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Ecological Impacts and Management of Acid Sulphate Soil: A Review

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Abstract: The ecological impacts of acid sulphate soils (ASS) and the management of the impacts are a major concern globally. The reasons being how to minimize the exposure, reduce the residual impacts and make available management options. Despite the numerous studies, no review exists that synthesizes the major findings of the impacts from an agricultural soil, water and environment pollution point of view. Therefore, this paper presents a synthesis of the impacts on water and the soil together with the management options that are available. The review also identifies main areas that need further investigations.

Key words: Acid sulphate soils, ecological impacts, management options.

Introduction

Acid sulphate soils are naturally occurring soil, sediment or substrate formed under waterlogged, reducing conditions (Wilson, 2005) with sulphide minerals containing sulphuric acid or have the potential to form it, in an amount that can have adverse impacts (Fitzpatrick et al., 2009). The diversity of ASS types formed is attributed to different soil forming factors, such as the natural environment, anthropogenic modified environments (e.g. barrages, locks and blocking banks) and changing climatic environments (drought triggered and winter rainfall events) that exist. In general, ASS can occur in subaqueous, waterlogged and drained conditions in coastal, inland, mine spoil and wetland environments (Fitzpatrick et al., 2008b; Fitzpatrick et al., 2010b). Acid sulphate soil contains oxidized minerals (pyrites; FeS_2) or their products and has been described as the "nastiest soil on earth" because of strong acidity, its ability to mobilize toxic elements and deoxygenate water systems (Dent and Pons, 1995).

In an undisturbed state below the groundwater table, ASS are benign. If the ASS is drained, excavated or exposed to air by lowering of the water table, sulphides present in the sulphidic soils get oxidized and react with oxygen to form sulphuric acid (H_2SO_4) (Nordmyr et al., 2008). Release of the sulphuric acid in turn releases iron species (Fe^{2t}, Fe^{3+}), aluminum (Al^{3+}) and other potentially toxic elements into the soil and water systems (Roos and Astrom, 2005; Nordmyr et al., 2008; Ljung et al., 2009; Poch et al., 2009; Ljung et al., 2010), and are the major cause of ASS impacts. Although ASS covers only a small area (17-24 million hectares) of the global soils (Figure 1), once the acid and the toxic elements are mobilized, they have a lot of ecological impacts (Simpson Pedini, 1985; Poch et al., 2009). To put the impacts into perspective, acidity discharged from farmlands into drains in Australia is estimated to be 400-3400 kg H_2SO_4 per ha annually (Cook and Gardner, 2001) whilst an estimated $189 million is spent in Queensland, Australia annually to manage a 2.3 million ha of ASS along its coastline (Sutherland and Powell, 2000).

There are sufficient literatures available on the impacts of ASS due to various investigations, however, there is no paper that comprehensively synthesizes the ASS literature of the impacts on soil, water and living things in ASS environments. Therefore, this paper aims to put together the ASS impacts on soil, water and living things as the three major components of a given ASS environment. The paper additionally highlights key areas in which further researches need to be conducted to manage this problem soil and discusses principal management options that are available.

Formation

Acid sulphate soils have formed within the last 10,000 years, after the last sea level rise (Dent, 1986; Cook et al., 2000; Joukainen and Yli-Halla, 2003; Wilson, 2005). When the sea level rose and inundated the land, sulphate in the sea water mixed with the iron oxides in the sediments and organic matter, allowing microorganisms to form iron sulphides (pyrites) under the anaerobic conditions (Bloomfield and Coulter, 1973) (reaction 1). The pyrite formed is stable under such conditions unless exposed to the air, is oxidized and sulphuric acid is produced (Bensryd et al., 1994; Lin et al., 2000; Sullivan et al., 2002; Ahern et al., 2004; Appleyard et al., 2004; Fitzpatrick et al., 2008). The exposure and oxidation of the pyrite occur as a result of natural isostatic land uplift (Dent, 1986; Cook et al., 2000), artificial drainage or natural phenomena such as falling water levels, e.g. droughts (Brown and Jurinak, 1989; Ward et al., 2004; Burton et al., 2008; Faltmarsch et al., 2009; Reid and Butcher, 2011).

Changes in soil moisture regimes caused by land use (Kawahigashi et al., 2008a), stockpiling of soils containing reduced inorganic sulphur during excavation of drainage canals (Sammut et al., 1996; Minh et al., 1998; Cook et al., 2000; Kawahigashi et al., 2008a), ploughed top soils, construction of ditches and drains, raised beds, excavated soil surfaces and destruction of impermeable soil layers are other sources of pyrite exposure (Minh et al., 1997). Ponded pastures, aquaculture ponds, sand/gravel extractions, roads, rail and golf courses on ASS are additional contributing factors (Powell and Martens, 2005). In the tropics, the distinct dry-wet season causes seasonal variations in acid formation with more acids being produced during the dry seasons and the ultimate reduction during the wet seasons (Minh et al., 1998; Husson et al., 2000).

The oxidation process leading to formation and generation of acidity (H^+) and the biochemical pathways that initially convert pyrite to ferrous iron (Fe^{2+}) and sulphate (SO_4^{2-}) have received considerable attentions and are well established (reactions 2–5) (Nordstrom, 1982; Lin et al., 2000; Thomas et al., 2003; Sullivan et al., 2009):

$$\text{(i)}\quad Fe_2O_{3(s)} + 4SO^{-2}_{(aq)} + 8CH_2O + \tfrac{1}{2}O_{2\,(g)} \longrightarrow 2FeS_{2(s)} + 8CHO_{3\ (aq)}^{-} + 4H_2O_{(aq)} \qquad (1)$$

$$\text{(ii)}\quad FeS_{2(s)} + 3\tfrac{1}{2}O_{2\,(g,\,aq)} + 3\tfrac{1}{2}H_2O \longrightarrow Fe(OH)_{3\ (aq)} + 4H^+_{(aq)} + 2SO_4^{2-}{}_{aq} \qquad (2)$$

$$\text{(iii)}\quad FeS_{2(s)} + 3\tfrac{1}{2}O_{2\,(g,\,aq)} + H_2O \longrightarrow Fe^{2+}_{(aq)} + 2H^+_{(aq)} + 2SO_4^{2-}{}_{aq)} \qquad (3)$$

$$\text{(iv)}\quad Fe^{2+}_{(aq)} + H^+_{(aq)} + \tfrac{1}{4}O_2 \longrightarrow Fe^{3+} + \tfrac{1}{2}H_2O \qquad (4)$$

$$\text{(v)}\quad FeS_{2(s)} + 14Fe^{3+}_{(aq)} + 8H_2O \longrightarrow 15Fe^{2+}_{(aq)} + 16H^+_{(aq)} + 2SO_4^{2-}{}_{(aq)} \qquad (5)$$

Based on the reactions presented, reaction 2 shows that each mole of pyrite consumed yields 4 moles of acidity and reaction 3 shows that the first stage of pyrite oxidation results in formation of ferrous iron (Fe^{2+}), acid and sulphate. The ferrous iron so produced is then further oxidized to ferric iron (Fe^{3+}) through a much faster reaction catalyzed by *Acidothiobacilus ferrooxidans* at low pH (4) (Nordstorm, 1982). At pH less than 4, the ferric iron remains soluble and can react as the oxidant with pyrite to produce further acid (5). The Fe^{2+} and H^+ can then end up back in reaction 2, and the reactions 2 and 3 can then drive the rate of pyrite oxidation further. Overall, reactions 2 and 3 produce two moles of acid for every mole of pyrite consumed. The ferrous iron may also be transported back up the soil profile (van Breemen, 1975) by capillary rise (Lin et al., 1998) or diffusion (Patrick and Delaune, 1972) and reaction 2 can recur if oxygen is present (Cook et al., 2000). If the ferric iron produced is transported down the soil profile to the pyrite then the iron generated in reactions 2 and 3 can produce oxidation of pyrite further, even when the pyrite is submerged.

Ecological Impacts

Historical studies on ASS impacts dates back to the early 1970s (Dent and Pons, 1995; Ljung et al., 2009) and since then, quite a lot of studies have been done to address the impacts (Simpson and Pedini, 1985; Tang and Yu, 1999; Sammut, 2004; Buschmann et al., 2008; Ljung et al., 2009). Many well-documented studies stressed that impacts on the soil, water and living things, which form the basis of this review are crucially important, as they are part of some important ecosystems in many environments with different ecological functions (Simpson and Pedini, 1985; Robarge and Johnson, 1992; Hanhart et al., 1997; Tang and Yu, 1999; Meda et al., 2001; Ward et al., 2002;

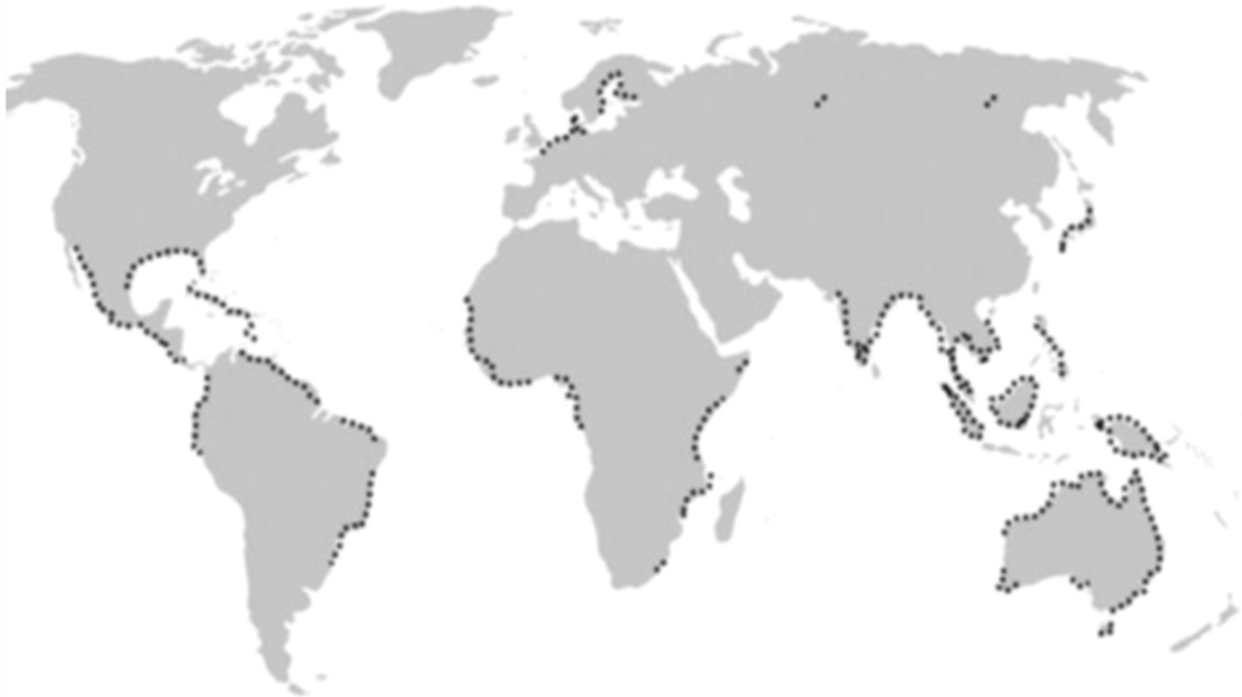

Figure 1: Of the estimated 17-24 million ha of global ASS, 6.5 million occur in Asia, 4.5 million in Africa, 3 million in Australia, 3 million in Latin America, 200 000 in Finland, 235 000 in Sweden and 100 000 in North America, respectively (Simpson and Pedini, 1985; Faltmarsch et al., 2009; Poch et al., 2009).

Joukainen and Yli-Halla, 2003; Sammut, 2004; Powell and Martens, 2005; Hinwood et al., 2006; Buschmann et al., 2008; Nordmyr et al., 2008; Roziere et al., 2009; Haling et al., 2010; Haling et al., 2011).

The major problem with ASS, apart from the sulphuric acid produced, is that when the soil pH drops below 5, many potentially toxic metals are solubilised into the soil solution and this is the most important process through which ASS toxify water and soil systems (Kinraide, 1991; Sammut, 2004; Fitzpatrick et al., 2008a; Fitzpatrick et al., 2008b; Ljung et al., 2009; Simpson et al., 2010). The acidification process is particularly severe when the ANC of the soil is low and acid production exceeds the soil's capacity to neutralize it (Ahern et al., 2004).

Impacts on Soil

Agricultural productivity, especially in the developing world, is an important component of the livelihood of communities where people depend on farm produce for daily food needs. In these communities, population densities are often high compared to the limited arable land available and more ASS wetlands are drained and converted into agricultural farmlands (Shamshuddin et al., 2004; Kawahigashi et al., 2008a; Österholm and Åström, 2008) and infrastructure installed (Buschmann et al., 2008), or old ASS continued to be ploughed and cropped; e.g. in Finland for over 100 years (Yli-Halla et al., 1999). Such activities have resulted in contamination of different agricultural soils due to the production of sulphuric acid, accumulation of mineral contaminants or deficiencies in essential soil nutrients, e.g. P deficiency (Ren et al., 2004) and foul smells (e.g. from H_2S or organo-S-compounds) (Fitzpatrick et al., 2008a) that make the soil environments unconducive for continuous cultivation (Desmond, 2000; Gosavi et al., 2004). Low soil fertility and unproductiveness due to oxidation of Fe compounds caused by reclamation and drainage has been observed as a cause of agricultural soil problem in ASS (van Breemen, 1975).

There is also a high risk of potentially toxic element mobility within the soil systems (Joukainen and Yli-Halla, 2003; Nordmyr et al., 2008) due to the low soil pH or changes in climatic conditions (Kawahigashi et al., 2008b; Simpson et al., 2010) that end up in neighbouring soil systems (Ren et al., 2004; Burton et al., 2006; Burton et al., 2008; Miller et al., 2010). A study tracking changes in metal distributions as a result of leakage from ASS has found that an appreciable amount of Al, Co, Cd, Cu, Mn, Ni and Zn were elevated in the surface and sub-surface of bottom sediments (Fältmarsch et al., 2008; Nordmyr et al., 2008). Such events cause uneven distribution of soil nutrients and affect crop productivities and yields. Similar observations were made elsewhere (e.g. Fitzpatrick et al., 2008a). Low soil pH coupled with prolong falling of soil water levels following events such as drought or inadequate management of soil have the potentials to affect soil biological activities (Kawahigashi et al., 2008a; Simpson et al., 2010).

Sheoran and Sheoran (2006) stated that microorganism such as sulphate reducing bacteria act on metal sulphides in ASS and convert them into hydrogen sulphide, which reacts with heavy metals to form highly insoluble metal sulphides, making ASS unconducive for microbial productivity. Soil acidity can have negative impacts on the formation and effectiveness of symbiotic associations (Maki et al., 2008). For instance, Bradyrhizbioum sp. growth at lower pH in vitro was slow and did not fix N despite nodule formation (Cline and Senwo, 1994). Trace element molybdenum, needed by legumes for nitrogen fixation and protein synthesis, was unavailable in acid soils (Duncan, 1999). Important ecological functions such as maintenance of fertility, nutrient cycling and decomposition of organic matter by microorganisms are regrettably affected by extreme acid soil conditions under ASS environments (McGrath et al., 1995; Oliveira and Pampulha, 2006). A study conducted to evaluate the changes in soil microbial characteristics showed that heterotrophic and asymbiotic N fixing bacteria, fungal and actinomycetes were sensitive to the presence of heavy metals (Oliveira and Pampulha, 2006).

Impacts on Water Systems

Water form an important component of many activities including farming, municipal and industrial uses, recreation, and sports (Sammut et al., 1995; Appleyard et al., 2004). In many places, disturbed ASS is a potent source of acidity in coastal and inland waterways (Cook et al., 2000; Baldwin and Fraser, 2009; Baldwin and Mitchell, 2012). Cook et al. (2000), for instance, monitored export of acidity in drainage water from ASS in East Trinity, Queensland, Australia and found considerable acid in the water. Where developments are disturbing ASS, mobilization of heavy metals into water is of major public concern (Hindwood et al., 2006; Hinwood et al., 2008) because of frequent massive fish kills; e.g. in New South Wales, Australia in the 1980s (Powell and Waite, 2000) and serious damages to aquatic biota (Faltmarsch et al., 2009), marine and terrestrial ecosystems including biodiversity and death of mangroves (Appleyard et al., 2004; Powell and Martens, 2005). In ASS disturbed locations in Swan Coastal Plain, Western Australia, tests have shown in ground or surface water elevated concentrations of heavy metals that could possibly have both water and human health impacts (Appleyard et al., 2004). Sammut et al. (1996) reported high concentrations of elements in a well in excess of drinking and recreational water quality guidelines and in some cases irrigation guidelines (Hindwood et al., 2006). Similar results were reported by workers in South Australia (Fitzpatrick et al., 2008a; Fitzpatrick et al., 2009; Fitzpatrick et al., 2010; Fitzpatrick et al., 2010b).

The concentrations of metals in produce irrigated with ASS contaminated water varies according to the types and parts of the vegetable tested (Hindwood et al., 2006). The water chemistry is also reported to affect the uptake by plants with more metals being taken up in acidic waters (Bensryd et al., 1994; Hinwood et al., 2008). A study tracking changes in metal distribution by Nordmyor et al. (2008) found large amounts of Al, Cd, Co, Cu, Mn, Ni and Zn being transported downstream in Vora River in Finland, which got deposited in estuarine sediments. They also observed that metals such as Fe, Cr and V, which are easily immobilized in ASS, were not enriched in the sediment cores, indicating a high mobility of mobile metals. Consequently, severe damages on aquaculture were reported (Hudd and Kjellman, 2002). Metal mobilization test in another study using sulphidic soils that have undergone oxidation have observed rapid release of metals and the dissolved concentrations often exceeded the Australian water quality guidelines for protection of ecosystem health (Fitzpatrick et al., 2008a).

One of the important ecological risks associated with oxidation of sulphidic sediment, apart from acidification and metal mobilization is deoxygenation of water systems (Fitzpatrick et al., 2008a), which results from oxidation of ferrous iron (Fe^{2+}) to ferric iron (Fe^{3+}) (Cook et al., 2000) which consumes oxygen in water (Sullivan and Bush, 2000) and it is believed to be one of the major impacts responsible for polluting aquatic and estuary ecosystems (Bush et al., 2002). Such an acidic water system with low level of oxygen and high concentration of toxic metals is undesirable for most forms of aquatic life (Cook et al., 2000). Drainage of acidity coupled with low levels of oxygen are likely to have major impacts on most aquatic life forms, inshore fisheries and breeding grounds of marine organisms. Example of such events that resulted in fish kill and other aquatic organisms in lagoons, rivers, aquaculture ponds, natural lakes and waterways have also been observed and reported (Barker, 1978; Simpson and Pedini, 1985; Sammut et al., 1995; Sammut et al., 1996; McCarthy et al., 2006; Stepens and Ingram, 2006). Powell and Martens (2005) indicated that reef fish are likely to be affected due to the impacts on spawning and nursery areas. Oxidation of toxic heavy metals (e.g. Cd) and metalloids (e.g. As) in water (Burton et al., 2008) can react with other metals (Baldwin and Fraser, 2009) and the oxidized minerals released directly into water systems can be taken up by aquatic plants or animal, which can eventually kill them (Lootermoser, 2007).

Impacts on Living Things

The effects of low soil pH is one of the major constraints to plant root growth and development, yet there is limited understanding of this potential stress, which affects over half of the world's productive agricultural land (Haling et al., 2011). Food crops grown by farmers are often affected by the increase in soil acidity (Meda et al., 2001) and accumulation of toxic elements previously discussed (Fältmarsch et al., 2010). For example, a study conducted to quantify the inhibitory effect of metals on wheat (*Triticum aestivum*) and cucumber (*Cucumis sativis*) found reduced seed germination, root elongation and coleoptile and hypocotyl growths (Munzuroglu and Geckil, 2002). In Malaysia, cocoa (*Theobroma cacao*) yield on ASS was reported to be low (Shamshuddin et al., 2004). Pasture grasses [cocksfoot (*Dactylis glomerata*), phalaris (*Phalaris aquatic*a) and weeping grass (*Microlaena stipoides*)] or crop plants such as *Triticum* spp. and barley (*Hordeum vulgare*) are also affected by low pH (Haling et al., 2010). A similar study reports that the root length of some plants were sensitive to acidity, with lateral root length more sensitive than seminal root length (Haling et al., 2011). In Thailand and Vietnam, Al toxicity from ASS was seen to stunt rice plants (van Breemen and Pons, 1978; Hanhart et al., 1997). The information, however on uptake and bioaccumulation including the analysis of mineral contaminants in the plant biomass is limited to a number of studies (Fältmarsch et al., 2010). The current knowledge on the chemical compositions of agricultural food crops grown on ASS is limited (Yli-Halla and Palko, 1987; Hindwood et al., 2006; Faltmarsch et al., 2009; Fältmarsch et al., 2010).

Faltmarsch et al. (2010) investigated the influence of geochemistry on the concentrations of certain elements in cabbage (*Brassica oleracea* L. *var. capitata*). This study reported that easily available element (Ca, P, Ni, Mn, Cu and Fe) concentrations were not elevated. In a similar study in Finland, Mn was however elevated in oat grains together with Co and Ni (Yli-Halla and Palko, 1987). The lack of sufficient information on crops grown on ASS is evident and more information are needed since metal accumulation in edible parts of plants serve as a direct entry into the human food chains (Hinwood et al., 2008; Fältmarsch et al., 2010). Information on the use of food crops such as wheat (*Triticum* spp.), rice (*Oryza sativum*), maize (*Zea mays*) or pasture plant such as *lucerne* grown under ASS, which are important to the well-being of people and livestock are limited in the ASS literature. Not a single study has been conducted to investigate the effects of ASS contaminated water on the performances of crop plants, livestock and aquaculture (Appleyard et al., 2004; Hindwood et al., 2006; Buschmann et al., 2008; Ljung et al., 2009).

A small number of studies using food crops have been conducted (Kochain et al., 2004; Hindwood et al., 2006; Haling et al., 2010), however the soil types used were not ASS, ASS contaminated or the experiments were carried out under controlled laboratory conditions with artificial supplements of metal contaminants. A study conducted on 67 residents in Perth, WA reported high concentration of metals in residential groundwater bores affected by ASS disturbance, with high proportion of the residents reported using the water for irrigation of home-grown produce (Hindwood et al., 2006). The latter study found Al and Pb at very high concentrations with elevated concentrations of Cd. A preliminary investigation carried out to assess human exposure to metals in groundwater affected by ASS disturbance of residents using bore water for home-grown produce irrigation showed higher metal concentrations in hair (Hinwood et al., 2008). Given increased use of bore water irrigation and greater groundwater use during drying periods, it was hypothesized that there is potential for human exposure. Studies elsewhere show that exposure to metals such as Cd, Pb, As and Al pose health hazards (Jansson, 2001; Järup, 2003; Järup and Alfvén, 2004). Moreover, information on disease incidences of animal production on ASS are limited (Duncan, 1999; Ljung et al., 2009) as metal such as Al is toxic to living things (ANZECC and ARMCANZ, 2000; Fitzpatrick et al., 2008a). The impacts of ASS on human well-being have been extensively reviewed and the conclusion drawn was that the moment of the acid and metals pose potential threats to human (Ljung et al., 2009).

Impact Management

Ecological impact assessment and management of ASS have become an important issue (Vegas-Vilarrubia et al., 2008) and many countries (e.g. Australia, Finland, Sweden) have developed resource use and management strategies and policy guidelines to manage the impacts (Ljung et al., 2009). The principal management options explored include minimize disturbance, application of an alkaline material to neutralize the actual acidity, incorporation of organic matter to improve the microbial activities of the soil and recycle nutrients (Oliveira and Pampulha, 2006), and use of plants to extract chemical pollutants from the contaminated soils (Haling et al., 2010). Many studies are of the opinion that the complex processes of ASS formation and development should be assessed carefully, together with the interactions between different ecosystems (soil and water, plant and soil or soil-water-living things) that exist (Ward et al., 2004; Hindwood et al., 2006; Kijne, 2006).

Policy and Legislation

The general understanding is that the impact assessment of ASS initially has to take into account the policy and the legal, as well as the social aspects of ASS impacts, including the exploration of best management strategies (Ljung et al., 2009). As it is sometimes difficult to get people to understand the legal aspects of managing and protecting areas that are impacted, proper policies should be put in place to make the management legal. When such policies are put in place, the people including other stakeholders should be made to fully understand the policies and how well they can help implement them and manage the impacted areas. At the same time, make available to impacted areas management strategies to consider through stakeholder consultations, workshops, field visits or demonstrations. Summary of such policies relevant to water quality management have been presented by Powell and Martens (2005) based on ASS management strategy of Queensland, Australia. The major elements presented were: policy, regulation and lead agent; awareness, education and training; mapping and assessment of ASS; planning, management and environmental advice; research and development; and regional community participation.

Inclusion of land-use decision making and planning policy together with proper ground and surface water management (Joukainen and Yli-Halla, 2003; Hindwood et al., 2006) and the soil physical conditions are a good component of the management of ASS impacts (Sammut et al., 1996; Sammut, 2004; Haling et al., 2011). Kijne (2006) for example, investigated the best water use management strategies for ASS in rice growing areas and found that the best option was to abandon the cultivated area and relocate the farmers. Such guidelines on land and resources uses are important because of the continuous pressure on coastal and inland ASS from urban development, which may result in hastened decisions regarding land development. Acid sulphate soil management knowledge is likely to be limited among urban decision makers (Ljung et al., 2009). As such, more people should be trained as part of the management plans on how to identify the occurrence, accurately measure the presence and at the same time carry out field impact assessments where ASS could be present, especially in developing countries where the impacts are poorly understood or management processes are not well established.

Impacted Soils

Neutralization of the actual (sulphuric) or potential (sulphidic) soil with an alkaline material such as agricultural lime (Ca_2CO_3) has been the common practice to manage soil acidity (Xu and Coventry, 2003; Rigby et al., 2006; Baldwin and Fraser, 2009). Under farming conditions, many farmers have the practical problems of maintaining and improving the conditions of the soils even if this management option is available (Hanhart et al., 1997). The reasons being that the lime is expensive, not readily available or the amount required is too much and therefore impractical (Shamshuddin et al., 2004). Under such situations, farmers tend to use alternative strategies such as maintaining high water levels in drainage canals by closing them at the lower end of the field or higher levels (Minh et al., 1998). In Vietnam, tillage, groundwater level control and mulching techniques have been applied to reduce evaporation and accumulation of Al^{3+} toxicity in the surface soils (Minh et al., 1998). Leaving the ASS undisturbed under their natural conditions, protecting them from disturbances or leaving them covered by water are the principal management strategies also considered in soils implicated with ASS disturbances (Ljung et al., 2009; Sullivan et al., 2009).

In ASS rice paddy, rotation with dry land crops has been considered as a general practice for better land use (Tri and van Mensvoort, 2004). After rotating the crops, the ASS was reflooded for paddy rice cultivation as a potential remediation technique (Dent, 1992). Sustainable production and use of perennial plants including grasses to manage the impacts have been considered as well (Robarge and Johnson, 1992;

Haling et al., 2011). Acid tolerant crop plants can be introduced to restore and rehabilitate the impacted soils (Kochian et al., 2004; Kijne, 2006; Baldwin and Fraser, 2009; Haling et al., 2011). From a soil fertility point of view, good maintenance of the soil microbial activities, enhancing the ANC of the soil (Madsen et al., 1985) and minimize use of chemical fertilizers containing elemental sulphur (Aulakh et al., 2002) were proposed (Xu and Coventry, 2003). Revegetation of ASS using pioneer plants associated with fungi and N-fixing bacteria for acquisition of mineral nutrition was reported and the results have demonstrated significant growth of pioneer grasses and legume shrubs in a soil at pH 3.4 (Maki et al., 2008).

Incorporation of organic matter on acid soil was seen to improve the soil chemistry and nutrients due to the release of organic carbon, increase soil pH and decrease exchangeable Al (Pocknee and Sumner, 1997; Meda et al., 2001; Xu and Coventry, 2003). Mixing of ASS with an alkaline material such as sandy loam was observed to prevent either a sulphidic soil from acidifying or neutralized the residual effects of a sulphuric soil (Michael et al., 2012). The current knowledge on understanding the impacts of plants growing on ASS chemistry under different moisture regimes are limited to very few studies (Reid and Butcher, 2011) and warrants further research (Michael, 2011). Our literature search done for current studies conducted to investigate the use of plant organic matter to manage ASS shows that effects of organic matter on ASS chemistry has never been investigated. However, a study conducted in Malaysia shows that Al^{3+} toxicity in ASS was reduced when peat, or peat in combination with green manure, rice straw, chicken dung and palm oil mill sludge were added (Shamshuddin et al., 2004). In many places where ASS are abundant, e.g. large parts of south-east Asia, the potential environmental threats posed are yet unknown and should be studied (Nordmyr et al., 2008).

Impacted Water Systems

The major concerns raised by many studies are not the primary impacts but the impacts of the polluted environments and their end products on water, which seem to have received less attention from researchers. This is due to the fact that secondary impacts are far greater than the primary, which are thought to be more damaging due to increase in concentrations or amounts of the pollutants over time, compared to the initial pollutants. Many impacted water systems are hard to restore to their original conditions and the management option is to rehabilitate and improve the ecological characters (Baldwin and Fraser, 2009). In the context of inland waterways rehabilitation impacted by sulphidic sediments, Baldwin and Fraser (2009) comprehensively discussed that formation of reduced organic sulphur in sediments should be initially minimized, then individual water bodies rehabilitated and finally the receiving waters protected.

To date, monitoring of new disturbances has been restricted to a number of developments requiring ASS management conditions only and little analysis of water quality trends have been undertaken in reef catchments, targeted monitoring and reporting of water quality (Powell and Martens, 2005). Similarly, Fitzpatrick et al. (2009) considered minimizing the disturbance and drainage of ASS, preventing oxidation of sulphidic materials, reducing the oxidation rates and isolating higher risk materials from exposure, or containing and treating acid drainage to minimize the risk of significant impact on water systems as alternative management strategies.

Separation of sulphidic materials, hastening oxidation, collecting and treating of acidic leachate, management of stockpiled soil materials, and planning and development controls were highlighted as important options by other studies (Sammut et al., 1996; Minh et al., 1998; Cook et al., 2000; Kawahigashi et al., 2008a). After working on verifying the presence or absence of the three main types of ASS [sulphidic, sulphuric and monosulphidic black ooze materials] in subaqueous, waterlogged and drained soil environments in nine unmanaged wetlands in the adjacent River Murray, Fitzpatrick et al. (2009) recommended that further researches needed to be carried out to investigate the potential hazards of lime application in aquatic systems. They pointed out that such researches should emphasize on the impacts on all stages of the life-cycle of invertebrate fauna and test the accuracy of the calculation of the amount of lime required for treatment in wetland basins on the basis that over-dozing with lime will further enhance ecological impacts. Not only that, research on developing procedures, monitoring protocols to plan and manage remedial re-wetting of ASS affected wetlands, with or without the application of lime (Baldwin and Fraser, 2009; Baldwin and Mitchell, 2012) and use of vegetation communities to understand the impacts and recovery processes are not well understood (Michael, 2011).

The problem related to liming is that the added reagents may have direct adverse effects on biota or indirect adverse effects due to the re-dissociation of precipitated metals. The other argument is that addition of Ca or Mn rich alkaline reagents might increase the hardness of water and turgidity that might prevent light penetration (Green et al., 2006). Limestone also forms precipitates, especially oxyhydroxides of Al and Fe^{3+} that can coat the limestone surface, thereby hindering the direct contact with the acid and have unnecessary impact on ecosystem (Hammarstrom et al., 2003). The literature on liming of acidified water from 1985 to 2004 have been reviewed and the conclusion drawn was that most studies concentrated on monitoring the impacts of liming on different fish and microscopic plants including some benthic organisms and no research on the use of plant health as biological indicators of ecological impacts of liming on wetlands or other plant-water ecosystems have ever been investigated (Clair and Hindar, 2005).

Conclusions

The ecological impacts caused by the sulphuric acid produced due to the oxidation of pyrite, accumulations of toxic metals as a result of the solubilized soil matrices which make metals to stay in solution and deoxygenation of water are the three major ASS environmental problems known. These include acidification and toxication of agricultural soils, pollution of water systems used for irrigation, recreational activities, fishing, aquaculture and domestic uses and the ultimate effects on living things. Deoxygenation of water, either in water systems or soils is the third major impact experienced in ASS environments. Many reports show that increased concentration of metals in food crops grown on ASS is evident and pose potential risk of human health. Thus, understanding the basic mechanisms of ASS formation and where they occur, how the presence can be identified and measured are crucially becoming important, especially in places where ASS is a problem (Figure 1).

The importance of ASS and the types of ecological impacts on the environment posed have resulted in many researches being carried out and information on the formation and occurrence, identification and measurement, and the possible management options that can be used to manage the impacts are available. The principal management options are minimizing disturbance of the sulphidic soil materials, avoid drainage of overlying water bodies and preventing the oxidation of the sulphidic material if exposed. Application of an acid neutralizing agent such as lime continues to be the common practice in many places but availability is a problem, especially in the developing countries and alternative management strategies still need to be developed and put in place. Many researchers have highlighted resource (soil and water) uses and management as one of the important management principles, as it can initially allow users to identify ASS presence at an early stage, which would minimize all the likely events that are associated with exposure and oxidation, production of sulphuric acid and resultant recruiting and accumulation of toxic metals in the soil and water systems.

Despite the numerous investigations carried out, many studies continue to highlight areas that need further investigations to fully understand the biochemical processes involved in producing and propagating the impacts. Major areas highlighted include use of plants or plant products (e.g. mulch) to manage the impacts, especially on soil and, avoid tissue accumulations of toxic elements by plants growing on ASS and contaminated water systems. Our current findings and ongoing studies seem to indicate that use of plant organic matter, which are readily available and relatively cheap compared to agricultural lime and alkaline soil material such as sandy loam (Michael et al., 2012) seem to have the potential to either prevent a sulphidic soil from acidifying or neutralize the residual effects of an existing sulphuric soil, especially under farmland conditions. Use of such materials need rigorous investigation as they are cheaply available and the findings would be easily adapted on a wider scale in the developing world.

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