s (the unit has a maximum pumping and dosing rate of 90 L/s), drain flows varied from less than 1 L/s to more than 30 L/s in less than a day, often after rain.

The concentrations of aluminium and iron (not manganese) and minor elements were reduced by more than 90% after treatment and settling (Table 9). It was not possible to be sure whether all minor elements were effectively removed because some (e.g. cadmium, selenium and nickel) started at very low concentrations. Further trials are required to ascertain whether hydrated lime dosing would have a similar benefit with drain water with higher concentrations of elements (particularly nickel).

Sludges trapped in the settling tanks were rich in iron and aluminium amorphous mineral precipitates (Fig. 49). These contained concentrations of lead (up to 155 mg/kg dry material) and nickel (up to 70 mg/kg dry material) that require further assessment of environmental risks depending on the proposed disposal or re-use (Department of Environment 2003; see also Table 7). Despite occasional arsenic detection in untreated drain water, it did not accumulate to levels of concern in the sludges. It was also notable that the minor element concentrations in the sludges broadly corresponded with what might be expected based on concentrations in untreated water (see Table 7).

Figure 49 Iron- and aluminium-rich sludge from the settling tanks of the hydrated lime dosing unit. The green is due to ferrous hydroxide which oxidises to orange iron oxide on exposure to air.

The sludges also contained additional neutralising capacity equivalent to up to 32% by weight of $CaCO_3$. Chemical analyses indicated that less than 10% of this neutralising capacity was reactive calcium carbonate which indicated that the balance was likely to be exchangeable calcium on the surfaces of the minerals formed in the sludges.

Regular de-sludging of the settling tanks is important to avoid bypass of solids, particularly with small tanks. Bypass occurred after several months of operation and resulted in some sludges flowing into the creek. This highlights a critical point in managing such treatment systems, particularly as the sludges probably contain some minor elements that may pose an environmental risk or smother aquatic organisms in creek beds (see Section 2.3). The physical characteristics of sludges often increase the risk of this occurring during low flow conditions.

iv. Design implications: The trial results indicated that hydrated lime dosing provides great certainty of treatment, but the price strongly depends on the cost of the neutralising reagent and choice of power source. Treatment costs for Wallatin Creek drain water varied from 35 to 89 cents/kL (not including freight for hydrated lime) with most of the variations due to the costs of diesel to run the generator (Table 8). Naturally, diesel generator fuel costs were lowest during months with highest average flows, highlighting that the treatment approach is perhaps best suited to drains with high flows and high acidity.

For hydrated lime treatment of a drain with average acidity of 0.6 g $CaCO_3$/L, being the median acidity in Wheatbelt drain water (see Section 3.2) and assuming average drain flows of 1 L/s, the cost was expected to be 46–134 cents/kL, assuming:

- 1.9–50 cents/kL operating costs for a 0.7 kW pump using grid electricity or a diesel fuelled generator
- 44–84 cents/kL for hydrated lime purchase and delivery: based on range from $148/t, including freight at $90/t for a 20% suspension of $Ca(OH)_2$ to $295/t, including freight at $45/t for a 40% suspension of $Ca(OH)_2$.
- Freight costs were for bulk freight to the central eastern Wheatbelt (Kellerberrin) and will be different for other locations.
- Each kilogram of acidity (as $CaCO_3$) requires 0.74 kg $Ca(OH)_2$ powder for neutralisation which is equivalent to 2.47 kg of 40% hydrated lime slurry (excluding impurities) or 4.11 kg of 20% suspension of $Ca(OH)_2$.
- 40% and 20% $Ca(OH)_2$ preparations are prepared as a weight made up to volume basis and not a proportion by weight basis (e.g. the 40% slurry contains approximately 300 kg $Ca(OH)_2$ in every tonne and not 400 kg in every tonne of slurry).

These operating costs do not include capital depreciation on the unit, time for reloading the hydrated lime, servicing and removing sludges from the settling ponds.

Ways of reducing costs associated with pumping and dosing costs will be most beneficial where a diesel generator is used; for example, intermittent treatment of water in large batches (megalitres at a time), using holding ponds for drain discharge between treatments with intermittent release during treatment. Using solar powered units may also be of benefit, although these require sufficient battery storage to power the unit during nights and cloudy days.

Take home message: Hydrated lime dosing of drain water is a treatment approach with a high certainty of success. The price of achieving treatment is dependent on the cost of the neutralising reagent and choice of power source.

8 Integration

This section covers principles for using combinations of treatment options to achieve greater confidence that acidic drain water will be treated in the short term and, more importantly, in the long term.

All individual treatment options have some risk that water may not be treated. This risk may be highest during certain flows or be due to relying on pumps and/or power systems with inherent risks of failure or due to factors that damage or erode treatment performance over time (e.g. blockages, iron oxide armouring of lime-sand or depletion of decomposable organic matter in composting systems, deterioration of pumps and valves). While these risks can to some extent be overcome by design or back-up systems, a more robust approach to lower risk is to use combinations of treatments distributed throughout a drainage system and/or at the end of a drain. These combination systems are likely to achieve greater stability and assurance of treatment than multiple installations of single option treatments.

Integrating treatment systems by coupling multiple treatment units within and/or at the end of a drain is similar to the treatment train principle used in managing water quality in urban drainage systems (Department of Environment and Swan River Trust 2006; Department of Water and Swan River Trust 2007). In stormwater drains, this involves using several different units to treat water in sequence or concurrently, working at various scales from source to discharge point to provide multiple barriers to protect water quality. Thus, contaminants are treated throughout the drainage system using a range of treatment units, each providing some incremental gain in water quality.

The level and consistency of acidity treatment required, and therefore the level to which treatment systems need to perform, are constrained by the water quality acceptable to the environment receiving the treated water (see Section 3.2.3). For sensitive environments, the objective might be to minimise all impacts associated with drainage acidity and therefore treat all water. This approach demands methods with a high guarantee of success for all undesirable aspects of acidic water during all types of flows (i.e. high flows after large rainfall events as well as mid-summer low flows). In other situations, the level of treatment may be less demanding with, for example, occasional deviations acceptable during high flow events, but consistent treatment of all low flows. Most single treatment options cannot cover all requirements, hence using several approaches in sequence or concurrently builds the level of confidence that treatment targets will be achieved.

A range of compatible treatment options for treating acidic drain water in and at the end of drains is suggested in Tables 10, 11 and 12.

Table 10 Potential combinations of in-drain options for acidic water treatment

Potential IN-DRAIN combinations	Comments
Subsoil carbonate/lime-sand beds with strategic tank reactors and/or diversion wells + carbonate-rich subsoils lining berms and spoil banks + in-drain composting beds in the lower reaches of the drain followed by oxidation pond	Bulk of acidity handled by carbonate/lime-sand beds and opportunistic lime-sand reactors (e.g. at road crossings). In-drain composting beds provide final polishing treatment to remove minor metals (plus residual acidity treatment). Retention of iron and aluminium sludges in final oxidation/settling pond (in tail channel). Carbonate-rich subsoils on berms will feed additional alkalinity to neutralise acidity mobilised in high flows.
In-drain composting beds coupled with lime-sand/subsoil carbonate beds + carbonate-rich subsoils lining berms and spoil banks	Partial acidity reduction using in-drain composting beds augmented by lime-sand/subsoil carbonate beds. Possible increased efficiency of carbonate use when incorporated throughout composting mix and drain sections downstream of in-drain composting beds. Carbonate-rich subsoils on berms will feed additional alkalinity to neutralise acidity mobilised in high flows.

Table 11 Potential combinations of end of drain options for acidic water treatment

Potential END of DRAIN combinations	Comments
Oxidation pond + lime-sand basin + composting wetland + oxidation/settling pond	Bulk acidity load treated by lime-sand basin with minor metal removal focused in small composting wetland (handling smaller acidity load).
Lime-sand tank reactor + composting wetland + oxidation/settling pond	Bulk acidity load treated by a single pass through lime-sand tank reactor (assuming on-site power available for pumping) followed by minor metal removal focused in small composting wetland (handling smaller acidity load).
Lime-sand tank reactor + lime-sand basin.	Bulk acidity load treated by a single pass through lime-sand tank reactor (assuming on-site power available for pumping) followed by treatment of residual acidity and removal of minor metals in small lime-sand basin (handling smaller acidity load). Not appropriate for drain water with high concentrations of minor elements such as cadmium, nickel, selenium and zinc.

Table 12 Potential combinations of in-drain and end of drain options for acidic water treatment

Potential IN-DRAIN and END of DRAIN combinations	Comments
In-drain composting beds + oxidation pond + lime-sand basin (for high flows)	Acidity reduction in in-drain composting beds, with treatment of residual acidity and any high flow acidity in lime-sand basin. Pre-treatment and flow compensation in oxidation pond to reduce armouring of lime-sand in basin.
In-drain composting beds + composting wetland (low flows) + lime-sand basin (for high flows)	Acidity reduction in in-drain composting beds, with treatment of residual acidity in low flows using a final composting wetland. High flows bypass treatment wetland and treated in a lime-sand basin.
Lime-sand/soil carbonate beds (in-drain) + composting wetland + oxidation/settling pond	Acidity reduction in lime-sand/soil carbonate beds in drains, with treatment of residual acidity in low flows using a final composting wetland (note: in-drain neutralisation reduces the overall size of end of drain wetland). Limited treatment of high flows.
Lime-sand/soil carbonate beds (in-drain) + lime-sand tank reactor + oxidation/settling pond	Acidity reduction in lime-sand/soil carbonate beds in drains, with treatment of residual acidity using a lime-sand tank reactor (assuming on-site power available for pumping). Limited treatment of high flows. Not appropriate for drain water with high concentrations of minor elements such as cadmium, nickel, selenium and zinc.
Lime-sand/soil carbonate beds (in-drain) or in-drain composting beds + hydrated–lime dosing unit + settling pond/tanks	Acidity reduction in lime-sand/soil carbonate beds and/or in-drain composting beds, with treatment of residual acidity and removal of minor elements at the end of the drain using a hydrated lime dosing unit (assuming on-site power available for pumping). Capacity to handle high and low flows.

Particular attention is required to the arrangement of treatments when combining carbonate and composting based treatment systems. Iron armouring and iron precipitate formation can greatly reduce the performance of carbonate-based treatment systems (e.g. lime-sand/limestone beds, subsoil carbonate beds or lime-sand basins), particularly in aerated conditions with iron-rich water (>10 mg/L; Younger et al. 2002). For best results, these treatment systems should not receive water from composting wetlands or drains without prior oxidation and settling of iron. The only exception is where carbonate-based neutralising materials are mixed with composting materials or are used to bolster treatment in drain sections immediately down gradient of in-drain composting beds (where there is limited aeration of water).

Appendix A — Elements

Elements referred to in text and measured in water samples collected during pilot treatment trials (detailed in the case studies).

Major metals, elements and compounds

Name	Symbol
Aluminium	Al
Ammonia	NH_4^+
Bromide	Br^-
Boron	B
Calcium	Ca
Carbon (organic C dissolved)	C
Chloride	Cl^- c
Fluoride	F^-
Iron	Fe (includes Fe^{2+} and Fe^{3+})
Magnesium	Mg
Manganese	Mn
Nitrate/Nitrite	NOx (NO_3^- and NO_2^-)
Nitrogen (total)	N
Potassium	K
Phosphorus (total)	P
Phosphate	PO_4^{3-}
Silica	SiO_2
Sodium	Na
Sulfate	SO_4^{2-}
Sulfur	S

Minor metals and other elements

Name	Symbol
Antimony	Sb
Arsenic	As
Barium	Ba
Cadmium	Cd
Cobalt	Co
Copper	Cu
Chromium	Cr
Molybdenum	Mo
Lead	Pb
Nickel	Ni
Selenium	Se

Minor metals and other elements

Name	Symbol
Silver	Ag
Strontium	Sr
Tin	Sn
Titanium	Ti
Vanadium	V
Zinc	Zn

Rare earth/actinide/lanthanide elements

Name	Symbol
Cerium	Ce
Dysprosium	Dy
Erbium	Er
Europium	Eu
Gadolinium	Gd
Hafnium	Hf
Holmium	Ho
Lanthanum	La
Lutetium	Lu
Neodymium	Nd
Praseodymium	Pr
Samarium	Sm
Terbium	Tb
Thorium	Th
Thallium	Tl
Thulium	Tm
Uranium	U
Yttrium	Y
Ytterbium	Yb

Appendix B — Assessment of fundamental acidic water quality properties for treatment

Estimating flow

An estimate of flow is essential for calculating instantaneous acidity loads. Flow estimation basically involves measuring the cross-sectional area of flowing water and the average velocity of this flow. To achieve accurate estimates, this normally involves detailed measurements of depths and flow velocities. Nonetheless, using the same basic approach can provide order-of-magnitude estimates.

Basic flow estimation is calculated as:

$Q = V \times A$. where Q is flow in m^3/s, V is average velocity in m/s and A is cross-sectional area (m^2).

Choose a straight section of drain (or a culvert) to measure the flow with constant width, depth and shape. Avoid sections where water ponds. Measure the width and average depth of the section and calculate the cross-sectional area. Where channel shape is box shaped, cross-sectional area is calculated as: width x depth. The cross-sectional area of V-shaped channels is ½ x width x depth and for semi-circular channels is $^2/_3$ x width x depth. Culverts can be a good place to take measurements, but only if flows do not bank up within the culvert.

Velocity can be measured by timing a floating object over a set distance (measured or paced) and taking 85% of this value to estimate the subsurface velocity.

If flow measurements are to be made consistently at a site, the reliability of estimates can be improved by installing a small V-notch weir where the theoretical relationship between depth of flow and discharge can be used to estimate flow or where flow can be measured by the time to fill a bucket (Younger et al. 2002; PIRAMID Consortium 2003).

Care needs to be taken with extrapolating point measurements of flow to estimate discharge over time in drains. For example, significant under- and over-estimation can occur if using a single measurement of flow to estimate daily discharge particularly if rain has fallen.

Estimating total acidity

Total acidity can be measured on samples taken in the field using kits involving simple drop-tests or more accurate metered dosing. The basic measurement involves taking a sample with a calibrated container, adding a solution that colourimetrically indicates pH (usually phenolphthalein) and adding a dilute mixture of sodium hydroxide (of known concentration) until the colour changes (indicating that a certain end-point pH is reached). The standard method involves measuring the amount of sodium hydroxide that needs to be added to reach a stable pH in the water sample of 8.3 (indicated by a change in colour by the pH indicator from clear to purple). Commonly available test kits such as those sold by HACH (drop test kit: 4–500 mg $CaCO_3$/L Model AC-6 Catalogue No. 2223-01 or digital pipette kit for acidity and alkalinity 10–4000 mg $CaCO_3$/L Model AL-DT Catalogue No. 20637-00). Other test kits can also be obtained from local laboratory supplies including the Hanna acidity test kit (drop test kit: 0–500 mg $CaCO_3$/L Model HI 3820) and the Acid Solutions test kit.

Water samples can also be taken and submitted to a local water analysis laboratory for analysis of total acidity. If following this path it is worthwhile noting that acidity measured by field or laboratory testing can underestimate the true acidity particularly for groundwater with pH >4 containing high concentrations of dissolved iron and manganese (Kirby & Cravotta 2005). In this situation, it is recommended that samples be taken to measure dissolved iron, aluminium and manganese (i.e. filtering a sample through 0.45 µm filter paper) and acidity calculated using the equation:

$\text{Acid}_{\text{calculated}} = 1.79 \times Fe^{2+} + 2.68 \times Fe^{3+} + 5.56 \times Al + 1.82 \times Mn + 50000 \times 10^{-pH}$

(after Watzlaf et al*.* 2004)

where Fe, Al and Mn are concentrations of dissolved elements in mg/L. This calculation requires consideration of the extent to which iron is present as Fe^{2+} or Fe^{3+}. It is often easier to assume dominance of one form, for example, if pH is less than 3.5 and the water is well aerated most iron is likely to be present as Fe^{3+}, in which case 0 is used as the concentration of Fe^{2+}. For higher pH water, it may be safer to assume that dissolved iron is present as Fe^{2+} (also see notes below and Glossary).

Iron

Ferrous iron (Fe^{2+}) is the only form of soluble iron that can be readily tested in the field. Ferric iron (Fe^{3+}) can also occur, particularly in water below pH 3.5 (Langmuir 1997; Kirby et al. 1999), which can also be useful to know when using calcium carbonate based neutralisation systems (e.g. lime-sand, limestone, subsoil carbonates). Some test kits can measure ferric iron but only by difference with total dissolved iron, hence it can be difficult to detect with any confidence.

Test strips that give an instant indication of ferrous iron concentrations are available through laboratory suppliers, though the accuracy is poor. Examples:

Macherey-Nagel Quantofix 100 test strips 0–1000 mg iron/L Article No. 913 02

Merk Merkoquant Iron Test 100 test strips 3–500 mg iron/L. Product code: 1.10004.0001.

Ferrous iron can also be measured cheaply, but more accurately using field test (e.g. HACH test kit for low range 0–10 mg iron /L Model IR-18C, Catalogue No. 26672-00 or HANNA Instruments 0–10 mg/L Kit number HI 3820).

Ferric iron (Fe^{3+}) is difficult to measure in the field and is most reliably estimated by submitting a sample to a laboratory for analysis of total iron (after measuring ferrous iron in the field).

Appendix C — Trigger levels for pH and selected elements

ANZECC and AMCANZ 2000 investigation trigger levels for pH and selected elements (in µg/L[1]) common in acidic Wheatbelt drain water for 99% and 80% levels of protection to aquatic species in fresh (<2500 mg TDS/L) and marine waters. Note: that there is no ANZECC limit for acidity or any derived limits for inland saline waterways or lakes.

Element	Concentration (µg/L[1])			
	Fresh water ecosystems		Marine ecosystems	
	95% protection	80% protection	95% protection	80% protection
pH	6.5	6.5	8.0	8.0
Aluminium (Al)[2]	55	150	ID	ID
Arsenic (As III)[3]	24	360	ID	ID
Arsenic (As V)[3]	13	140	ID	ID
Cadmium (Cd)[4]	0.2	0.8	5.5	36
Chromium (Cr III)[4]	ID	ID	27.4	91
Chromium (Cr VI)	1	40	4.4	85
Copper (Cu)[4]	1.4	2.5	1.3	8
Iron (Fe)	ID	ID	ID	ID
Lead (Pb)[4]	3.4	9.4	4.4	12
Manganese (Mn)	1900	3600	ID	ID
Nickel (Ni)[4]	11	17	70	560
Selenium (Se total)	11	34	ID	ID
Uranium (U)	ID	ID	ID	ID
Zinc (Zn)[4]	8.0	31	15	43

[1] Concentrations equivalent to ppb. Divide values by 1000 to calculate trigger limit as mg/L concentration.

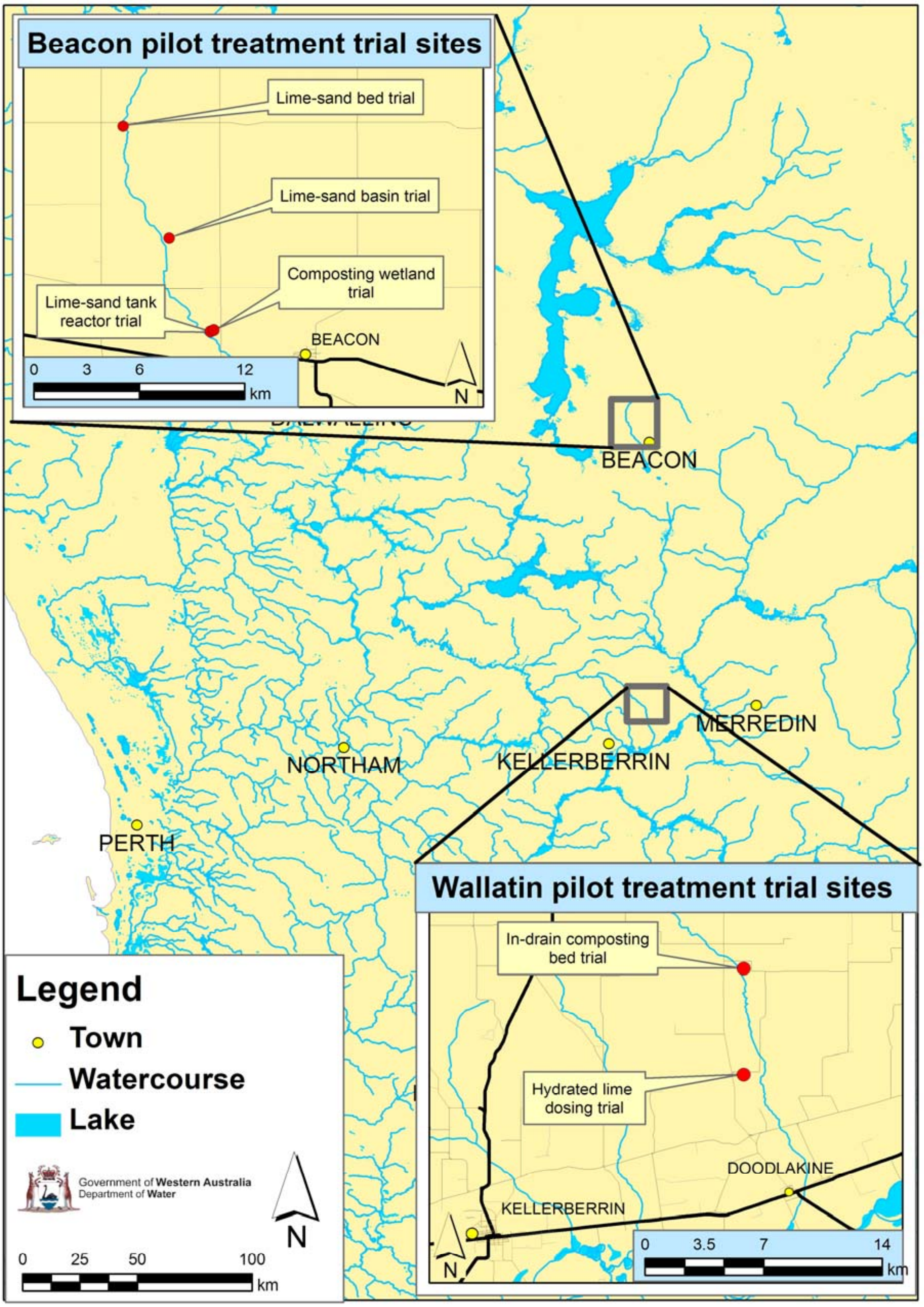
[2] Trigger limit for aluminium only applies to water with pH>6.5. None derived for water with pH<6.5.

[3] Arsenic III (as arsenite) generally occurs under reducing conditions, whereas Arsenic V (as arsenite) generally occurs under oxidising conditions.

[4] Guideline limit is sensitive to the hardness of water in freshwater, where increasing hardness reduces the toxicity of the elements (i.e. increases the trigger limit). Hardness depends on the concentrations of calcium and magnesium in water.

ID: Insufficient data to be established or not available

Appendix D — Map of case-study locations



Appendix E — Material safety data sheet for hydrated lime (slurry)

MATERIAL SAFETY DATA SHEET

40% Hydrated Lime (Slurry)

Date of Issue 01/11/02

STATEMENT OF HAZARDOUS NATURE

Hazardous according to the criteria of Worksafe Australia

COMPANY DETAILS

Company: Lime Industries Pty Ltd

Address: 43 Hector Street, Osborne Park WA 6017

Telephone Number: (08) 9446 8644

Emergency Telephone Number: 131 126 (National Poisons Information Centre)

IDENTIFICATION

Product Name: **ENVIRONMENTAL LIME**

Other Names: Lime Putty, Hydrated Lime slurry

Manufacturer's Product Code - Low Silica

UN Number: Not Applicable

Dangerous Goods Class: Not applicable

Hazchem Code: Unclassified

Poisons Schedule Number: N/A

Use: Used in various applications for pH adjustment

PHYSICAL PROPERTIES

Product description: Environmental Lime is an off-white slurry produced from Quicklime which results in a suspension of Calcium hydroxide in water.

Viscosity: At rest, environmental lime is very viscous; however, because it is thixotropic it can be easily pumped.

pH: 12

Percentage solids: 38% +/-2

Boiling Point/Melting Point: Water boils off as steam at 100 ºC. When all water is gone the resultant lime powder boils at 2850 ºC.

Vapour Pressure: Not applicable

Flashpoint: Not	applicable
Flammability Limits:	Slurry is Non-combustible
Other properties:	Not explosive. Environmental Lime has a slightly earthy odour. Reacts violently with acid.

INGREDIENTS

Chemical Composition

COMPONENTS	CAS No:	PROPORTION
Calcium Hydroxide:	1305-62-0	30-45%
Magnesium Hydroxide:	1309-42-8	0-5%
Silicon Dioxide:	14808-60-7	0-1%
Calcium Carbonate:	471-34-1	0-15%
Aluminium Oxide:	1344-28-1	0.1-1%
Water	7732-18-5	to 100%

HEALTH HAZARD INFORMATION

No specific data is available for the product for chronic exposure symptoms. The ingredients are not listed as carcinogenic in Worksafe's document "Exposure Standards for Atmospheric Contaminants in the Occupational Environment"(May 1995)

Health Effects

Ingestion:	Environmental Lime has a caustic reaction therefore will burn the mouth and throat if swallowed.
Eyes:	Corrosive and may cause severe burning of the eyes.
Skin: Irritating.	
Inhaled:	Due to the product form, inhalation hazard is low.

First Aid

Ingestion:	Do not induce vomiting. Wash mouth and lips with copious amounts of water, and give limited amounts of water or milk to drink. Seek urgent medical attention.
Skin:	Quickly, but gently wipe material off skin. Immediately remove all contaminated clothing including footwear. Wash affected area thoroughly with soap and water. If any effects persist, seek medical attention.
Eyes:	Gently flush with running water, holding eyelid open for a 15 minute period. Seek medical attention if irritation persists.
Inhalation: Not	applicable.
Advice to Doctor:	Contact Poisons Information Centre on 131 126.

PRECAUTIONS FOR USE

Incompatible with:	Strong oxidizing agents (i.e. chlorine, peroxides) and acid.
Ventilation:	Use in a well ventilated area. As no vapour hazard exists, no special precautions are required.
Protective Equipment:	Splash-proof goggles, work boots/clothes are recommended when handling Environmental Lime. Some individuals with sensitive skin may require PVC/leather gloves if prolonged skin contact occurs.

SAFE HANDLING INFORMATION

Storage and Transport:	Environmental Lime should be stored in a cool, protected place, away from strong oxidants or acids. Environmental Lime is not regulated for transport purposes. However, transport is usually containerized bags or in bulk road tankers.
Spills and Disposals:	Suitably attired personnel should clean up spillages with a broom or shovel. Materials should be recycled or neutralised with diluted hydrochloric acid (HCl) to a pH of 7–9.
Fire/Explosion Hazard:	Environmental Lime is non-combustible. Violent reaction with all forms of acids.

OTHER INFORMATION:	Reacts with all types of acidic materials.
CONTACT POINT:	LIME INDUSTRIES PTY LTD (08) 9446 8644

Glossary

Acidity	The amount of acidity associated with all dissolved ions in a solution expressed as an amount of pure calcium carbonate needed to neutralise these. This includes hydrogen ions (as measured by pH) and commonly ‘free’ dissolved metals such as aluminium, iron and manganese. Other dissolved metals such as lead and zinc only begin to make a significant contribution to this acidity at high concentrations (>1 mg/L) – rare in acidic Wheatbelt water (Degens et al. 2008b). Organic acids may also contribute to acidity in some natural water, particularly those originating from peat wetlands but their contribution is insignificant. Acidity is found by measuring the amount of sodium hydroxide (NaOH or caustic soda) needed to raise the pH of a litre of the solution to 8.3 (see Appendix B; Kirby & Cravotta 2005). Acidity can also be calculated from measurements of pH, soluble aluminium, iron and manganese concentrations (in mg/L or ppm) using the formula (after Watzlaf et al. 2004): $\text{Acid}_{\text{calc}} = 1.79 \times \text{Fe}^{2+} + 2.68 \times \text{Fe}^{3+} + 5.56 \times \text{Al} + 1.82 \times \text{Mn} + 50000 \times 10^{-\text{pH}}$ Note that when water pH exceeds 3.5, most of the dissolved iron is likely to be present at Fe^{2+} with iron as Fe^{3+} probably only present in significant concentrations when water pH is less than 3.5 (Langmuir 1997). From this equation, simple ‘rules of thumb’ are that every g/L of dissolved aluminium requires 5.6 g $CaCO_3$/L to neutralise, every g/L of iron (as ferrous iron) requires 1.8 g $CaCO_3$/L to neutralise and every g/L of manganese requires 1.8 g $CaCO_3$/L to neutralise.
Acidity load	The total quantity of acidity in water for a particular period of time (day, week or year). This can be *estimated* by multiplying the flow for an arbitrary period by the average acidity concentration for that period. For example, given an annual flow of 30 million litres and an average acidity of 0.5 g $CaCO_3$/L, the annual acidity load is equivalent to 15 tonnes of $CaCO_3$ (which is the same as 15 tonnes of sulfuric acid). The acidity load can also be calculated by summing the products of water volume and acidity for intervals measured within a period of time (e.g. for an annual load, summing monthly flows multiplied by average monthly acidity concentrations). It must always be borne in mind that the actual acidity load can sometimes be very different from the estimated acidity, particularly when based on one or two measurements (Degens & Donohue 2002). The accuracy and precision of load estimates can be increased by taking more frequent measurements of flow and acidity to calculate acidity load (Degens & Donohue 2002).

Alkalinity	Largely the concentration of bicarbonate (HCO_3^-) in most water (except when pH >8.3) and can be measured in any water with pH >4.5 by measuring the amount of hydrochloric acid (HCl) needed to lower pH of a litre of the solution to 4.5 (see Kirby & Cravotta 2005). Sometimes alkalinity is due to ions other than bicarbonate, such as dissolved silicate, phosphate or ammonia (Langmuir 1997), but only if in relatively high concentrations (i.e. tens of mg/L).
Baseflow	Steady flow conditions when groundwater discharge is the main source of water in drain (i.e. no water from recent rainfall flowing out of the system)
Bioaccumulate	Concentration and storage of, usually, organic (e.g. persistent hydrocarbons) or inorganic (e.g. metals) contaminants within organisms often by organisms consuming others that have accumulated these.
Cellulosic materials	Generally, woody plant materials with a high proportion of cellulose and other complex carbohydrates that make up the structural components of plant cells. Often slow to decompose.
Composting bed	An organic mixture formed into a submerged layer which, in the context of this guideline, is decomposed by microbes under permanently water-logged conditions.
Microbial-sulfate reduction	A biochemical process involving sulfate reduction carried out by specialist types of bacteria living in waterlogged organic muds. When oxygen is scarce, these bacteria use dissolved sulfate as a surrogate for oxygen, which results in sulfate chemically reduced to sulfide (of which ‘rotten egg’ gas i.e. hydrogen sulfide is one form).
Metal acidity	Term applied to the acidity due to dissolved metals in acidic water (commonly aluminium, iron and to a lesser extent manganese in Wheatbelt water).
Net acidity (or net acidic)	This property of water accounts for dissolved alkalinity (such as bicarbonate or carbonate) in describing the degree of acidity in water. Some groundwater and water from treatment systems at pH > 4.5 can contain both dissolved acidity (as mineral acidity) and alkalinity. Net acidity = acidity – alkalinity (as dissolved bicarbonate or carbonate) measured in each case as units of $CaCO_3$ equivalent/L. See also – Acidity and Alkalinity.
Non-cellulosic materials	Generally organic materials with high proportions of simple carbohydrates and less complex sugars such as starch. These materials can be decomposed rapidly by microbes and include manures, digested or composted plant or food waste, spent or spoilt grains and young plant

	materials.
Oxidation	Oxidation is the reverse of reduction and, in chemical terms, occurs when the transformation of an atom from one oxidation state to another involves the gain of electrons. An example is when iron reacts with air and begins to rust. The iron atoms in this case lose an electron to oxygen and form rust.
Redox condition	Basically a measure of the extent to which an environment is oxidising or reducing. Its measurement is based upon the availability of electrons within the environment (measured in the pore water) reflecting the balance of biogeochemical reactions. For example, in oxidising environments, where there is a prevalence of oxidation reactions (see Oxidation), there is a tendency for electrons to be more available. In reducing environments, where reduction reactions are dominant (see Reduction), there is general lack of free electrons and greater tendency for electrons to be consumed. The availability of electrons, reflecting the redox status, can be measured using a glass redox probe and is reported as a voltage, where negative voltages indicate a flow of electrons into an environment (from the probe) reflecting their consumption by reduction reactions. Waterlogging an organic-rich material can create a reducing environment where entry of oxygen impeded by water and microbial activity drives the consumption of electrons to the extent that nitrogen can be reduced (to nitrogen gases) and sulfate (to hydrogen sulfide).
Reduction	Reduction is the reverse of oxidation and can greatly influence the reactivity of some acidic drain water constituents such as iron, manganese and sulfate. In chemical terms, reduction is the transformation of an atom from one oxidation state to another involving the loss of electrons. An example is dissolved sulfate transformed to hydrogen sulfide gas (rotten egg gas) by bacteria. During this transformation, the sulfur atoms in sulfate gain electrons (losing oxygen atoms in the process) and become sulfide atoms within hydrogen sulfide gas.
pH	This is a measure of the concentration of hydrogen ions (H^+) in water, measured on a scale of 1–14, where the mid-point 7 is neutral. In technical terms, pH values are calculated as the negative of the base-10 logarithm of the hydrogen ion concentration. In practical terms, this means that each unit change in pH represents a 10-fold change in hydrogen ion concentration so that, for example, a fall in pH from 4 to 3 represents a 10-fold increase in hydrogen ion concentration. The measure also applies to soils and muds, but only because pH is measured in the water held in the pores of these materials (i.e. it is not possible to measure pH in a dry soil).

Polishing water	A final treatment that may be needed to aerate water or allow materials to settle out before release to the environment or re-use. This often involves holding water in a pond for a set period of time.
Salinity (water)	This is a measure of the total soluble (or dissolved) salt (TDS) in water. This is sometimes measured by weighing the residue after completely drying a sample of filtered water. This material is often mostly a mix of predominantly sodium, calcium, potassium, magnesium as chloride, sulfate and bicarbonate (for alkaline waters) salts expressed as a weight per volume of water (mg/L).
	Salinity can also be calculated by summing the major ions (as in this report) or estimated from measurements of electrical conductivity (EC) by normalising to a measurement equivalent to water at 25°C (often called temperature compensation) and converting to TDS by multiplying the temperature compensated EC by 5.072 for EC 0–261 mS/m, by 5.889 for EC 261–1701 mS/m and by 7.820 for EC greater than 1701 mS/m.

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Department of Water

168 St Georges Terrace, Perth, Western Australia
PO Box K822 Perth Western Australia 6842
Phone: (08) 6364 7600
Fax: (08) 6364 7601
www.water.wa.gov.au