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Hamdan Jol
Shamshuddin Jusop
Mohd. Izuan Effendi Halmi

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Preface

Land is an indispensable resource for the survival and prosperity of mankind as well as for the maintenance of all terrestrial ecosystems. In agriculture, potential production of arable land and its susceptibility to degradation are dependent on the management strategies employed and on inherent soil and other characteristics. Over the years, increased demand and pressure on land resources results in competition for land use, declining crop production and degradation of land quality in agriculture areas.

In the tropics, it is known that agricultural intensification of an area for cropping causes decline in soil quality. Transforming the natural ecosystem to an agro-ecosystem can involve significant changes in organic matter input and turnover rates. These changes alter the infiltration, water retention, bulk density, soil strength, mineralization of nutrients and their availability to plants, cation exchange capacity, soil fertility and accumulation of heavy metals and toxic chemicals.

This book embodies the results of the on-going research on tropical soils as well as those completed in the last several years by the academic staff of the Department of Land Management, Faculty of Agriculture, Universiti Putra Malaysia and collaborative research works with other research agencies in Malaysia. Eight articles were submitted and published, covering wide range of topics in the area of soil genesis, soil physic, soil chemistry, soil microbiology and strategic soil fertility management in the tropics, particularly Malaysia.

The editors wish to thank the Dean, Faculty of Agriculture, Universiti Putra Malaysia (UPM) and the Head, Department of Land Management, Faculty of Agriculture, UPM, for their encouragement, support and invaluable comments. Last but not least, we acknowledge all our colleagues in the Department of Land Management for their strong commitment in making this important publication a reality.

Hamdan Jol
Shamshuddin Jusop
Mohd. Izuan Effendi Halmi
Serdang, 2017

1

Can Oil Palm Cultivation in the Tropics Contributes to Global warming and/or Land Degradation Significantly?

Shamshuddin Jusop

Introduction

People cut down forest on the Earth surface throughout human history for different reasons. The most important reason being to grow crops to feed the ever increasing world population. Much to our regret, deforestation is getting out of control in many parts of the globe, especially in Central and South America, Southeast Asia and Africa, leading to decrease in biodiversity as well as enhancing the processes that contribute to global warming and/or land degradation in the affected countries. Come what may, it has been proven true by a group of people calling themselves the Union of Concerned Scientists; they wrote that tropical deforestation contributed to about 15% of global warming pollution worldwide (Boucher *et al*., 2011). This paper discusses whether cutting forest in the tropics for oil palm cultivation significantly enhances the processes of global warming and/or land degradation.

Concern citizen of the world assume that global warming phenomenon is partly contributed by the release of CO_2 into the atmosphere when trees in the tropical forest are cut down to make way for agriculture; by the way, carbon is the most important component of tree biomass. This popular notion is in line with the study of Houghton (2010) who found that in tropical regions croplands contributed the most to the carbon emission. The question is which crop and in what area of the tropics this phenomenon is rampant? Does oil palm cultivation play an important role in this so-called carbon emission phenomenon?

Be that as it may, soils in the upland areas of the tropics are mostly highly weathered with pH below 5; however, oil palm (Figure 1) is acid tolerant and as such it can grow quite well on the soils because of proper agronomic management (Shamshuddin and Fauziah, 2010; Shamshuddin *et al.*, 2015). Without doubt oil palm cultivation on such soils contributes significantly towards sustainable economic growth of the poor and the under privileged throughout the tropical regions. This is particularly obvious in Indonesia and Malaysia, the current two most important producer/exporter of palm oil in the world. Unfortunately, of late, Indonesia has been accused of being the main contributor to the environmental degradation and global warming arising from oil palm cultivation (IPOA, 2013). Malaysia is not spared from being accused of similar offence.

Figure 1 A well maintained oil palm plantation in Malaysia

For all intents and purposes, Malaysia firmly believes that her oil palm industry is sustainable as it follows strictly to the guidelines imposed by the importers of palm oil. For instance, during land preparation at re-planting phase of oil palm cultivation, zero-burning is imposed to all oil palm plantations throughout the country. This sustainable practice had received due recognition from the United Nations for keeping up with the acceptable environmental services.

For the good of everyone, palm oil which is used in everything from fuel to instant noodles, is currently the most used cooking oil in the world. The oil can be used to manufacture a wide range of consumer (in the kitchen) as well as industrial products (in the factory) as oleo-chemicals and others. Global supply of palm oil, the world's most tradeable vegetable oil, is maintained due to the strong demand from the traditional market in China, India, Pakistan and European Union. Indonesia and Malaysia account for about 85% of the total world's palm oil production (STARBIZZWEEK, 2015). For these two countries, oil palm grown at settlement schemes throughout the length and breadth of the countries is used as a means of eradicating poverty of the landless and have not. Many say this is the corporate responsibility of the governments in the tropics that care about the living conditions of their people.

Worldwide Distribution of Oil Palm Cultivation

Oil palm is a hardened crop that is suitable for cultivating in the tropical regions of the world based on climatic conditions as well as the suitability of the soils (Shamshuddin *et al.*, 2015). As the palm is originated from the swamp of Africa, it can even be grown successfully on peat soils with high water table. As this paper shows, oil palm is cultivated the most in Southeast Asian region with increasing area over the years (Figure 2), and Indonesia and Malaysia, in decreasing order, take the lead in oil palm production in the world. We know that the area allocated to oil palm cultivation in Southeast Asia is much less compared to the combined area used for rubber, cocoa, fruits and rice production. However, oil palm is a very important crop, although it does not occupy the largest cultivated land in the region.

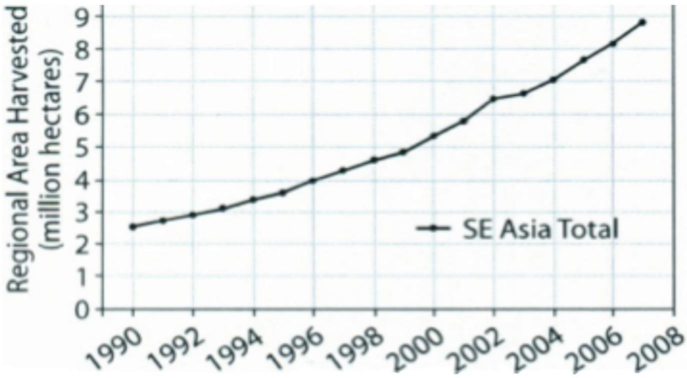


Figure 2 The growth in area harvested for palm oil [(Modified from (FAO, 2011)]

It is known, of late, that large scale cultivation of oil palm is not yet popular in Africa as it is in Southeast Asian countries, but it may very well be in the future considering the economic benefit it brings about to the rural poor in the continent. On the other hand, oil palm area is in the upward trend in Columbia, South America. However, the area cropped to oil palm in the above countries is nowhere near that of the Indonesia plus Malaysia.

Effects of Deforestation on the Environment

It has been known for years that Latin America has the largest tropical forest in the world, especially within the Amazon basin (Boucher *et al.*, 2011). Be that as it may, the countries in this region have probably contributed the most to the forest clearing in the past many years. Among the important effect of this deforestation is land degradation due to soil erosion which in some ways contributes to global warming. Houghton (2010) had studied CO_2 emissions from deforestation and land degradation under shifting cultivation, pasture, industrial harvest, croplands and afforestation in tropical regions (Asia, Africa and Latin America). His study concluded that, in Asia, croplands emitted CO_2 the most, while in Latin America it was pasture. Which crop in Asia is mostly responsible for the emission of the gas? This paper tries to put solid arguments that oil palm cultivation is not the main cause of the problem in question.

In recent years, oil palm cultivation is encroaching into the so-called fragile peatlands of the tropics, especially in Indonesia (Kalimantan and Sumatra) and Malaysia (Sarawak). Note that the expansion is for a good reason as it caters to the need of the people in the two countries; increase in oil palm cultivation requires more land area. Sad to say that, the cultivation of oil palm on the peatlands has been blamed for accelerating CO_2 emission into the atmosphere. The government of the two countries firmly believes, supported by strong evidence, that this accusation is unfounded and therefore false. Recent studies conducted in Indonesia had proven beyond doubt that they were right (IPOA, 2013). The findings of the study of the peatland in Sarawak by Melling *et al.* (2005) further supported the conclusion by the Indonesians.

Can Oil Palm Plantation Be Considered as Forest?

What is a forest? According to IPOA (2013), oil palm plantation fits very well into the accepted definition of forest proposed by many countries in the world. We know that mature oil palm has a canopy cover of almost 100%, with palm height reaching above 5 m in normal plantation. Furthermore, in many parts of the world, timber plantation and forest plantation are considered as forest. In relation to that, FAO (2005) had already classified rubber plantation as forest. As we know it, there is little difference between rubber and oil palm plantation in terms of the land area under cultivation and their economic life of about 25 years. Once planted on the land allocated to them, the crops would grow like forest species that release O_2 during the day and absorb CO_2 at night time. For the information of everyone, oil palm is known to be a net absorber of CO_2 (IPOA, 2013). This means that oil palm plantation can be considered as forest, just like rubber plantation is. How can we blame oil palm cultivation contributes significantly to global warming? It does not make sense, to say the least.

The Main Agents of Deforestation in Tropical Region

We take for granted that the principal agents of deforestation in the tropics are:

1. Agriculture;
2. Logging; and
3. Mining.

Agricultural activities that contribute to deforestation in the tropics include cattle rearing which uses big area for small amount of food it produces. According to Boucher *et al*. (2011), the pasture used for cattle rearing made up about 70% of global agricultural land. As Brazil is among the biggest exporter of beef in the world, it is expected that a large area of the Amazon forest has been cleared for cattle production. Brazil is also a major producer of soybean. Putting the two together, deforestation is expected to be huge in the Amazon basin. Who can deny that?

It is already mentioned that oil palm cultivation has not been spared from being blamed for contributing to deforestation in the tropics. Is this right? It has been mentioned earlier that the area cropped to oil palm in Southeast Asia, especially in Indonesia and Malaysia, is in the increasing trend over the years (Figure 2). It is undeniably true, but the increase in the area cropped to oil palm is for very good reasons. However, still the area under oil palm cultivation is small in comparison with that used for cattle grazing and/or agricultural crops in Brazil. Additionally, in certain parts of the tropics, the rural poor cut down forest for fuelwood. This, to a certain extent, contributes to the overall deforestation in the world.

Nowadays, logging activities, legal or otherwise, are common in the forest of Brazil, Indonesia and Malaysia (particularly in Sarawak). This, in some ways, enhances the economic wellbeing of the local inhabitants in particular and the countries in general, but at what expense? Forest and land in the affected areas are degraded, sometimes beyond imagination. Many areas in the remote parts of Brazil and Indonesia have been spotted via satellite images to have been subjected to rampant logging activities. Hence, it could very well be that logging activity is one of the most important drivers of deforestation in the tropics.

Peninsular Malaysia has a long history of tin mining industry with many of its cities namely Kuala Lumpur, Ipoh and Taiping have prospered because of the precious metal. Due to tin mining activities for long time, a large tract of a luxuriant forest area in the Kinta Valley, Perak, had been destroyed to make way for it. What was once a luxuriant forest area in Kinta Valley had been turned into wastelands (sandy areas, slime and lakes) unfit for agriculture and to date the land affected by tin mining activities is about 200,000 ha (Shamshuddin, 1990). Tin mining industry is no longer important option for the economic growth of Malaysia.

In recent time, mining contributes to about 5% to the GDP of Indonesia (Devi and Prayogo, 2013). The areas where mining is active are West Papua, East Kalimantan and West Nusa Tenggara where coal and nickel are mined with the growth rate of about 10% per year. Other minerals of importance to the Indonesian economic growth are gold and copper. For sure, large forest area has been deforested for mining purposes. Likewise, mining activities are very active in the outback of the Amazon basin where it is difficult to control by the government of the day. All in all, deforestation due to mining activities in the tropics is significant. Considering the three economic activities in proper perspective, we should not point our fingers at oil palm for enhancing global warming, let alone to be single out as the driver of deforestation in the tropics.

Is Oil Palm Cultivation in The Tropics Sustainable?

Oil palm starts to bear fruits after 30 months of field planting and will continue to be productive for the next 25 years (Figure 1). In general, it produces 4-5 t/ha/year of crude palm oil (CPO), a very good yield indeed. Furthermore, oil palm produces about 1 t of palm kernels per year by the same ha of land. Although the price of the latter is lower compared to that of the former, it gives extra income to the growers. Hence, oil palm cultivation helps contribute to the eradication of poverty in the tropical regions of the globe.

Results of many studies conducted in Malaysia indicated that oil palm is the most efficient oil-bearing crop in the world, requiring only 0.26 ha of land to produce 1 t of oil; in comparison, soybean, sunflower and rapeseed require 2.22, 2.0 and 1.52 ha, respectively, to produce the same amount of oil (MPOC, 2012). It is therefore consistent with the study of Rival and Levang (2014) who found that oil palm production is very efficient. According to them, oil palm occupies only 7% of the land devoted to oil-producing plants worldwide, but produces 39% of the global supply of vegetable oil. On the other hand, soybean produces 27% of the world's vegetable oil, but occupies 61% of the land used to produce the oil. This means that oil palm cultivation worldwide is economically viable and/or sustainable in the long run. Oil palm is a crop to watch in the future for human survival.

It has been known for many years that soil erosion leads to serious land degradation in tropical regions where rainfall erosivity is usually high. This is well known notion among the soil fraternity worldwide. Soil erosion is one of the most important factors that reduce the productivity of soils in the upland of Malaysia, especially in areas covered by highly weathered soils (the Ultisols and Oxisols) which are mostly located in undulating and sloping lands. This is made worse by the action of heavy raindrops during the wet season. Research conducted in the tropics found that the rate of soil erosion in areas under forest is low, estimated at 0.03-6.2 t/ha/year (Wiersum, 1985). However, when the forest is cleared to make way for agriculture, soil erosion rate is accelerated, depending on soil management practices as well as crop type.

In Malaysia, Lim (1990) found that on bare Rengam Series (an Ultisol developed on granite, a rock that covers about 50% of Malaysia's land surface) with slope of 3-5^{o} lost 28 t/ha/year soil due to erosion. Further research in the country which had been conducted by the Malaysian Palm Oil Board (MPOB) concluded that bare Munchong Series (an Oxisol developed on shale) with slope of < 5^{o} had an erosion loss of 12.5 t/ha/year (PORIM, 1990).

It was found that in areas cropped to oil palm, soil erosion could be higher compared to those in the natural forest, but with proper soil management practices in the plantations, it can be kept to the minimum (Khalid and Tarmizi, 2008). To keep the oil palm in the plantations growing healthily, NPK fertilizers and soil amendments are regularly applied. It has been found that the long-term effects of applying them were positive by way of maintaining/increasing soil pH, exchangeable bases and cation exchange capacity (Shamshuddin and Noordin, 2011; Shamshuddin and Fauziah, 2010; Shamshuddin *et al.*, 2015). Thus, oil palm cultivation on such soils (Ultisols and Oxisols which occupy about 70% of Malaysia's land area) is best described as sustainable; hence, it is an excellent option to be used for crop production.

Malaysian government through its Department of Agriculture, Peninsular Malaysia, Ministry of Agriculture, has instructed that land with slope > 25^0 should not be used for agricultural activities. This is clearly defined in the new guidelines proposed for good agricultural practice in Malaysia (myGAP). These guidelines have all along been mostly followed by plantation sector in the country. Hence, oil palm in Malaysia is either cultivated on flat or to undulating land with slope not exceeding the level outlined in myGAP.

In the Malaysian estates where oil palm is grown, there are areas such as the inter-rows where soil erosion can still serious occur. These areas have to be appropriately conserved in order to sustain oil palm production. To achieve this purpose in the long run, the inter-rows are planted with cover crops (Figure 3). It clearly shows that the inter-rows of the oil palm are fully covered by legume species. After 4 years or more from the planting date, oil palm canopy closes up completely (Figure 1) and the legume dies off that contributes to a fair amount of nitrogen into the soils. A part from increasing soil fertility significantly, the cost of fertilizer application can be somewhat reduced. In this way, soil erosion rate is reduced or curtailed, while soil productivity is maintained or increased; hence, crop growth is unaffected but the yield improves. This is translated into better income to the farming community than it otherwise is.

Figure 3 The topsoil in the plantation is protected against erosion by cover crop during the early stage of oil palm growth

The cover crops normally planted in oil palm plantations so discussed above belong to the legume group of plant species. It means that this legume can fix nitrogen from the atmosphere and this nitrogen stays in the plant until it is dead. Nutrient contents of legume are high, contributing 113 kg N/ha, 11 kg P/ha, 106 kg K/ha and 9 kg Mg/ha to the soils (Khalid *et al*., 2000). This in the end becomes part of the organic matter in the soils. When this newly added organic matter is mineralized, essential nutrients (NPK) are made readily available to the growing oil palm in the field. This practice is for sure would cut down the cost of production a bit as less NPK fertilizers are applied so as to keep the crop in the field going. While the cover crop is in the field, it helps prevent soil erosion by reducing the impact of raindrops during high rainfall (thunder storm). Furthermore, its extensive root system encourages soil aggregation that improves as well as stabilizes soil structures, further enhancing soil productivity. Growing cover crops in oil palm estates is known to be one of the most innovative agricultural practices in the plantation sector. This is one of the most important scientific contributions of soil scientists and/or agronomists in Malaysia to the industry.

Further option for conserving topsoil in oil palm plantations in Malaysia is construction of terraces, which is usually being carried out in fragile areas, such as on sloping lands with slope > 5^0 (Figure 4). During heavy rain, some topsoil in the plantations is eroded, transported and finally deposited on the terraces and hence the fertile materials are kept within their confine. Under normal circumstances, the areas in between the terraces are covered by cover crops which by themselves are able to reduce the rainfall erosivity. By the act of constructing these terraces in oil palm estates, soil erosion is reduced to the minimum level. Hence, oil palm cultivation on slightly sloping lands does not result in serious land degradation.

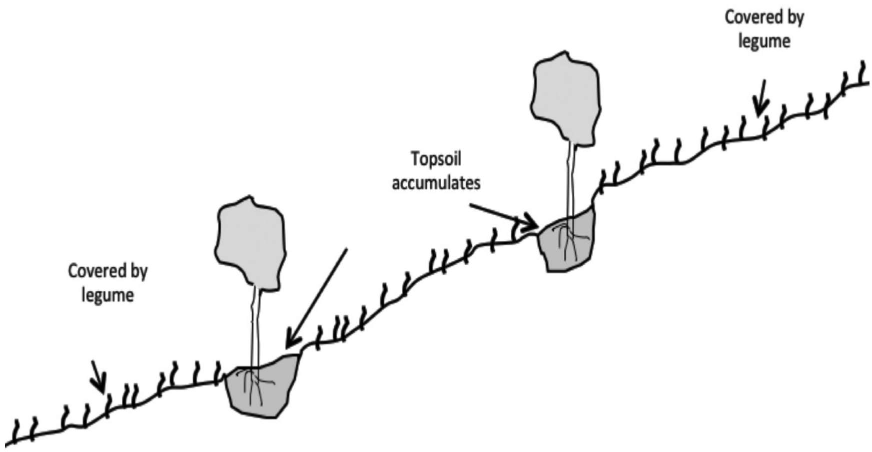


Figure 4 Constructing terraces in oil palm plantation to reduce topsoil erosion (After Shamshuddin *et al.*, 2015)

Oil Palm Cultivation Eradicates Rural Poverty

Oil palm cultivation contributes towards sustainable economic growth of rural folks and the under privileged in the different part of the tropics. For instance, it has been reported in the Malaysian national press (NST, 2015) that palm oil production was a catalyst for Sarawak's economic growth. According to Sarawak Deputy Chief Minister, the state was trying to transform oil palm cultivation into a modern, dynamic and competitive sector. To attain that desire, Sarawak State adopted a policy of allocating 4 ha of land per farmer to grow oil palm on sustainable basis using the latest agronomic practice. This is expected to generate income for each family of USD 600-900 a month; hence, hard core poverty common in the state could slowly be alleviated. The same is true for the farming communities in the other parts of the world, such as in Indonesian outback (Kalimantan and Sumatra). This is not to mention about the economic benefits gained by the rural poor in tropical Africa and South America where oil palm can be suitably cultivated based on climatic conditions and soil properties.

Conclusion

It is opined that the economic benefit gained from growing oil palm in the tropics supersedes the problem related to global warming and/or land degradation. In Malaysia and Indonesia, oil palm is grown simply because it has to (for economic reason), not due to greed. Sure enough agricultural activities contribute to deforestation and/or land degradation in the tropics. As this write-up has shown it, oil palm cultivation is not the main player in the deforestation. In Brazil, forest is cleared in a big way to make way for soybean cultivation as well as for cattle rearing; these are big business though for the country. Another important contributor to deforestation in the tropical region is mining activity, either legal or illegal. For years, a lot of land in Indonesia are cleared for coal and nickel mining, especially in the remote parts of the republic. It has been shown in this paper that growing oil palm on tropical peat soils did not have significant effect on CO_2 emission. So, we cannot blame oil palm cultivation on tropical peat soils for the enhanced global warning.

Thinking aloud about deforestation at large scale in the tropics, there was a documentary film entitled "The Fight for Amazonia" being aired on Malaysian TV (Aljazeera channel) on April 15, 2015. This special documentary film described in detail about the widespread deforestation that has taken place in Brazil of late. It gave clear pictorial accounts of the widespread destruction of the Amazonia by the unscrupulous farmers and/or poachers trying to make a living via agricultural and logging activities or even mining for precious metals to be sold at the marketplace. Illegal mining activities involving local inhabitants in the Brazilian outback had also been clearly shown. It was explained that even the forest reserves were not spared by these activities. In the end, patches of areas in the vast Amazon basin which were previously covered by luxuriant natural forest have been cleared and/or destroyed forever.

Oil palm, for reasons best known to Brazil, is not cultivated in the Amazon. This proves yet again that oil palm cultivation is not the main cause of land degradation in the tropics, let alone be considered as the driver of deforestation currently taken place throughout the globe. For the growers of the golden crop in Indonesia and Malaysia, they can say that there is more to oil palm cultivation than meet the eye! For all intents and purposes, we have to respect their view.

References

Boucher, D., P. Elias, K. Lininger, C. May-Tobin, S. Roqemore and E. Saxon. 2011. The Root of the Problem. Union of Concerned Scientists. UCS Publications, Cambridge, USA.

Devi, B. and D. Prayogo. 2013. Mining and Development in Indonesia: An Overview of the Regulatory Framework and Policies. AusAID, Canberra.

FAO. 2005. Global Forest Resources Assessment. Food and Agricultural Organization, Rome.

FAO. 2011. Southeast Asian forest and forestry to 2020: Sub-regional report of the 2nd Asia-Pacific forestry sector. Food and Agricultural Organization, Bangkok.

Houghton, R.A. 2010. How well do we know the flux of $CO_2$from land-use change? Tellus 62B: 337-351.

IPOA. 2013. Indonesian and oil palm plantations amid global environmental issues. Indonesian Palm Oil Association, Jakarta, Indonesia.

Khalid, H., Z.Z. Zin and I.M. Anderson. 2000. Nutrient cycling in oil palm plantation: the effect of residue management practices during replanting on dry matter and nutrient uptake of young palm. J. Oil Palm Res. 12(2): 29-37.

Khalid, H. and A.M. Tarmizi. 2008. Techniques of soil and water conservation and nutrient cycling in oil palm plantation on inland soils. Oil Palm Bull. 56: 1-11.

Lim, K.H. 1990. Soil erosion under oil palm on slopes. In: *Proc. PORIM Interna*tional *Palm Oil Development Congress.* Jalani *et al*. (eds). Palm Oil Research Institute Malaysia, Bangi.

Melling, L., R. Hatano and K.J. Goh. 2005. Soil CO_2 flux from ecosystem in tropical peatlands of Sarawak, Malaysia. Tellus 57: 1-11.

MPOC. 2012. The oil palm tree. Http: www.mpoc.org.my/The_Oil_Palm_Tree.aspx (retrieved on March 30, 2015).

NST. 2015. Palm Oil: A catalyst for Sarawak's economic growth. New Sunday Times, March 15, 2015. Page 7.

Rival, R. and P. Levang. 2014. Palms of Controversies: Oil Palms and Development Challenges. CGIAR Consortiun Office, Montpellier, France.

Shamshuddin, J. 1990. Sifat dan Pengurusan Tanah di Malaysia. Dewan Bahasa dan Pustaka, Kuala Lumpur.

Shamshuddin, J and C.I. Fauziah. 2010. Highly Weathered Tropical Soils: The Ultisols and Oxisols. UPM Press, Serdang.

Shamshuddin, J. and Noordin Wan Daud. 2011. Classification and management of highly weathered soils in Malaysia for production of plantation crops. In: Principle, Application and Assessment in Soil Science. E. Burcu and G. Zkaraova (eds.), published by Intech. pp: 75-86.

Shamshuddin, J., W. Wan Noordin, I. Roslan, C.I. Fauziah and Q.A. Panhwar. 2015. Ultisols and Oxisols: Enhancing their productivity for oil palm, rubber and cocoa cultivation. UPM Press, Serdang.

STARBIZZWEEK. 2015. The Star, January 31, 2015.

Wiersum, K.F. 1985. Effects of various vegetation layers on *Acasia auriculiformis* forest plantation on surface erosion in Java, Indonesia. In: Soil Erosion and Conservation, S.A. El-Swaify, W.C. Moldenhauer and A. Low (eds). Soil Conservation of America. pp: 19-89.

2

Remediation of Metal(loid)s-rich or Contaminated Soils: In-situ Immobilization Technique

Che Fauziah Ishak and Rosazlin Abdullah

Introduction

Food safety which is an integral part of food security is gaining prominence as a global issue. Soil plays a central role in food safety as it has a direct influence on the composition of food and feed at the root of the food chain. However, the quality of soil resources as defined by their potential impact on human health by propagation of harmful elements through the food chain has not been given the proper emphasis due to lack of data of adequate detail and reliability. The impact of metal(loid)s on the environment should also be given due consideration.

Unlike organic contaminants, metal(loid)s do not undergo microbial or chemical degradation and persist for a long time unless transported out of the site. Metal(loid)s can be retained in the soil by sorption, precipitation, and complexation reactions, and are removed from soil through plant uptake, leaching and volatilization. From our earlier work undertaken to study the heavy metals distribution in agricultural soils of Malaysia (Zarcinas *et al.*, 2004), concentrations of Co, Ni, Pb and Zn in the soils were strongly correlated with soils concentrations of Al and Fe, which suggest evidence of background variations due to changes in soil mineralogy. Chromium was correlated with pH and EC, Na, S and Ca suggesting association with acid sulphate soil and soil salinity components, while Hg was not correlated with any of these components, suggesting diffuse pollution by aerial deposition. Arsenic, Cd, Cu were strongly associated with aqua-regia soluble and available P, and organic matter suggesting these metals are associated with agricultural inputs in agricultural fertilisers and soil organic amendments. This indicates metal(loid)s contamination to the majority of agricultural soils in Peninsular Malaysia is due to anthropogenic activity (possibly added in fertilizers, wastes, pesticides, effluents or atmospheric sources) that may pose a risk to the environment or human health.

Dynamics of Heavy Metal(Loid)S in Soils

The basic principle involved in the immobilization technique is that the metal(loid) of concern is removed from soil solution either through adsorption, complexation, and precipitation reactions, thereby rendering the metal(loid) unavailable for human and plant uptake and leaching to groundwater. Heavy metal incorporation in the soil is controlled by adsorption processes, such as surface complexation and ionic exchange, but other mechanisms such as precipitation are likely to contribute to metal retention in the soil (Sastre *et al.*, 2006). Adsorption is defined as the accumulation of ions at the interface between a solid phase and an aqueous phase. Adsorption isotherms have been widely used in studies on adsorption phenomena, supplying numerical parameters that provide information on the retention capacity and intensity of the metal by the soil (Casagrande *et al.*, 2008). The advantage of these equations is that they can be applied to adsorption of any ions and gives straight forward parameters which can be related to soil properties.

Soil Amendments for Remediation and Approaches Taken in Remediation

In many cases, a byproduct may not be ideal by itself for land application. Through co-utilization of byproducts, more products that are agronomically useful may result. The benefits of co-utilization may include nutrient balance, reduction of toxins or contaminants, improved moisture content, improved economic value, and improved soil conditioning effects. In-situ immobilization is a cost-effective approach where land-applied amendments are used to stabilize contaminants via adsorption and/or precipitation reactions that render the contaminant immobile (Adriano, 1986). Numerous inorganic amendments such as clays, Al/Fe/Mn oxides and hydroxides may be land applied to metal rich or contaminated soils as means of reducing metal mobility. Nowadays, there is pressure on waste management managers to find ways to convert wastes into resources instead of sending them to the landfill. Amongst the mineral wastes recommended for in-situ immobilization of heavy metals are red gypsum, coal fly ash and drinking water treatment residues. There has also been report that combination of mineral byproducts with biosolids or organic materials is in fact a more viable option than applying the byproducts singly for soil fertility improvement for crop production.

Studies have been conducted on selected amendments for the immobilization of metal(loid)s (Table 1). These amendments can be categorized as phosphate compounds, liming materials, organic materials and metal oxides (Fe-rich materials).

Table 1 Selected soil amendments in the immobilization of metal(loid)s in soils

Amendments	Metal(loid)s	Observations	References
Liming materials			
Ground magnesium limestone (GML)	Mn	Manganese concentration in the soil solutions decreased exponentially with increasing pH after application of ground magnesium limestone.	Shamshuddin *et al*. 2009
Red gypsum (RG) Water treatment residue (WTR)	Cd, Cu, Ni, Pb and Zn	RG and WTR can be recommended as soil amendments because of their effectiveness in reducing metal(loid)s in the soil through changes in soil pH. However for WTR, P deficiency in plant has to be monitored because of WTR ability to fix P.	Fauziah *et al.* (2011), Nur Hanani *et al*. (2008)
Organic matter/Biosolid			
Biochar Oil Palm Empty Fruit Bunch – (EFB) and Rice Husk (RH)	Cd and As	Oil palm EFB and RH biochars indicate that these commercially produced biochar have good potential to be used as sorbents for As and Cd.	Sari *et al*. 2014

cont'd Table 1

Coal Fly ash (CFA)	Cd, Cu, Ni, Pb and Zn	CFA can significantly reduce the Zn derived from sewage sludge from being taken up by maize plants at application ratio of (10%:5%) and up to ratio 4:1 (20%:5%) coal fly ash to sewage sludge. However, CFA was not a good liming agent since the CCE was low.	Nur Hanani *et al*. 2010
Phosphate compounds			
Egyptian rock phosphate (ERP) Bone meal (BM) triple super - phosphate (TSP)	Pb	Applications of P-amendments shows effectiveness of TSP in reducing lead in soils. The best treatments are as follows: TSP>ERP>BM. These treatments were recorded able to stabilize the Pb as per indicated in the percentage reduction in phytoavailable form pools into a more stable form of complex. (Fig. 18)	Naim *et al*. (2017)

Liming Materials

Malaysian soils dominantly fall (about 75%) into the Ultisol and Oxisol Orders in Soil Taxonomy. These soils are generally acidic, pH of 4.0-5.0 and contain essentially of variable charge minerals, namely sesquioxides and kaolinite, thus, have low cation exchange capacity or cations retention capacity. In Malaysia, liming is the most common management practice used to overcome the problems associated with soil acidification. Most plants grow well at a pH range of 5.5–6.5 and liming is aimed to maintain the pH at this range. The main purpose of liming is to reduce aluminium toxicity in highly weathered acidic tropical soil, but at the same time, this practice can help reduce heavy metals availability to plants via precipitation process. Liming experiments on typical Ultisols (Shamshuddin *et al.* 1991; Ismail *et al.* 1993) and Oxisols (Shamshuddin *et al.* 1992) have indicated the need for liming for annual crop production. In a study on an Oxisol grown with cocoa, the application of lime at 2 t ha^{-1} reduced soil solution Mn concentration in the 0-15 cm layer from 27 to 12 µM after 3 months (Shamshuddin *et al.*, 1991). Manganese concentration in the soil solutions decreased exponentially with increasing pH after application of ground magnesium limestone (Figure 1). Applying lime in combination with gypsum would bring more Ca and/or Mg further down the soil profile (Shamshuddin and Ismail 1995), thus alleviating to some extent subsoil acidity. Applications of GML, usually known as dolomitic limestone, would also supply the necessary Ca and Mg needed for corn and groundnut growth. The presence of more Ca in the soils arising from lime and/or gypsum applications is also beneficial because Ca can to a certain extent alleviate Al toxicity (Alva *et al.* 1986). The increase in solution pH would certainly affect the availability of other metal(loid)s in the soil.

The increase in solution pH resulting from GML application is due to production of hydroxyl ions when GML is dissolved and subsequently hydrolysed:

$$Ca,Mg\,(CO_3)_2 + H_2O \rightarrow Ca^{2+} + Mg^{2+} + 2CO_3^{\;2-}$$

$$CO_3^{\;2-} + H_2O \rightarrow HCO_3^{\;-} + OH^-$$

The hydroxyl ions then reacts with Al in the solution to precipitate as aluminum hydroxide, which over time may crystallize into gibbsite [Al $(OH)_3$]:

$$Al^{3+} + 3OH^- \rightarrow Al\,(OH)_3$$

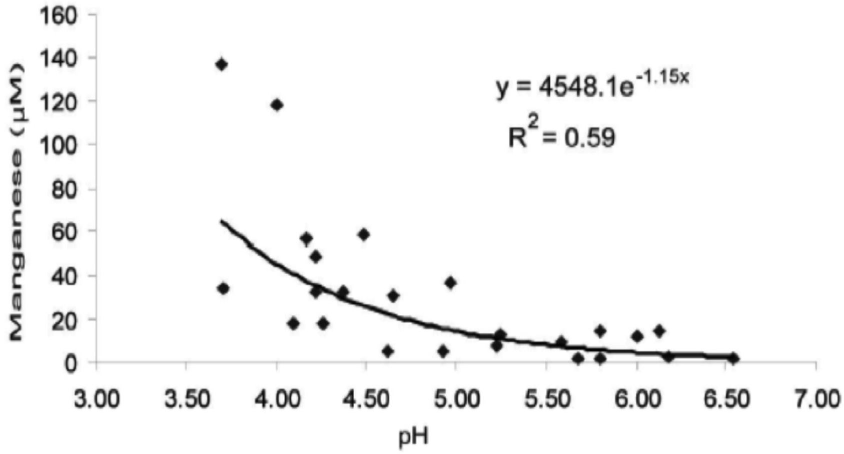


Figure 1 Relationship between soil solution Mn and pH (Shamshuddin *et al*., 1991)

Biochar

Biochar is a carbonaceous material which can adsorb metal(loid)s in soil and water. The main factors influencing the sorption behaviour of biochars are pyrolysis condition and the feedstock type in the production of biochars. One type of biochar may not be appropriate for all cases of remediation. The application of biochar to soil may improve the sorption capacity of metal(loid)s in soil. This carbonaceous product was reported to have many functional groups with high surface areas, which are likely related to its potential to act as an adsorbent.

In terms of remediation of heavy metal contaminated soils through their retention in the soil system, biochar has been considered to be potentially effective. A number of studies have also demonstrated that biochar has a high capacity to adsorb pollutants in contaminated soils (Beesley and Marmiroli, 2011; Yuan and Xu, 2011).

Biochar can stabilize the heavy metals in the contaminated soils, improve the quality of the contaminated soil and has a significant reduction in crop uptake of heavy metals. Biochar is a fine-grained charcoal-like material produced through pyrolysis, which is heating of biomass to temperature of 300-600 °C under air deprived conditions. Through pyrolysis, the feedstock changes chemically to form structures that are more resistant to microbial degradation than the original material.

Utilization of biochar as a soil amendment has attracted great interest globally due to the apparent benefits to soil fertility and plant growth as well as the potential to store or sequester C in the soil system. It has been reported that activated carbon, which is a subset of biochar, had been used as a substrate to improve the adsorption of heavy metals such as mercury, a process which is termed as chemisorption. The mechanism of heavy metal retention in soil by biochar can be categorized as physical or chemical in nature. The physical aspects deal more with filtering mechanism of the heavy metal due to its structure or size by the pore size of the biochar. It is important to characterize the pore size distribution of biochar, the percentage of macropore, mesopore, nanopore, because the type of pores dictate the extent of liquid-solid adsorption processes.

In Malaysia, a pilot scale biochar manufacturing plant using a modern engineering system has been built by Universiti Putra Malaysia (UPM), in collaboration with a private company (Nasmech Technology Sdn. Bhd.). The plant was built to produce biochar from oil palm empty fruit bunches (EFB) and is capable of producing 20 t of biochar daily. Additionally, biochar derived from rice husks has been produced commercially in Malaysia to avert wastage of large quantities of rice husks (RH). It is reported that 97,980 million tonnes of rice husk was produced annually during the processing in the mills (Bernas Sdn. Bhd.).

The surface morphology of biochar samples was observed under Jeol JSM-6400 scanning electron microscope (Sari *et al.*, 2014). Figure 2 shows that EFB biochar possesses uniform pores and smooth wall surfaces with maximal 20 µm in diameter. Small particle-like ashes were found scattered on the surface area of EFB biochar as observed in Figure 2. In comparison, the pores on rice husk biochar are not well-shaped with diminished structure of pores (Figure 3). Small pores on the rough rice husk biochar surface was observed as shown in Figure 3. Pyrolysis temperature can attribute to the pores formation and destruction on biochar. When low temperature was applied, the biochar cell structure and arrangement was found similar to the cell structure and arrangement of the original biomass (Pavithra, 2011). The stack of biochar cells and pores were arranged accordingly and well-shaped as found in the SEM image of EFB biochar. However, as the temperature increase, the pore size become enlarged and the walls between adjacent pores were destroyed (Zhang *et al.*, 2004), which explained the diminish pores on rice husk biochar. The lack of biochar structure also might be due to the volatilization process during the biochar production.

From the observation on the SEM images (Sari *et al.*, 2014), both biochar generally exhibit macropores with internal diameter size of 10 µm. The macroporosity (>50 nm) of biochars are relevant for soil aeration and water movement (Troeh and Thompson, 2005). Macropores also facilitate the root movement through the soil and act as habitats for the soil microbes (Saito and Muramoto, 2002). Hence, biochar has the potential to improve soil physical properties such as soil water retention and porosity. Basso *et al.*, (2013) reported the addition of biochar on sandy loam soil increased the water- holding capacity by 23% compared to the non-amended soil. Glaser *et al.* (2002) also found the increase of soil moisture at field capacity with the increase of char surface area and porous structure. The macropores are also important as feeder pores to transport adsorbate molecules to the meso- and micropores.

A mixture of meso- and micropores were also present on EFB and rice husk biochar surface. The micropores of biochar make the greatest contribution to total surface area, hence responsible for the high adsorption capacities of molecules (Rouquerol *et al.*, 1999). Mesopores are also of importance for many liquid-solid adsorption processes, as reported by Lua *et al.*, (2004), on pistachio-nut shells. Thus, based on the EFB and rice husk biochar structural surface, they have the potential to sorb metal and metalloid to reduce the mobility of these trace elements in soil.

The Brunauer, Emmett and teller (BET) surface area of biochar indicates the physical changes of biomass during the pyrolysis process. The surface area depends largely upon the carbon (C) mass removed during the processing, creating pores in the materials (Zabaniotou *et al.*, 2008). The sorption ability of biochar can be determined from its surface area, where high surface area will increase the sorption capacity. Surface area and porosity of EFB and rice husk biochar are presented in Table 2.

Biochar produced from EFB had a larger surface area than RH biochar. The higher surfaces area of EFB biochar may indicate the adsorption capacity of heavy metals compared to RH biochar. In general, biochar surface areas can be influenced by biochar's micropore volume, choice of feedstock and pyrolysis processing condition (Boateng *et al.*, 2007). The micropore volume of EFB biochar was found to be same as RH biochar (~0.01 cm^3/ g). Internal surface area of biochar which represent pore on the inner wall resulted from interior crack was referred to as micropore area. Meanwhile, the average pore diameter for both biochar are in the range of mesopores diameters, with the internal pore width between 2 to

50 nm. This indicates the potential of adsorption capacity EFB biochar and RH biochar in liquid-solid adsorption (Bagreev *et al*., 2001).

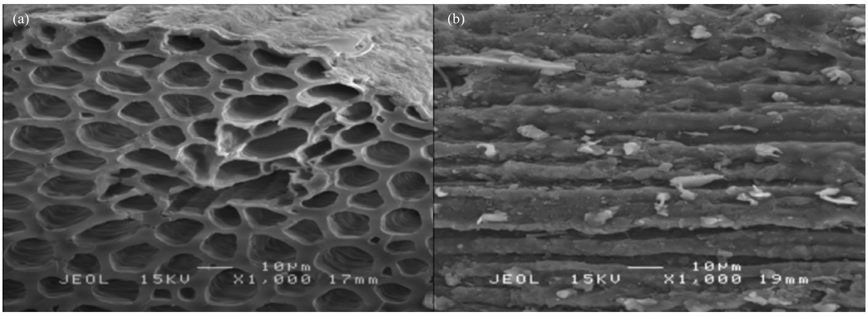


Figure 2 SEM image of EFB biochar at 1000 x magnification (Sari *et al*., 2014)

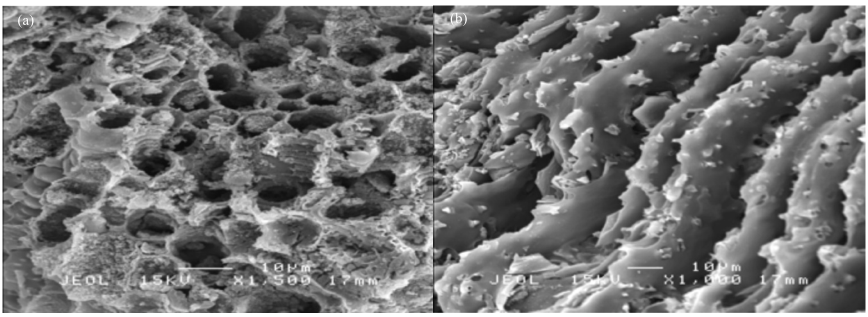


Figure 3 SEM image of rice husk biochar at (a)1500 and (b)1000 x magnification (Sari *et al*., 2014)

Table 2 BET surface area and porosity of biochars

Biochar	BET surface area (m^2/g)	Pore volume (cm^3/g)	Pore surface area (m^2/g)	Average pore diameter (nm)
EFB	46.32	0.01	0.61	3.85
Rice husk	23.22	0.01	1.41	4.34

The adsorption isotherm data were fitted to the Langmuir's adsorption model. Table 3 shows the values of adsorption isotherm parameters for EFB biochar and RH biochar. The maximum adsorption capacity (q_{max}) of EFB biochar for As was 0.424 mg g^{-1}, which is higher than RH biochar (0.352 mg g^{-1}). Similar trend was found on q_{max} of Cd with the values of 15.15 and 3.19 mg g^{-1}, for EFB biochar and RH biochar, respectively. The parameter b is related to the affinity of the binding sites, which allows comparisons of the affinity of biochar toward the metal(loid) ions. EFB biochar had a higher affinity for As than did RH biochar. In contrast, the binding affinity (b) of Cd for RH biochar is higher than EFB biochar. There are several factors attributed to sorption mechanism of trace elements with addition of biochar, of which the most important are pH and CEC (Kumpiene *et al.*, 2008). The alkaline properties of biochars increased the solution pH, which induced metal immobilization through metal precipitation and decreases metal solubility (Rees *et al.*, 2013). Value of R^2 shows correlation or linear relationship, whereas the relationship become more linear when the value is closer to 1. The high correlation coefficient values (R^2) which ranged from 0.98 to 0.99 indicate that the Langmuir isotherm best fitted the experimental data.

Table 3 Sorption isotherm obtained by fitting the data with the Langmuir isotherms for the EFB biochar and RH biochar

Biochar	Heavy metal	Langmuir model		
		q_{max} (mg g^{-1})	b (L mg^{-1})	R^2
EFB	As	0.4240	0.7299	0.9890
	Cd	15.1515	0.1142	0.9921
RH	As	0.3522	0.0248	0.9823
	Cd	3.1908	0.6920	0.9984

Samsuri *et al.*, (2013) reported coating the biochars with Fe (III) greatly increased their adsorption capacities for both As (III) and As (V). The results indicate that the commercially produced EFB and RH from Malaysia have good potentials to be used as adsorbents for As (III) from aqueous solutions. Furthermore, coating the EFB and RH with Fe (III) increased their adsorption capacities for both As (III) and As (V) making the biochars more effective as adsorbents for both As (III) and As (V).

Soil solution study of incubated arsenic-rich Histosol amended with biochar was conducted to evaluate the effects of EFB biochar and RH biochar on water-soluble As naturally present in Histosol (Figure 4) (Sari *et al*., 2014). Empty fruit bunch and RH biochars exhibit important feature as adsorbent with the porous structure and alkaline properties. The sorption experiment has shown the potential of these biochars to immobilize As in the soil system (Sari *et al*., 2014). The decreased of As concentration and increased of soil pH in soil solution study indicate the ability of biochar to reduce the phytoavailable As in contaminated soil (Figure 5).

The pot experiment was conducted to determine the optimum rates of biochars to reduce arsenic (As) uptake by sweet corn (Figure 6). Two types of biochars, EFB and RH with 5 rates (0, 2.5, 5, 10, 20 t C/ha) application were applied to 15 kg naturally contaminated soil in polybag. After 56 days of growth, biochar reduced the concentration of As in foliar tissue by 58% and 61% with the highest application of EFB and RH biochars at 20 t/ha C compared to the non-amended soil. The study shows the effectiveness of biochar in reducing the availability of As uptake by sweet corn as trace elements concentration decreased with increasing rate of biochar.

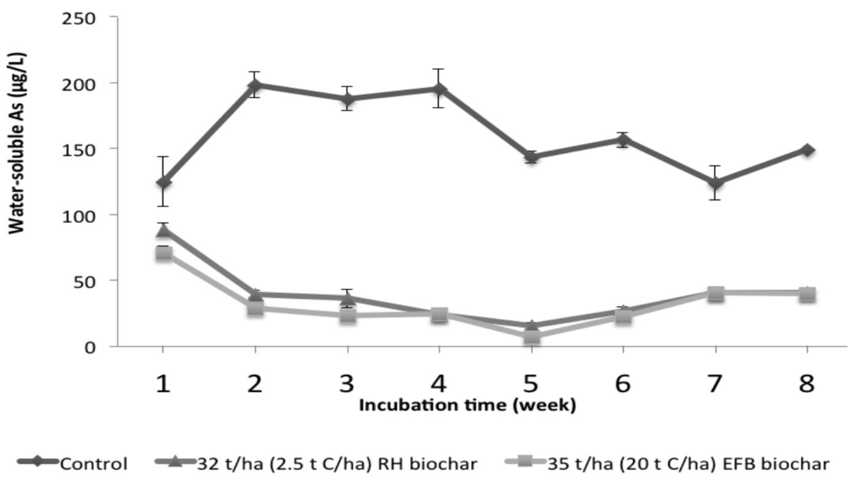


Figure 4 Effects of biochar on water-soluble arsenic in pore water

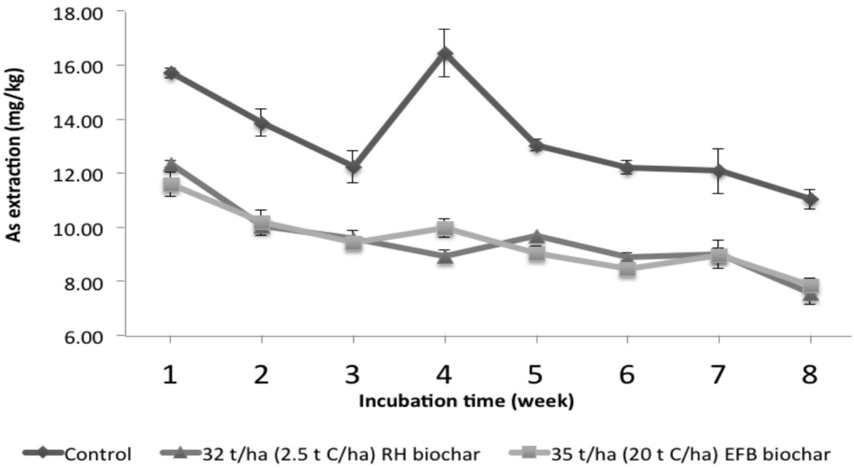


Figure 5 Effects of biochar on extractable arsenic in soil

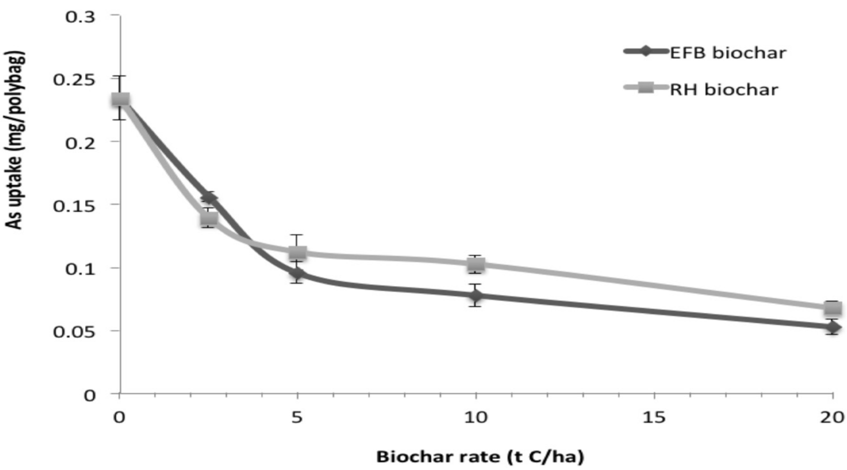


Figure 6 The uptake of arsenic by sweet corn plant after 56 days of planting

Industrial by-Products/Minerals

Agriculture will never be sustainable as long as soil organic matter levels are on a multi-year downward trend. Thus, soil organic matter must be restored to near original levels. Use of calcium helps build up the supply of soil organic matter in ways other than those for which calcium sources are usually added to soil. Calcium supplied as gypsum will be a major means for increasing the efficiency of accumulation of soil organic matter. The role of calcium in stabilizing soil organic matter needs more study. The mechanism that binds organic matter to clay particles in soil is polyvalent cations (Muneer and Oades, 1989). Efforts to increase levels of soil organic matter have overlooked this important phenomenon. Calcium is the most important cation for this purpose. Just how much calcium is necessary to bind organic matter to clay and how calcium relates to "slow" and passive soil organic matter need to be subjects for study. Iron is also a good bridging agent. Use of calcium and iron with organic matter could be the great secret for soil improvement.

Addition of industrial by-products to soil has gained importance recently as an alternative to remediate heavy metal contaminated soil (Table 4). For example, by-products from steel (iron oxides) and energy industries (ashes) will be assessed as an environmentally and resource-efficient option due to their alkalinity or acid neutralizing capacity and high specific surface area. The ANC is usually expressed as $CaCO_3$ equivalent and one of the most important factors used to evaluate the value of industrial by-products to be used as a liming agent on acidic soil.

Red Gypsum

Red gypsum (RG) is a waste material from the extraction of Ti for industrial purposes. Titanium is extracted from the mineral ilmenite ($FeTiO_3$) by sulfuric acid digestion. Red gypsum is produced by further increasing the pH of the effluent to about 5.0 by using calcitic limestone ($CaCO_3$), at which point the remaining sulfate precipitates along with iron oxides. The latter, derived from the iron contents of the ilmenite, are responsible for the red color of the material. Production process of red gypsum can be concluded by using these two equations as shown below.

First stage of reaction:

$$H_2SO_4 + CaCO_3 \rightarrow CaSO_4 + H_2O + CO_2$$

Second stage of reaction:

$$MSO_4 + Ca(OH)_2 \rightarrow M(OH)_2 + CaSO_4$$

(Sulphate metal)

Normally, this waste product is disposed off outside the titanium dioxide plant. Such byproducts might be suitable for use in agriculture in situations where mined gypsum has been used in the past. Red gypsum can be of great economic value due to its very high Ca and S content. Moreover, there are several reports on application of red gypsum as a soil amendment to immobilize As, Cd, Cu and Pb in heavy metal-contaminated soils (Lombi *et al.*, 2004; Illera *et al.*, 2004). In addition, the presence of the iron oxide responsible for the red color of RG might make it more effective as a soil amendment (Fauziah *et al.*, 1996) rather than as a source of Ca and S fertilizer. Dissolution of the gypsum and subsequent supply of sulfate S to crops might be affected by the presence of the oxides, which have the possibility to adsorb sulfate.

Table 4 Industrial byproducts amendments to sewage sludge (5%) treated soils (glasshouse study)

Amendments	Metal	Effects	Reference
Red Gypsum	Cu	Decreased sig. at > 10% level	Fauziah, *et al.* 2011
	Zn	Decreased significantly at > 5% level	
	Fe	Decreased sig. at 20% level	
	Cr	Decreased sig. at 2.5 to 10% level. At 20% highest Cr level – not sig. different from control	
Coal Fly Ash (CFA)	Cu	Decreased significantly at 5% - 10%, but increased sig. at 20%	Nur Hanani *et al.* 2010
	Zn	Decreased significantly at greater than 5% treatment	
	B	Increased significantly compared to control	
Water Treatment Residue (WTR)	Cu	Decreased significantly compared to control	Nur Hanani *et al.*, 2008
	Zn	Decreased significantly compared to control	

The RG was alkaline in nature, with a pH of 7.98 due to the presence of residual $CaCO_3$ in RG (Fauziah *et al.*, 1996). This property can be exploited to reduce the solubility and hence, phyto-availability, of some heavy metals in the soil system. The acid neutralizing capacity is the most important characteristic in evaluation of the value of the material as a liming agent. Red gypsum is not a good liming agent, with only 1.79% calcium carbonate equivalence. However, high rates of application (> 2.5%) can have significant influence on the pH of the soil system (Nur Hanani *et al.*, 2009). Iron oxide-rich gypsum by-products, including red gypsum (119 m^2 g^{-1}), have very large surface areas (Peacock and David, 2000). The surface area for the red gypsum (pulverized and sieved through 2.0 mm sieve size) was 39.8 m^2 g^{-1}. The high surface area plays a central role for adsorption behavior. Furthermore, the presence of Fe oxide can contribute to the co-existence of positive and negative charges on the variable charge oxide surface (Figure 7).

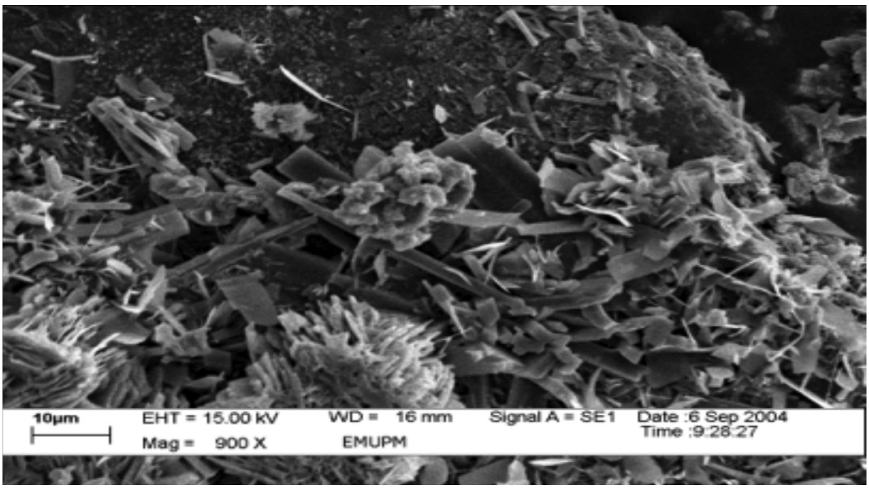


Figure 7 Fibrous Crystal Aggregates of Gypsum with Some Coatings of Iron Oxides

A column leaching study was conducted to investigate the red gypsum for in-situ immobilization of arsenic in the soil system. In this experiment, the treatment used was the different rates of red gypsum. The treatments were applied at the top soil only. The treatments were: T1 : no red gypsum (control),T2 : 25 t/ha red gypsum,T3 : 50 t/ha red gypsum,T4 : 100 t/ha red gypsum. From this study, red gypsum application has the potential to immobilize arsenic in the soil system and thus prevent arsenic from being taken up by the crop grown on arsenic contaminated soil. The presence of Fe in red gypsum can help surface adsorbed or co-precipitate As in the soil system (Figure 8 and 9).

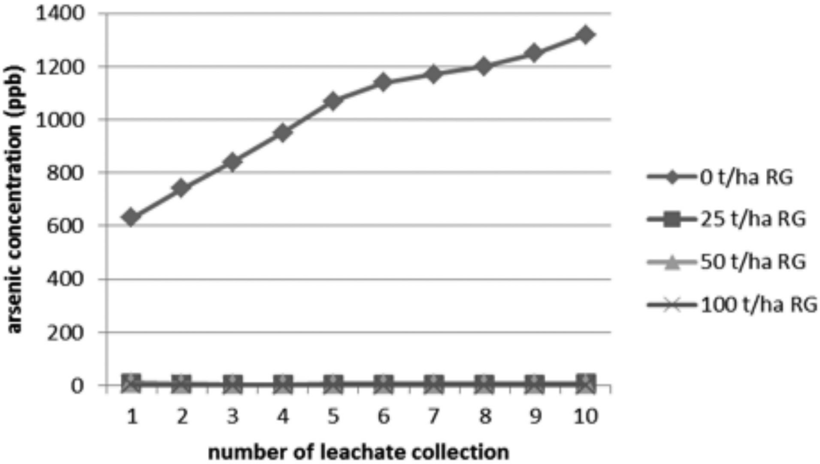


Figure 8 Arsenic concentration in each of the leachate collection (50 ml) up to one pore volume for each treatment.

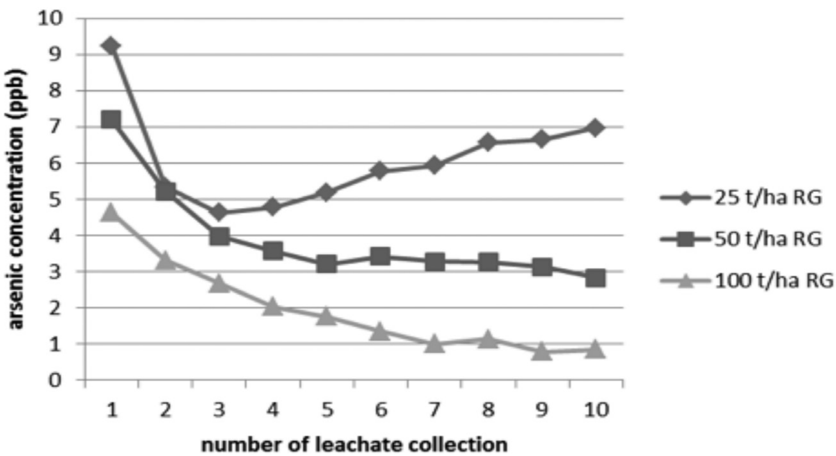


Figure 9 Arsenic concentration in each of the leachate collection (50 ml) for 25 t/ha, 50 t/ha and 100 t/ha red gypsum

In a soil incubation study, RG was applied to sewage sludge treated soil (Fauziah *et al*, 2011). Sewage sludge tend to have high concentrations of Cu and Zn and its application to soil increase these metals content. There seemed to be a lag-phase in the release of Zn from the RG minerals into the soil solution (Figure 10). The Zn concentrations in the soil solution started to increase only after 5 weeks of incubation for the treatments with low rates of RG application. The reason for this slow release of Zn into the soil solution is not known. However, this study demonstrated that increasing the RG amendment rates (5%, 10%, 20% and 40%) clearly reduced the Zn concentrations in soil solution after ten weeks of incubation. Thus, RG has the potential to fix Zn in the soil system and make it less phyto-available.

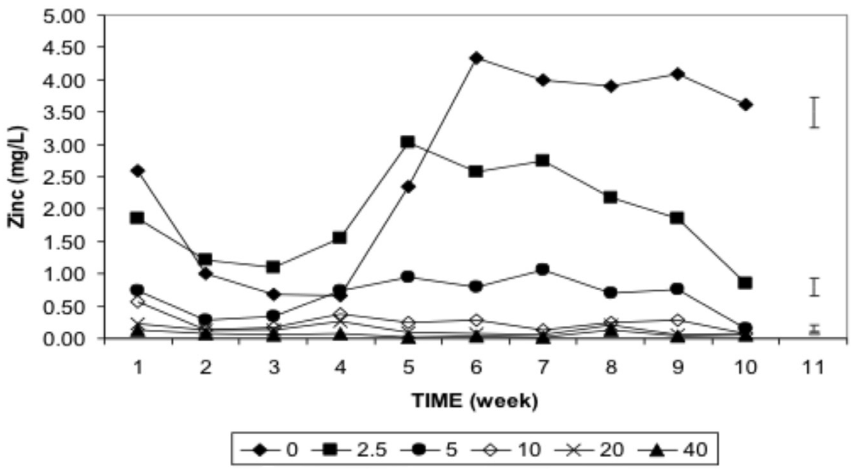
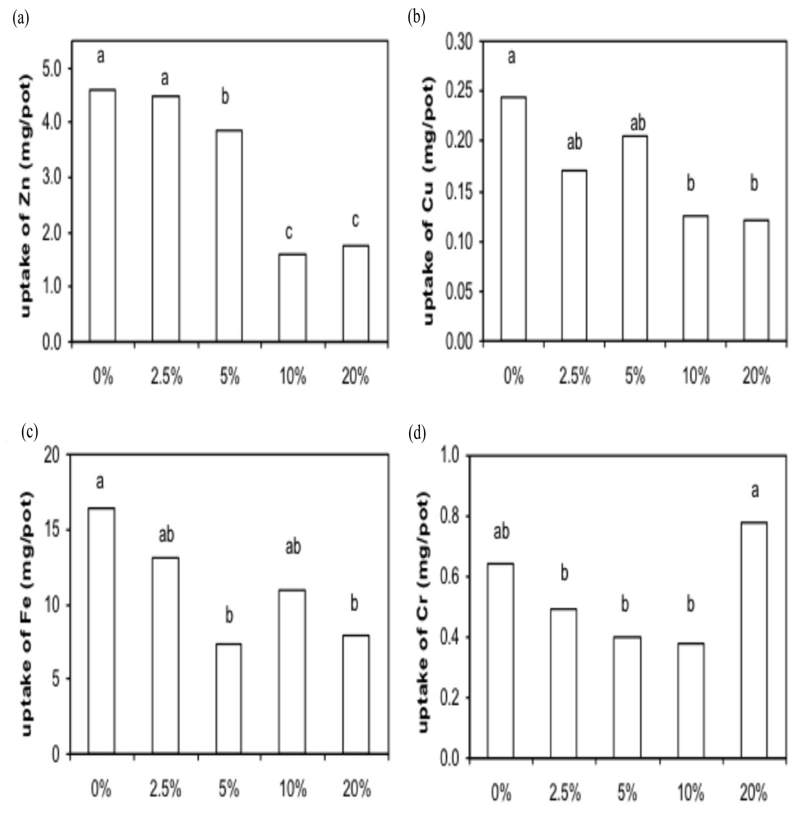


Figure 10 Soluble Zn at different rates of RG-contaminated soil treatments

Increasing the rate of red gypsum application resulted in decreasing uptake of Zn, Cu and Fe by the corn plants (Figure 11). This is due to the increase in soil pH. The residual alkalinity plus the buffer capacity of iron oxides (goethite and hematite) (Fauziah *et al.*, 1996), allow red gypsum to consume protons from an acid soil. However, the results for Cr seemed to be rather varied. The grasses grown on heavy metals contaminated soil remediated with red mud (a by-product of the bauxite industry) had high Cr concentrations (Zhao *et al.*, 2005; Snars *et al.*, 2004). Thus, there may be room for speculation that RG also contain high levels of Cr. However, the levels of Cr in red gypsum were found to be low. This study did not ascertain whether the organic matter in sludge alleviates the effect of excess Ca and Fe in the mixed soil system. This needs to be investigated. Furthermore, co-mixing two products such as RG and compost can turn the by-products into a more useful soil amendment as the amending capability of the by-product can be complemented and further enhanced by the co-mixed by-products (Fauziah *et al.*, 2011).



Similar letters above bars indicate that the values represented by the bars are not significantly different at the 1% level, according to the Duncan New Multiple Range Test (DMRT)

Figure 11 Uptake of microelements (mg/pot) using contaminated soil amended with red gypsum

A glasshouse study was then conducted with the same treatments as the soil incubation study using sweet corn as the test crop. Two set of experiments were established with 4 treatments and 4 replicates. Treatments of experiment are: Red Gyspsum + EFB Compost with different rate of red gypsum (2.5, 50, 100 and 200 t/ha) (Table 6). For Fe concentrations, application of RG+EFB compost show significant decrease in Fe concentrations in foliar tissues at the rate of 100 and 200 t/ha compared to the lower rates. The toxic level of Fe for corn is >350 mg/kg, thus there is no problem of Fe toxicity to plants in this case. For Zn concentration, significant decrease in Zn concentration were found at the rates of 100 and 200 t/ha compared to the lowest rate of RG. For Cd, significant decreased in Cd concentrations with the increasing rates of RG+EFB compost found at the rates of 100 t/ha and 200 t/ha compared to the lowest rate of RG+EFB compost used.

Table 6 Effects of treatments on heavy metals in foliar tissues

Treatment	mg/kg			
	Fe	Zn	Cd	Cr
RG 2.5t/ha + EFB Compost	305 ab	83.75 a	0.16 ab	0.98 ab
RG 50t/ha + EFB Compost	477 ab	75.65 a	0.08 ab	0.07 b
RG 100t/ha + EFB Compost	237 bc	39.20 b	0.03 b	0.14 ab
RG 200t/ha + EFB Compost	94 bc	9.63 b	0.05 b	0.06 b

Mean having the same letters within column are not significantly different at p>0.05

Co-application of red gypsum amended soil with biosolids was carried out in field condition with 5 treatments and 4 replicates at Lanchang, Pahang. Sweet corn (*Zea mays L.)* was used as the test crop. Rate of red gypsum used was 100 tonnes/ha combination with EFB compost, EFB biochar and chicken dung to the ratio 1:4 volume/volume basis. Co-application of RG with biosolids shows a decrease in Cd concentration compared to the control. The Cd concentration is above the 95th percentile; nevertheless, it is below the maximum allowable limit of the Canadian soil regulation. For Cr and Ni concentrations, application of RG:EFB compost shows a significant decrease in Cr and Ni concentrations compared to the control. For Pb concentrations, application of RG: EFB compost shows significant decrease in Pb concentrations compared to the control and other treatments, Application of RG and biochar shows significantly higher Pb concentration compared to the control and other treatments, chromium, Ni and Pb are below the 95th percentile or investigation level and also below the maximum allowable limit of the Canadian soil regulation. In conclusion, application of RG and EFB compost shows the best co-application on soil towards the corn growth based on the significant effect on the uptake of metal(loid)s.

Table 7 Co-application of red gypsum amended soil with biosolids (field study)

Treatment	mg/kg			
	Cd	Cr	Ni	Pb
Control	3.81 a	1.93a	0.22a	3.49a
RG only	3.79 ab	1.59ab	0.19a	3.79ab
RG: EFB Compost	2.74 c	0.21c	0.05b	0.62d
RG: Biochar	3.20b	1.35ab	0.26a	3.92a
RG: Chicken Dung	3.17b	1.44ab	0.22a	2.73c
95th percentile- Investigation Level	0.30	60	45	65
Maximum Allowable Limit (Canada)	8	75	100	200

Mean having the same letters within column are not significantly different at p>0.05

Coal Fly Ash

Coal fly ash (CFA) is an amorphous aluminosilicate material, a by-product of coal combustion and is composed of particulate matter collected from flue gas stream. Coal is one of the alternative natural resources used for the production of electricity in Malaysia. The increase use of coal for electric power generation will generate large quantities of CFA. Kapar power station in Selangor, Malaysia, produced around 200 Mg CFA per day. Currently, only 20% of the CFA is utilized as a component in the cement mixture, the rest is left stacked within the vicinity of the power plant.

Coal is known to contain every naturally occurring element, and therefore, it is not surprising that CFA can have beneficial effect on solving certain problem of soil quality. Use of CFA as a soil amendment is hindered by the lack of macronutrients in the ash and also concern on its high concentration of microelements, especially boron. The CFA is an alkaline residue produced during the burning coal for the generation of electricity which is enriched with CaO and MgO and has a pH around 8 to 12. The pH of CFA can vary depending on the S contents of the coal source, with high S generally producing acidic material and low S producing alkaline material (Adriano *et al.*, 2002). The pH of CFA used in this study was 8.34. In some cases, alkaline agent was used as a stabilization agent for contaminated soil to reduce pathogen and heavy metals availability (McGrath *et al.*, 1995; Zhang *et al.*, 2007).

Large surface area determination of CFA was probably due to large number of spongy irregular carbon-rich particles of unburnt coal (Fauziah, 1993). Surface area determination for this CFA was 7.5 m^2/g. Hence, the particle size distribution will provide information relating to land application of the ash, in term of trace elements solubility and effect on soil physical properties (Figure 12).

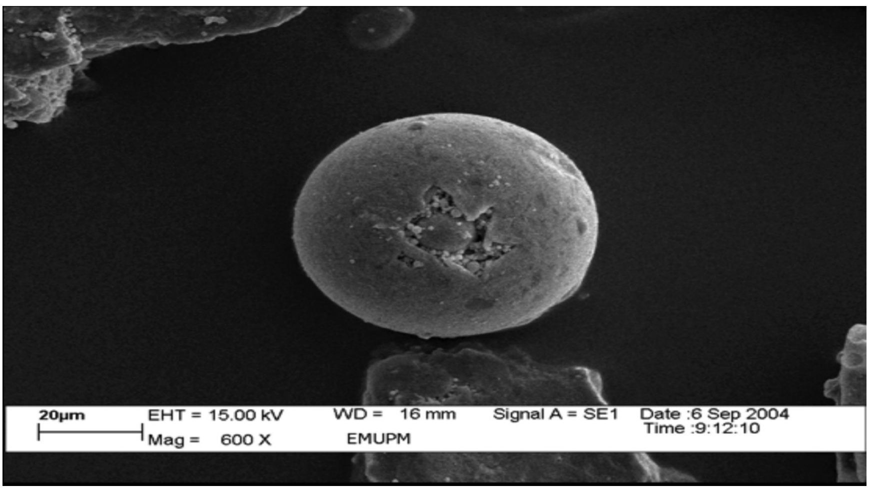


Figure 12 Cenophere Shape of Coal Fly Ash

The neutralization of acid by CFA is a relatively slow process that mainly involves the particle surfaces (Wong *et al.*, 2002).The CFA was not a good liming agent, with only 0.504% $CaCO_3$ equivalent (CCE). Based on the low level of Ca, this CFA it is considered only as a Class F fly ash (Bilski *et al.*, 1995).Therefore, considerably large quantities of this CFA compared to lime will be required to raise the pH of soil to some target level.

A soil incubation study was also conducted whereby, CFA was applied on sewage sludge treated soil (Nur Hanani *et al*., 2010). Increasing the CFA amendment rates clearly reduced the Zn concentrations in soil solution for the ten weeks of incubation (Figure 13). The reduced concentration of Zn probably can be explained by the higher adsorption and precipitation of Zn with an increase in pH (Sims and Kline, 1991; Jackson *et al*., 1999). The control treatment (0% CFA) had Zn concentrations in the soil solution which ranged from 1.47 mg L^{-1} to 0.67 mg L^{-1} for the four weeks of incubation and increased drastically at week five to 5.0 mg L^{-1}. It is not known with certainty why there was a delayed dissolution of Zn from the sewage sludge. The Zn concentration for the control treatment after week five until week ten of the incubation was still high (> 3.73 mg L^{-1}) compared to other treatments. Treatments using 2.5% and 5% CFA ranged less than 2.15 mg L^{-1} whereas treatments using greater than 10% CFA had the lowest Zn concentrations which were less than 1 mg L^{-1}. This indicates that CFA was feasible as a stabilization agent to reduce heavy metal toxicity in the sewage sludge-treated soil.

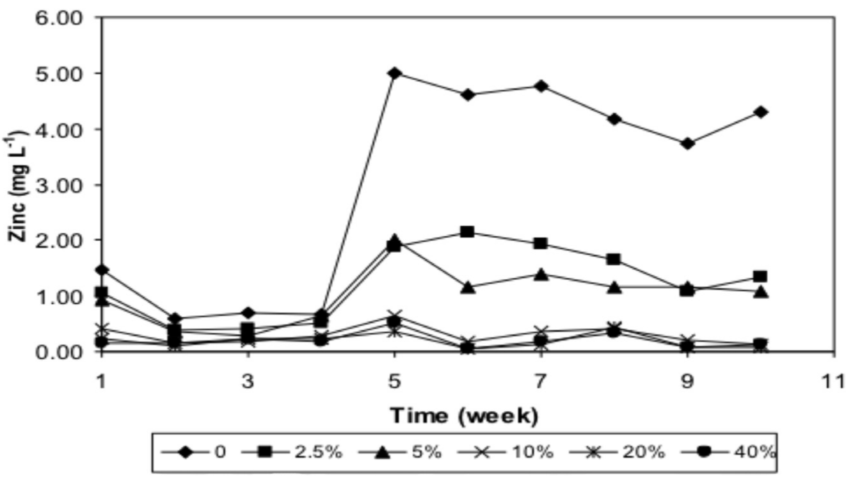
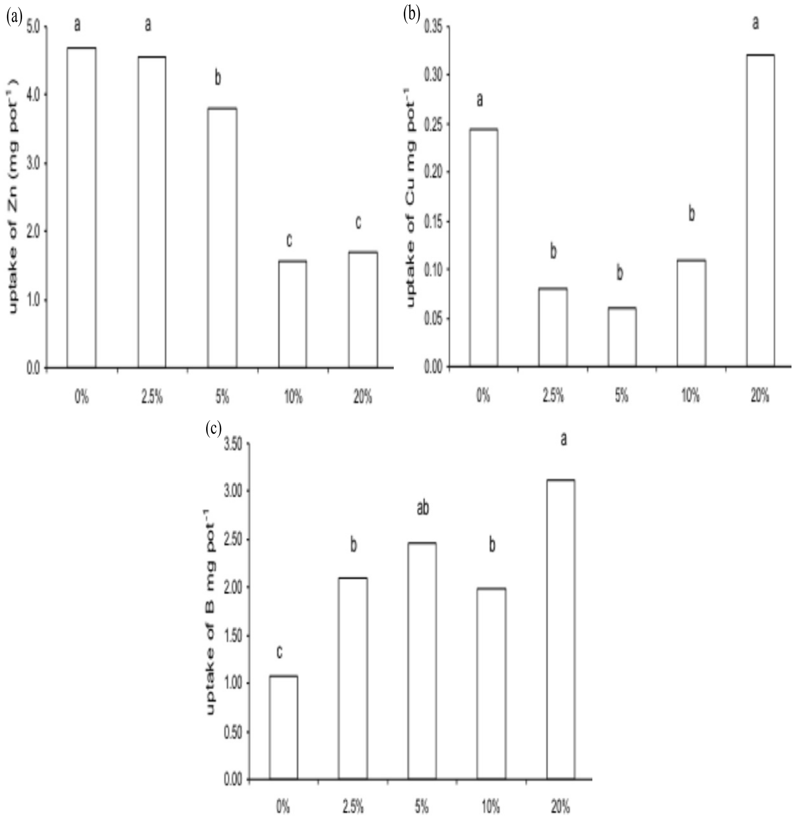


Figure 13 Soluble Zn at different rates of CFA-contaminated soil treatments

The Zn uptake by maize for treatment using CFA is shown in Figure. Overall, the concentration of Zn uptake by maize significantly decreased at higher rates of CFA treatments. Usage of 2.5% CFA did not show any significant result as compared to the control treatment. However, addition of more than 5% CFA significantly reduced Zn concentration in maize. This showed that the CFA was useful as a soil amendment to fix Zn in the contaminated soil.

The results showed application of CFA up to 10% reduced Cu uptake by the maize plants compared to the control (Figure 14). However, there was no significant difference in Cu uptake by maize between the control and the 20% CFA treatment indicating that CFA can be beneficial as a soil amendment to reduce Cu uptake by plant but, the amount of CFA should be applied at a proper rate to avoid Cu toxicity (Nur Hanani *et al.*, 2010).



Note: Similar letters above the bars indicate that they are not significantly different at 99% confidence level, according to the Duncan New Multiple Range Test (DMRT)

Figure 14 Uptake of heavy metals (mg/pot) using soil amended with CFA

Drinking-water Treatment Residues

In Malaysia, a low-cost and potentially effective substitute for remediation could be drinking-water treatment residues (WTRs). The pH for WTR for this study was close to being neutral just as stated in the reports of Gallimore *et al*., (1999), Ippolito *et al*., (2000) and Elliot *et al*., (1990) . The WTR has a pH of 7.07, the mineral present in WTR, such as kaolinite, gibbsite and Fe-oxides, provide surfaces for the adsorption of heavy metals. Value for WTR surface area was 28.3 m^2/g and this value was largely dependent on the size of the sample which was less than 2 mm due to the grinding process. Surface area determination can be used to estimate the amount of surface sites available for surface complexation reaction (Figure 15). Butkus (1998) reported a surface area of WTR was 10 m^2/g. Dzombak and Morel (1990) estimated that WTR can bind with protons, cations and anions based on the range of sorption maxima reported from 160 m^2/g to 600 m^2/g. The ANC of WTR was 0.504% CCE. Thus, WTR cannot be considered a good liming material compared to the pure $CaCO_3$, but perhaps usage at high rates of this material can still increase the pH of acidic soil.

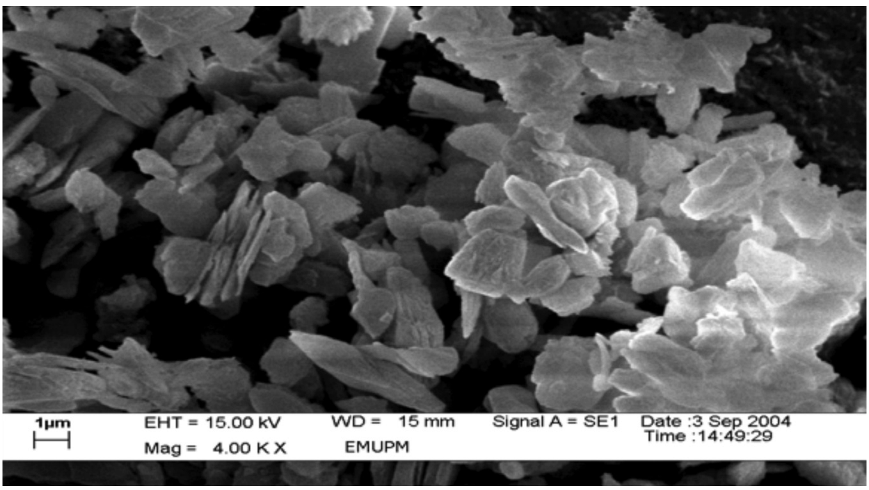


Figure 15 Presence of Kaolinite (Hexagonal Shape) and Illite Flakes of WTR

Also, in an incubation study of sewage sludge treated soil, treatment using the highest rates of WTR (40%) gave the lowest Zn concentration in the soil solution (Figure 16) (Nur Hanani *et al*, 2008). Perhaps, the high Zn concentrations and high pHs at the higher WTR rates led to low solubility of Zn due to the pH effect and also the phenomenon call ageing (Lock and Janssen, 2003). The trend of Zn solubility indicates slow dissolution of Zn minerals at the initial stage, and then the concentration dropped again due to the precipitation or ageing affect. Zinc concentrations were found to be low in all treatments using different rates of WTR (2.5, 5, 10, 20, and 40%) compared to the control (0% WTR). Addition of WTR did reduce the release of Zn from the sewage sludge. Therefore, WTR can be considered to be a potential soil amendment to fix Zn in contaminated soils.

Addition of WTR significantly reduced Zn uptake by corn plants compared to the control (Figure 17). This results show that the usage of WTR mixed with sewage sludge can significantly reduce the Zn uptake by corn. The major effect of high pH was to reduce the solubility of all micronutrients, especially Zn. Meanwhile, addition of more than 5% WTR, significantly reduced Cu uptake compared to the control. This results show that the usage of more than 5% WTR in sewage-sludge-amended soil can significantly reduce the Cu uptake by corn (Nur Hanani *et al*., 2008).

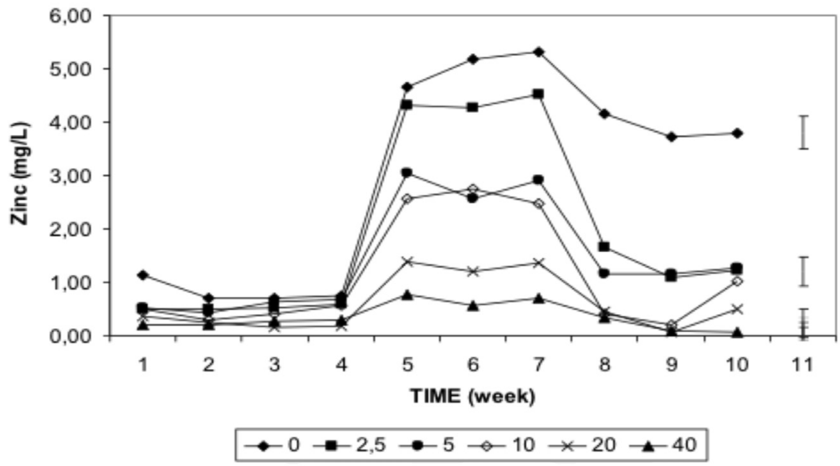
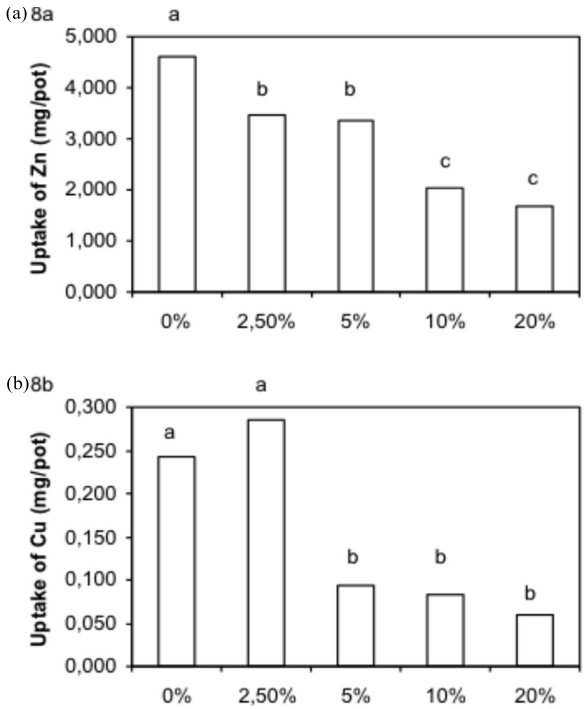


Figure 16 Soluble Zn at different rates of WTR-contaminated soil treatments



Similar letters above bars indicate that the values represented by the bars are not significantly different at the 1% level, according to the Duncan New Multiple Range Test (DMRT)

Figure 17 Uptake of heavy metals, Zn and Cu (mg/pot) using soil amended with WTR

Phosphate Materials

Phosphate compounds enhance the immobilization of metal(loids)s in soils through various processes including direct metal(loid) adsorption/ substitution by P compounds, P anion-induced meal(loid) adsorption, ad precipitation of metal(loid)s with solution P as metal(loid) phosphates. Depending on the source, soil application of P compounds can cause direct adsorption of metal(loid)s onto these compound through surface charge and enhanced anion-induced metal(loid) adsorption.

Phosphate rock (PR) belongs mainly to sedimentary, slightly to igneous and negligibly to metamorphic rocks. Eight percent of the world PR production is derived from the deposits of sedimentary marine origin, some 17% is derived from igneous rocks and their derivatives and the remainder comes from residual sedimentary deposits. About 90% of these deposits are used as raw materials for the manufacturing of phosphate fertilizer.

The dissolution of PR may be expressed by the equation;

$$\underset{\text{(Phosphate rock)}}{Ca_{10}(PO_4)_6F_2 + 12H_2O} \rightarrow \underset{\text{(Dissociation products)}}{10Ca^{2+} + 6H_2PO_4^- + 2F^- + 12OH^-}$$

Although the above reaction is for a fluorapatite PR, it applies to other members of the apatite minerals including reactive PR. As indicated in the above equation, the dissolution of PR results in the release of hydroxyl ions into the solution. Neutralization of the hydroxyl ions released by soil acidity enables the PR dissolution process to continue. Thus, an adequate supply of hydrogen ions is of primary importance for the continued dissolution of PR. Soil pH shows the magnitude of hydrogen ion supply, thus the use of PR depends on its reactivity and generally recommended for soils with a pH of 5.5 or less (Corley and Tinker, 2003).

Precipitation as metal(loid)-P has been proven as one of the main mechanism for the immobilization of metals such as Pb and Zn in soils. These fairly stable metal-P compounds have extremely low suitability over wide pH range, which make P application as attractive technology for managing metal(loid)- contaminated soils. Thus, application of apatite as amendment appears to be a promising soil additive for immobilizing metals in polluted soils (Soltan *et al.* 2012). Its effectiveness has been reported in several studies to reduce Pb uptake by plants (Cotter-Howells and Caporn 1996; Laperche *et al.* 1997; Hettiarachchi *et al.* 2000).

A glasshouse study has been conducted to assess the effectiveness of Pb immobilization due to chicken manure application using different sources of phosphate materials; bone meal (BM), Egyptian rock phosphate (ERP) and triple super phosphate (TSP) (Naim *et al.*, 2017). From the fractionation of glasshouse study, the percentage of exchangeable fraction of Pb was reduced with application of P-amendments with the highest of 20.2% of reduction recorded for 2 t/ha application of TSP. This is followed by reduction in exchangeable fraction for others treatments: 2 t/ha of BM (4.1%), 4 t/ha of BM (5.1%), 1 t/ha of ERP (8.1%) and 2 t/ha of ERP at 17.6%. These treatments were recorded as being able to stabilize the Pb as indicated in the percentage reduction of phytoavailable pools into a more stable form of residual pool (Figure 18).

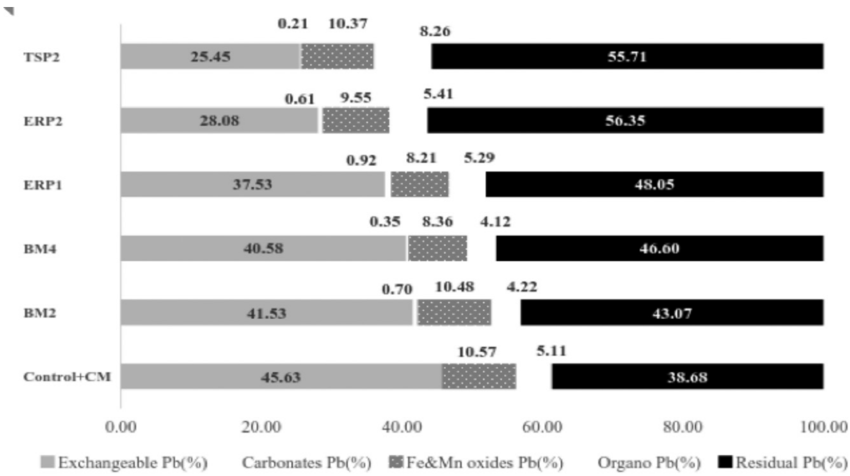
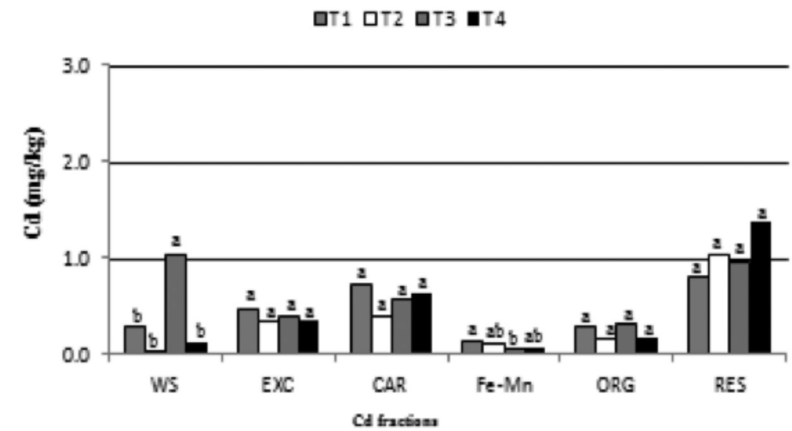


Figure 18 The lead content expressed as percentage in 5 fractions; 1) exchangeable, 2) carbonates, 3) Fe & Mn oxides, 4) organic matter and 5) residual with application of different P-amendments.

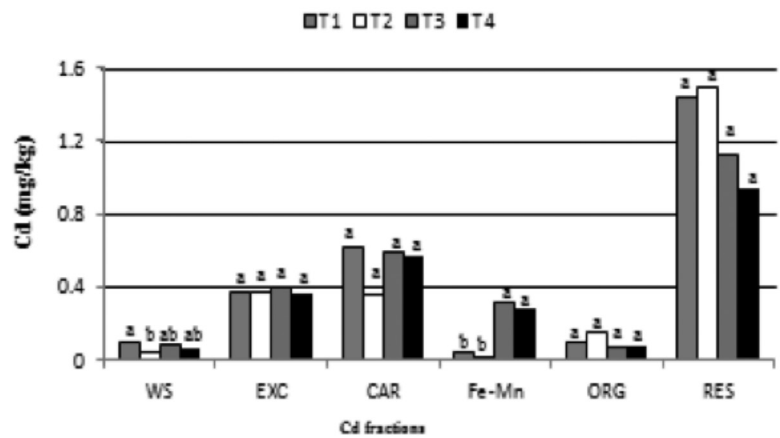
Another glasshouse study was conducted to determine whether lime and POME amendment at 4 different rates can help reduced Cd uptake by oil palm seedlings using Gafsa Phosphate Rock as source of P fertilizer (Aini Azura *et al.*, 2012).

Segamat Series amended with POME showed significant differences ($p<0.05$) of water soluble and Fe-Mn fractions between four rates of treatment (Fig. 5.19). This treatment decreased the exchangeable fraction whilst increasing the residual fraction even though the data were not statistically different. Meanwhile, cadmium in Segamat Series amended with lime was highest in the residual fraction followed by the exchangeable, carbonate, Fe-Mn, organic and lastly water soluble fractions (Fig. 5.20).



Letter with the same alphabet on the bars within the same soil fractions are not significantly different at p>0.05. (Comparison made within rates of treatment)

Figure 19 Concentrations of Cd fractions in Segamat Series amended with four rates of POME



Letter with the same alphabet on the bars within the same soil fractions are not significantly different at p>0.05. (Comparison made within rates of treatment)

Figure 20 Concentrations of Cd fractions in Segamat Series amended with four rates of lime

Issues Pertaining to Immobilization Technique

A major issue associated with immobilization technique is that although metal(loid)s become less available for plant uptake, their total concentrations in soils remain unchanged unless transported out of the soil system through leaching, colloid facilitated transport through surface runoffs, etc. The immobilized metal(loid)s may become plant available with time through natural weathering processes through the breakdown of high molecular weight metal(loid)s complexes or change in soil condition such as under waterlogged condition.

Most of studies on immobilization of metal(loid)s are conducted on laboratory or glasshouse scale. More field studies are required to demonstrate the values of a range of immobilizing soil amendments to remediate contaminated soils. These field studies need to also examine the impact on the presence of co-contaminants and the long-term effectiveness of using soil amendments should be investigated.

Conclusion

There is great potential of practising in-situ immobilization technique using soil amendments on agricultural soils that are high in metal(loid)s content. These amendments can reduce metal(loid)s uptake by the crops to below maximum permitted concentrations as gazetted in the Malaysian Food Act (1983) and Food Regulation (1985) and thus considered safe for human consumption.

References

Adriano, D.C. 1986. Trace Elements in the Terrestrial Environment. New York: Springer-Verlag.

Adriano, D.C., Weber, J., Bolan, N.S., Paramasivam, S., Bon-Jun Koo, and Sajwan, K.S. 2002. Effects of high rates of coal fly ash on soil, turfgrass and groundwater quality. *Water, Air and Soil Pollution.* 139: 365-385.

Aini Azura, A., Fauziah, C.I. and Samsuri, A.W. 2012. Cadmium and zinc concentrations in soils and oilpalm tissues as affected by long term application of phosphate rock fertilizers. *Soil and Sediment Contamination*. 21(5): 586-603.

Alva, A.K., Asher, C.J. and Edwards, D.G. 1986. The role of calcium in alleviating aluminum toxicity. *Aust. J. Soil Res.* 37: 375-383.

Bagreev, A., Bandosz, T.J. and Locke D.C. 2001. Pore structure and surface chemistry of adsorbents obtained by pyrolysis of sewage-derived fertilizer. *Carbon.* 39: 1971-1979.

Basso, A.S., Miguez, F.E., Laird, D.A., Horton, R., and Westgate, M. 2013. Assessing potential of biochar for increasing water-holding capacity of sandy soils. *GCB Bioenergy.* 5: 132-143.

Beesley, L., and Marmiroli, M. 2011. The immobilisation and retention of soluble arsenic, cadmium and zinc by biochar. Environ. Pollution. 159:474-480.

Boateng, A.A. 2007. Characterization and thermal conversion of charcoal derived from fluidized-bed fast pyrolysis oil production of switch grass. *Industrial Engineering and Chemical Research.* 46: 8857-8862.

Bilski, J.J., Alva, K., and Sajwan, K.S. 1995. Fly Ash. In: Rechcigl, J.E. (Editor). Soil Amendments and Environmental Quality. Boca Raton, Florida: Lewis Publisher.

Butkus, M.A. 1998. Surface complexation modeling on phosphate adsorption by water treatment residue. *Journal of Environ. Qual*. 27:1055-1063.

Casagrande, J.C., Soares, M.R., and Mouta, E.R. 2008. Zinc adsorption in highly weathered soils. *Brasília.* 43:131-139.

Dzombak, D.A., and Morel, F.M.M. (1990). Surface Complextion Modelling. Hydrous Ferric Oxide. New York: John Wiley and Sons.

Elliot, H.A., Dempsy, B.A., Hamilton, D.W., and De Wolf JR. 1990. Land Application of Water Treatment Sludges: Impacts. Denver, Co: Am. Water Works Assoc. Res. Foundation.

Fauziah, C.I. (1993). Characterization and Trace Contaminant Chemistry of Coal Combustion By-products. PhD Thesis, University of Georgia, Athens.

Fauziah, C.I., Nur Hanani, M., Zauyah, S, Samsuri, A.W., and Rosazlin, A. 2011. Co-application of red gypsum and sewage sludge on acidic tropical soils. *Communications in Soil Science and Plant Analysis.* 42(21): 2561-2571.

Fauziah, C.I., Zauyah, S., and Jamal, T. 1996. Characterization and land application of red gypsum: A waste product from the titanium dioxide industry. *Sci. Total Environ.* 188: 243 –251.

Gallimore, L.E., Basta, N.T., Storm, D.E., Payton, M.E., Huhnke, R.H., and Smolen, M.D. 1999. Water treatment residue to reduce nutrients in surface runoff from agricultural land. *J. Environ. Qual.*, 28:1474-1478.

Glaser, B., Lehmann, J., and Zech, W. 2002. Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal: A review. *Biology and Fertility of Soils*. 35: 219-230.

Illera, V., Garrido, F., Serrano, S., and Garcia-Gonzalez, M.T. 2004. Immobilization of the heavy metals Cd, Cu and Pb in an acid soil amended with gypsum- and lime-rich industrial by-products. *European J. of Soil Sci.* 55: 135-145.

Ippolito, J.A., Barbarick, K.A., and Rendte, E.F. 2000. Co-application of water treatment residues and biosolids on two range grasses. *Journal Environ. Qual.* 28:1644-1650.

Ismail, H., Shamshuddin, J., and Syed Omar, S.R. 1993. Alleviation of soil acidity in a Malaysian Ultisol and Oxisol for corn growth. *Plant Soil*. 151: 55-65.

Jackson, B.P., Miller, W.P., Schumann, A.W., and Sumner, M.E. 1999. Trace element solubility from land application of fly ash/ organic waste mixtures. *J. Environ. Qual.* 28:639-647.

Lua, A.C., Yang, T. and Guo, J. 2004. Effects of pyrolysis conditions on the properties of activated carbons prepared from pistachio-nut shells. *Journal of Analytical and Applied Pyrolysis.* 72: 279-287.

Luizão, F.J., Petersen, J, and Neves, E.G. 2006. Black carbon increases cation exchange capacity in soils. Soil Science Society of America Journal. 70: 1719-1730.

Kumpiene, J., Lagerkvist, A, and Maurice, C. 2008. Stabilization of As, Cr, Cu, Pb and Zn in soil using amendments: A review. *Waste Management.* 28: 215-225.

Lock, K., and Janssen, C.R. 2003. Influence of ageing on zinc bioavailability in soils. *Environ Pollution.* 126: 371-374.

Lombi, E. Hamom, R.E. Wieshammer, G., McLaughlin, M.J., and McGrath, S.P. 2004. Assessment of the use of industrial by-products to remediate a copper- and arsenic-contaminated soil. *J. Environ. Qual.* 33: 902-910.

McGrath, S.P., Chaudri, A.M., and Giller, K.E. 1995. Long-term effects of metals in sewage sludge on soils, microorganisms and plants. *Journal of Industrial Microbiology*. 14: 94-104.

Naim, K.M., Fauziah, C.I. and Rosenani, A.B. 2017. In-situ immobilization of lead using different source of phosphate amendments for the organic production of misai kucing (*Orthosiphon stamineus*). *Soil Environ*. 36(1): 20-27.

Nur Hanani, M., Che Fauziah, I., Samsuri, A.W., and Zauyah, S. 2010. Formulation of coal fly ash and sewage sludge mixtures to reduce impacts on the environment when used as soil ameliorant for acidic tropical soils. *Malaysian J. of Soil Sci.* 14(1): 53-70.

Nur Hanani, M, Che Fauziah, I., Samsuri, A.W., and Zauyah, S. 2008. Utilization of drinking-water treatment residue to immobilize copper and zinc in sewage-sludge-amended soils. *Land Contamination and Reclamation.* 16(4): 319-332.

Pavithra, S. 2011. Evaluation of Thermochemical Decomposition of Various Lignocellulosic Biomasses for Biochar Production, Master Thesis. McGill University, Ste Anne De Bellevue, Quebec, Canada.

Peacock, S., and David, R.L. 2000. The suitability of an iron oxide- rich gypsum byproduct as a soil amendment. *J. Environ. Qual*. 29: 1969-1975.

Rees, F., Simonnot, M.O., and Morel, J.L. 2013. Short-term effects of biochar on soil heavy metal mobility are controlled by intra-particle diffusion and soil pH increase. European journal of Soil Science. 65 (1): 149-161.

Rouquerol, F., Rouquerol, I., and Sing, K. 1999. Adsorption by Powders and Porous Solids. London, UK: Academic Press.

Saito, M. and Marumoto, T. 2002. Inoculation with arbuscilar mycorrhizal fungi: The status quo in Japan and the future prospects. Plant and Soil. 244: 273-279.

Samsuri, A.W., Sadegh-Zadeh, F, and She-Bardan, B.J. 2013. Adsorption of As(III) and As(V) by Fe coated biochars and biochars produced from empty fruit bunch and rice husk. Journal of Environmental Chemical Engineering. 1 (4): 981-988.

Sari, N.A., C.F. Ishak and Bakar, R.A. 2014. Characterization of oil palm empty fruit bunch and rice husk biochars and their potential to adsorb arsenic and cadmium. Am. J. Agric. Biol. Sci. 9: 450-456.

Sastre, J., Rauret, G. and Vidal, M. 2006. Effect of the cationic composition of sorption solution on the quantification of sorption-desorption parameters of heavy metals in soils. Environmental Pollution. 140: 322-339.

Shamshuddin, J., Che Fauziah, I., and Sharifuddin, H.A.H. 1991. Effects of limestone and gypsum application to a Malaysian ultisol on soil solution composition and yields of maize and groundnut. Plant and Soil. 134: 45-52.

Shamshuddin, J. Jamilah, I., Sharifuddin, H.A. H., and Bell, L.C. 1992. Changes in solid phase properties of acid soils as affected by limestone, gypsum, palm oil mill effluent and rock phosphate. Pertanika. 15(2): 105-114.

Shamshuddin, J. and Ismail, H. 1995. Reactions of ground magnesium limestone and gypsum in soils with variable-charge minerals. Soil Sci. Soc. Am. J. 59: 106-112.

Shamshuddin, J., Fauziah, C.I., and Bell, L.C. 2009. Soil solution properties and yield of corn and groundnut grown on Ultisols as affected by dolomitic limestone and gypsum applications. Malaysian. J. Soil Sci. 13: 1-12.

Sims, J.T., and Kline, J.S. 1991. Chemical fractionation and plant uptake of heavy metals in soil amended with composed sewage sludge. J. Environ. Qual. 13:344-349.

Snars, K.E., Gilkes, R.J., and Wong, M.T.F. 2004. Use of red mud residue from bauxite processing to ameliorate soil acidity. In Proc. the 6th International Symposium on Plant-Soil Interactions at Low pH, Aug. 1-5, Sendai, Japan.

Troeh, F.R. and Thompson, L.M. 2005. Soils and Soil Fertility. Iowa, USA: Blackwell Publishing.

Wong, S.C., Li, X.D., Zhang, G., Qi, S.H., and Min, Y.S. 2002. Heavy metals in agricultural soils of the Pearl River Delta, South China. Environ. Pollut. 119:33-44.

Yuan, J.H. and Xu, R.K. 2011 Progress of the research on the properties of biochars and their influence on soil environmental functions. Ecological and Environmental Science. 20:779-785.

Zabaniotou, A., Stavropoulos, G. and Skoulou, V. 2008. Activated carbon from olive kernels in a two-stage process: Industrial improvement. Bioresource Technology 99: 320-326.

Zarcinas, A.B., Fauziah, C.I., McLaughlin, M.J., and Gill, C. 2004. Heavy metals in soils and crops in Southeast Asia. 1. Peninsular Malaysia. Environmental Geochemistry and Health. 26(4): 343-357.

Zhang, T., Walawender, W.P., Fan, L.T., Fan, M., Daugaard, D. and Brown, R.C. 2004. Preparation of activated carbon from forest and agricultural residues through CO_2 activation. Chemical Engineering Journal. 105: 53-59.

Zhang, H., Sun, L, and Sun, T. 2007. Solubility of ion and trace metals from stabilized sewage sludge by fly ash and alkaline mine tailing. Journal of Environmental Science. 20: 710--716.

Zhao, F.J., McGrath, S.P., Dennis, P.G., Dunham, S.J., and Gray, C.W. 2005. In-situ remediation of a heavy metal polluted soil using red mud. In Proc. 8th International Conference on the Biogeochemistry of Trace Elements, April 3-7, Adelaide, Australia.

3

Biochar - Soil Biological Interactions

Noraini Md Jaafar, Lynette K. Abbott, Zakaria M. Solaiman, and Radziah Othman

Introduction

Biochar, the solid product of pyrolysis process, has gained considerable attention as an alternative soil amendment for soil sustainable land use. This carbon stable organic matter is produced from various feedstocks originating from plant and animal materials (Lehmann *et al.* 2011). The pyrolysis processes, including the temperature and duration time, transform the feedstocks to biochar with various particle sizes and porosity. Biochar is a relatively stable and porous form of organic matter compared with other forms of organic matter applied to soil. Further activation of biochar greatly enhances the porosity of biochar. Biochar has been advocated to overcome soil fertility problems and mitigate against global warming through carbon sequestration due to its long residence time in soil (Nichols *et al.*, 2000).

Biochar can play roles in nutrient retention in soil and reduce leaching (Ding *et al.*, 2010; Mukherjee and Lal, 2013). Retention of nutrients by a combination of biochar and clay was shown to be associated with reduced cumulative NH_4^+ leaching (Knowles *et al.*, 2011). Similarly, biochar addition to soil reduced nitrate leaching from biosolids amended soil (Dempster *et al.*, 2012). Other biochar benefits include amelioration effects on soil pH, increased cation-exchange capacity, water retention and improvement to soil biological properties (Solaiman *et al.*, 2010; Lehmann et. al., 2011). Several mechanisms have been proposed with regards to the beneficial effects of biochar on biological activity in soils and plant roots (Warnock *et al.*, 2007; Jaafar *et al.*, 2014; Jaafar *et al.*, 2015). For example, the porous structure and high surface area may have a role as a habitat for soil biotaincluding arbuscular mycorrhizal (AM) fungi (Warnock *et al.*, 2007). However, the fate of biochar once amended in soil is often overlooked. It is important to understand the interactions of biochar-soil biota over the longer-term.

Modification of the physical and chemical properties of biochar particles and the attachment of soil particles to biochar surface might alter habitat potential by biochar and microbial activity as well as AM fungal infectivity and effectiveness (Thies and Rillig, 2009; Lehmann et. al, 2011; Hammer at al., 2014). Ameliorative effects of biochar on soil biology have a close relationship with the way biochar is managed in soil. Management practices associated with use of biochar require consideration for their potential to provide beneficial contribution to soil fertility and sustainability.

Biochar Fate in Soils

Once biochar enters the soil, the particles undergo several changes following interactions with soil and biological activities. Soil particle accumulation in or on biochar and microbial attachment to biochar surfaces and within biochar pores may take place. Concurrently, soil fauna such as earthworm may ingest and break down the biochar particles and ameliorative effects of biochar may follow. Larger particles of biochar may gradually break into smaller particles over time. However, while biochar particles can undergo physical changes, the duration of this process is not well understood and may differ for biochars maded from of different forms of organic material.

Microbial attachment and colonization of biochar pores might alter soil microbial communities and their activities (Atkinson *et al.*, 2010; Sohi *et al.*, 2010). Other changes related to the use of biochar include the influence on soil chemical and physical properties. These changes in soil properties are mainly influenced by the amount and source of biochar. Soil pH and water retention due to biochar amendment may improve with biochar application, and this could be favorable to soil biota generally. Soil attached to biochar may facilitate the introduction of microorganisms, substrate, nutrients and water sources in or on biochar particles. In addition, biochar surfaces when exposed to air will gradually oxidize, promoting opportunities for soil microorganisms and root growth, leading to improved biological activities. The changes that occur to biochar surfaces may also influence water flow and nutrient retention in soils (Joseph *et al.*, 2010).

Biochar Influences on Soil Biology

Biochars have been shown to both positively or negatively influence the soil beneficial soil microorganisms, and the mechanisms may either be direct or indirect (Warnock *et al.*, 2007; Lehmann *et al.*, 2011). Among these mechanisms are: (i) biochar may protects soil organisms from predators via its porous nature which has potential to create a habitat for soil microorganisms, (ii) moderation in soil acidity through biochar addition can affect important soil properties by increasing soil pH and water retention, leading to better conditions for soil organisms, (iii) biochar interactions with soil fauna may stimulate soil microorganisms and soil physical properties, and (iv) indirect biochar effects can lead to soil modification changes including microbial enzyme and other processes that alter nutrients retention or release from soil (Warnock *et al.*, 2007). The mechanisms by which biochar might affect soil biology depend on biochar management practices, such as biochar placement and particle sizes (Jaafar, 2014). Biochar management factors and the growth of soil organisms and microbial related processes can be interconnected (Figure 1).

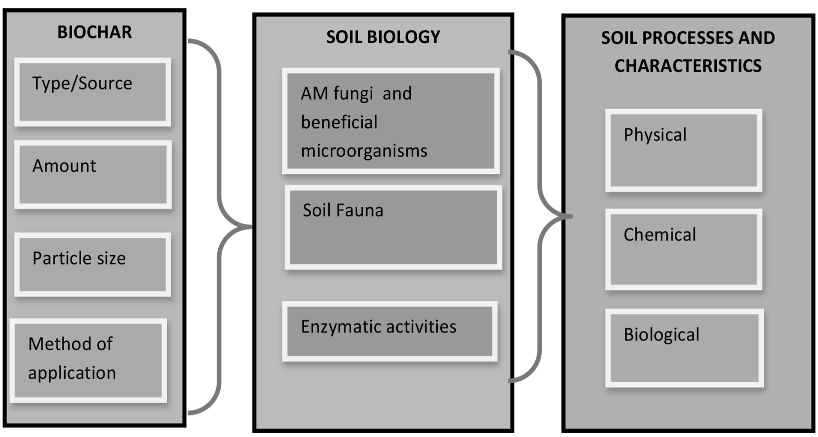


Figure 1 Examples of biochar factors and soil processes likely to influence soil health

Biochar Management for Soil Biological Improvement

Biochar management that includes selecting the optimum amount, source, and particle size for improved soil fertility must take into account its impact on soil biology. Understanding both biochar as well as its reactions in soil are necessary for attaining desired responses and balance to the soil system.

Biochar characteristics have been shown to be a driving factor for soil biological development and microbial population in soil (Thies and Rillig 2009; Rillig *et al*., 2010). The heterogeneity found in biochar are the results from from various sources and pyrolysis processes, which are the main factors associated with ameliorative effects of biochar on soil ecosystems (Gundale and DeLuca, 2006; Chan *et al*., 2008; Downie *et al*., 2009). In addition, biochar sources also determines the amount and methods of application for soil and plant benefits.

The physical features of biochar have been widely characterized for various biochars prepared using a range of pyrolysis parameters. The source of biochar feedstock contributes to the heterogeneity of biochar characteristics, which also occur even within one source. Pyrolysis processes and biochar sources create biochar with varying particle size fractions from large (>4 mm) to fine particles (Downie *et al*., 2009; Verheijen *et al*., 2009). The range in particle size may influence microbial attachment and faunal ingestion. Generally, it is expected that the larger the particle sizes have lower surface area than the finer particles. However, a study of pore properties of a woody biochar and 3 different particles sizes showed that external surface area did not differ much across 3 varying particle sizes 0.5-1.0, 1-2 mm to 2-4mm (Figure 2; Jaafar *et al*., (2015a)). Biochar pores quantified and characterized however, may not reflect the actual habitable spaces for microorganisms.

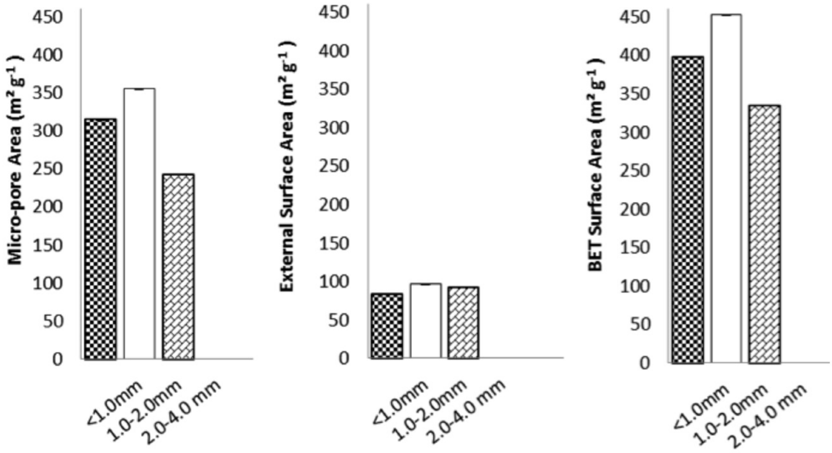


Figure 2 Biochar surface and pore characteristics on varying particle sizes among Wundowie (woody) biochar from jarrah (*Eucalyptus marginata*) and wandoo (*Eucalyptus wandoo*) (after Jaafar et.al., 2015a).

The rate of application of biochar to soil may influence biological activity and other soil processes. The most effective amount of biochar applied to soil is normally related to its chemical properties including nutrient concentrations and background soil characteristics such as texture and soil management practices (Yeboah *et al.*, 2009; Beesley *et al.*, 2010; Kimetu and Lehmann, 2010). The responses of soil microorganisms to increasing rate of biochar applied to soil vary from decreased microbial biomass (Dempster *et al.*, 2012), to trends of increased microbial activity (Rutigliano *et al.*, 2014) and AM fungi (Solaiman *et al.*, 2010). For example, Steiner *et al.* (2008) observed that microbial biomass and respiration increased when biochar was increased from 50 to 150 g kg^{-1} soil.

The other biochar management practice relevant to effects of biochar is its placement in soil. Biochar can be banded, thoroughly mixed with topsoil, or surface applied. Any effects of placement on soil microbiology may be due their access to biochar particles (Blackwell *et al.*, 2010; 2015; Solaiman *et al.*, 2010. Banded or homogenously spread biochar in soil systems may affect the microhabitat of soil organisms. However, experimental evidence on the effects of biochar placement on microbial activities is lacking.

A number of factors can influence interactions between soil and biochar (Table 1). However, there are mixed results from positive, no effects to negative effects on biochar management in soil on soil microbial abundance and development of microorganisms and soil fauna.

Table 1 Biochar management factors likely to influence soil biota

Biochar application	Effects on soil microorganisms
Biochar source	Positive, negative and no effect on microbial respiration, microbial C, arbuscular mycorrhizas (only root colonisation) as well as fauna
Application amount	Effective amount is soil specific (soil history, type, management). Lower amounts must be accompanied with other nutrient source (e.g. fertiliser or labile organic matter) for microbial activity and arbuscular mycorrhizal colonisation
Particle size	Effects on soil microorganisms in bulk soil is not known. The emphasis has been on microbial attachment in microsites
Application method	Microbial colonisation in deposited biochar (local or close proximity) versus bulk soil differ. Biochar dilution effects in soil and with plant roots are not known
Organic amendment with biochar	Previously focused on biochar priming effect; organic matter helped rate of biochar mineralisation

Mechanism Underlying Biochar – Soil Biology Interactions

Biochar as a habitat

Biochar has a high surface area which provides opportunities for microbial attachment, whereas the pore characteristics (high porosity, high numbers of larger or smaller pores) reflect the potential microbial habitat (Thies and Rillig, 2009; Jaafar *et al*., 2014). In terms of porous structure, activated biochar and woody biochar normally had higher porosity and pore properties than other organic derived biochar. Activated biochar also has high surface area (Steinbeiss *et al*., 2009; Ahiduzzaman and Islam, 2016). Both laboratory observations of fungi on biochar and those from biochar extracted from soils support the suitability of biochar as a refuge to soil biota (Hockaday *et al.* 2007; Ascough *et al.*, 2010). Biochar surfaces were colonized by fungal hyphae. Soil organisms can establish on biochar biochar surfaces and in exposed pores. Weaker physical structures such as cracks in biochar particles may also facilitate microbial colonization (Verheijen *et al.*, 2009; Ascough *et al.*, 2010).

Pore connectivity and pore size are likely to influence microbial colonization of biochar. Pores with diameter from 2– 80 μm normally found in wood were beneficial for fungi as well as bacteria (Thies and Rillig, 2009; Hammer *et al.*, 2014). These pore sizes would enable soil bacterial cells and fungal hyphae to enter internal spaces within biochar (Swift *et al.*, 1979). Jaafar *et al.* (2014) visually observed heterogeneous properties of biochar (Figure 3a) and found fungal attachment in biochar pores as well on biochar surfaces after retrieval from soil (Figure 3b). Attachment of soil particles onto biochar particles and soil particles were also observed (Jaafar *et al.*, 2015a). However, potential impacts of soil particles that clog biochar smaller pores (20 μm diameter) and cement biochar surfaces is not clear. This clogging could positively or negatively change the biochar pore and surface structure and overshadow the potential for biochar pores to provide microbial habitat. Adsorption of organic matter to biochar, thus blocking biochar pores can also decrease porosity and lead to competition for space, air and water by microorganisms (Kwon and Pignatello, 2005).

(a) (b)

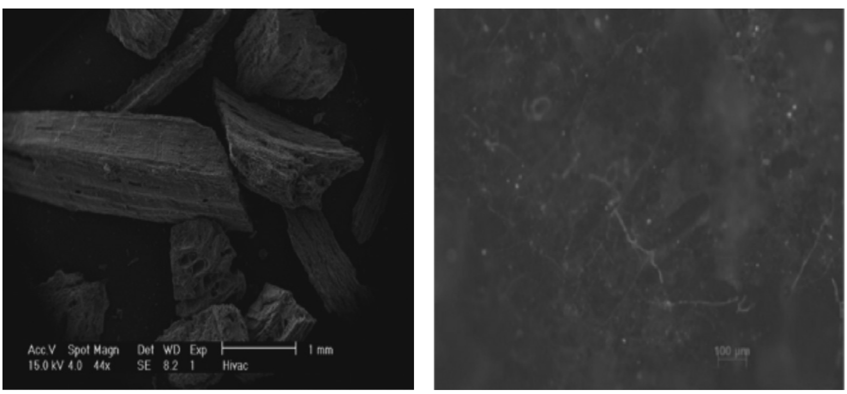


Figure 3 a) A SEM image of biochar heterogenous properties and b) fluorescence image of fungal hyphae attached onto biochar

The degree of microbial colonization in or on biochar may vary depending on the pores size. Larger pores (about 100 μm diameter) within biochar could be suitable for fungal habitat (Jaafar *et al*., 2014). Pyrogenic biochar has been shown to be colonized by fungi on both interior and external surfaces (Hockaday *et al*., 2007). Soil fungi residing in biochar pores and surfaces may obtain benefits, and this could lead to influences on plant growth. Hammer *et al*. (2014) showed how mycorrhizal fungi can improve phosphorus acquisition while colonizing biochar surfaces and microsites. Sporulation on extraradical mycorrhizal hyphae has been shown to occur in micropores (Saito and Marumoto, 2002). Activated biochar can adsorb microorganisms strongly according to its hydrophobicity (Rivera-Utrilla *et al*., 2001). The habitat attachment by soil microorganism was also accompanied with microbial activities as measured in soil respiration and biomass (Table 2).

Contrasting effects of biochar with varying particles sizes retrieved from the soil was noted for microbial biomass C, whereas smallest particle size studied (0.5-1.0 mm) resulted in the lowest in microbial biomass P after 28 days of incubation (Jaafar *et. al*, 2015a). Following this study, comparison of biochar (activated biochar versus biochar from pyrolysis) was made in 4 types of soil histories (Jaafar *et.al*.,2015b). In general, stimulation of microbial respiration was greater in activated biochar-treated soil than in soil amended with normal pyrolyzed woody biochars (Table 2). This could be attributed to greater porosity of activated biochar than normal pyrolysis biochar (Downie *et al*., 2009; Rillig *et al*., 2010). Other studies conducted on soil biology including mycorrhizal fungi and soil fauna in response to biochar amendments is presented in Table 3. The proposed mechanisms varied from habitat provision to soil biota provided by biochar and the subsequent soil properties affecting these soil microorganisms and fauna.

Table 2 Effects of biochars on soil microbial respiration in four soils after 28-d incubation period. Soils 1, 2 and 3 were collected from the alternative crop area, agroforestry area and pasture, respectively, in Chittering, Soil 4 from the pasture in Bullsbrook, WA. (after Jaafar et.al., 2015b)

Biochar Sources	Cumulative Carbon Dioxide (mg kg^{-1}) after 28 days			
	Soil 1	Soil 2	Soil 3	Soil 4
Control	32.81	34.74	61.80	22.65
Simcoa	37.08	40.61	54.02	32.59
Activated Biochar	43.99	47.58	60.15	30.24
Wundowie	33.57	32.05	48.12	26.72

Ingestion of Biochar by Soil Fauna

A mechanism by which biochar interact with soil biota is through biochar influences on soil fauna. While biochar creates a condusive environment and refuge for soil microorganisms, their influence on soil fauna as microbial predator is not known. Soil fauna may be attracted to biochar particles, preying for fungi and bacteria but the potential influence soil microorganism-predator cycles remain unclear. Direct biochar effects of soil fauna can influence microbial communities via their grazing activities, in combination modification of soil structure (McCormack *et al*., 2013). Earthworms for example, are able to ingest biochar particles. Through ingestion, they excrete fecal and producing biochar complexed with minerals (Ponge *et al*., 2006). The minerals released would be beneficial for both microorganisms and plants.

Table 3 Summary of several experiments on the application of biochar and the effects on soil biota (after Warnock *et al*., 2007; Lehmann *et al*., 2011; McCormack *et al*., 2013).

Biochar Feedstock	Main Effects on Soil Microorganisms	References
Poultry	Increased microbial biomass C	Chan *et al*. (2008)
Bull manure (dairy) and pine (*Pinus* spp.)	Respiration increased with increasing charcoal amount	Kolb *et al*. (2009)
Papermill waste	Increased microbial activity (in ferrosol) but decreased microbial activity in calcarosol . Decline in microbial activity when unfertilised	Van Zwieten *et al*. (2010)
Pinus radiata	Increased fungal and bacterial abundance; promotion of P solubilising bacteria	Anderson *et al*. (2011)
Wood (Eucalyptus)	Decreased microbial biomass C	Dempster *et al*. (2012)
rice husk	Increased % mycorrhizal colonisation	Ishii and Kadoya (1994)
Woody biochar	Increased % mycorrhizal colonisation	Solaiman *et al*. (2010)
Hardwood dust	Increased % mycorrhizal colonisation	Elmer and Pignatello (2011)
Natural wood charcoal	Increased earthworm *P. corethrurus* activity	Topoliantz and Ponge (2005)
Wood ash	Reduced abundance of fungivorous nematodes	Liiri et al. (2007)
Wood ash	Decreased abundance of *Cognettia sphagnetorum* (Enchytraeids)	Nieminen (2008)
Wood waste	Increased nematodes abundance	Husk and Major (2010)

The effects of biochar on earthworms is normally observed in earthworms avoidance tests. These are conducted to observe on toxic effect of the biochar and ecological preferences of the organisms. Responses include earthworm preference (Busch *et al.*, 2012; Hale *et al.*, 2013), as well as avoidance (Chan *et al.*, 2008; Van Zwieten *et al.*, 2010; Tammeorg *et al.*, 2014). This approach is normally used to predict any potential negative responses (Domene *et al.*, 2014). Effects of biochar were also observed on nematodes and enchytraeids. Although Zhang *et al.* (2013) reported total nematode abundance did not change with 2.4 Mg ha^{-1} wheat straw biochar addition, they also observed higher diversity of nematodes and an increase in the abundance of fungivores. Van Zwieten *et al.* (2010) reported papermills waste biochar influenced the preference of earthworms (*Eisenia andrei*) in an acid soil this was not observed in a basic soil. The underlying explanations might be the excessive liming effects associated with toxic effects of biochar, as well as water content (Li *et al.*, 2011).

Ameliorative Effects of Biochar

Biochar as a soil microbial habitat has been linked to enhancement of soil properties as shown in soil microbial biomass and other microbial indicators (Zackrisson *et al.*, 1996; Pietikäinen *et al.*, 2000; Hockaday *et al.*, 2007).

Soil pH

Possible changes in soil pH triggered by biochar amendment could affect many aspects of microbial activity. Under extremely acidic or alkaline conditions, microbial activity is decreased. Neutral to alkaline pH levels are generally favorable for the microbial degradation of various compounds (Leahy and Colwell, 1990). Neutralizing the soil pH to 7.0 resulted in increased soil respiration rate. Bacterivore soil fauna group may gain benefits in acid soils due to increased pH associated with biochar applications that also supported the bacteria (McCormack *et al.*, 2013). Van Zwieten *et al.* (2010) showed that papermill waste biochar increased soil pH and higher microbial activity when applied at 2 and 1.5% biochar to the acid soil. On the contrary, alkaline properties of biochar, especially the activated biochar, did not improve soil pH (Jaafar *et al.*, 2015b).

Substrate and Water Availability

Biochar has the distinct feature that it is rich in C and relatively inert, with minimal chemical or biochemical alteration (Nichols *et al*., 2000). However, Frink (1992) and Bird *et al*. (2002) showed that pyrogenic biochar had a reduced C content. Microorganisms may have a biodegrading effect and their interactions with biochar could affect nutrient cycling and nutrient availability to microorganisms and to plants. Other biochar benefits may indirectly influence soil biology including increased nutrient availability, substrate via carbon and water retention, and increased pH especially when applied to acid soils (Lehmann *et al*., 2011). Large surface area of activated biochar may provide more opportunity for organic compounds to be absorbed, enhancing their access to microorganisms (Cornelissen and Gustafsson, 2004; Foo and Hameed, 2010).

The porous structure of biochar and its strong affinity for nutrients could provide microorganisms with substrates and protect them from their natural predators. Biochar can interact with soil particles and influence soil microbial communities by introducing labile organic compounds for microbial growth and activating nutrient retention (Graber *et al*., 2010; Cornelissen *et al*., 2005; Keech *et al*., 2005). Surface-associated fungi and bacteria may able to degrade substrates and biochar surfaces (Atkinson *et al*., 2010; Sohi *et al*., 2010). Biochar can adsorb cations and organic matter that are beneficial for soil microorganisms (Liang *et al*., 2006)

Hammer *et al*. (2014) observed that biochar can be used by AM fungi as a surface for hyphal growth and a nutrient source. Through their monoxenic culture studies, they observed that AM fungi hyphae can grow on and into biochar particles. Through the contact made between AM fungal hyphal and biochar surfaces, uptake of P sources was permitted and subsequent translocation to the associated host roots was reported. This scenario implies that biochar surfaces improved P translocation to the host roots via interaction with AM fungal hyphal attached to its surfaces. AM fungal hyphae which become attached on biochar surfaces, have potential to subsequently access microsites within biochar. Finally, nutrient-containing substrates might be preferentially adsorbed by biochar resulting in increased nutrient availability for microbial colonization. Higher concentration of substrate on biochar surfaces is advantageous to microorganisms to (Swaine *et al*., 2013).

Conclusion

The underlying mechanisms of biochar and soil fauna and microorganisms could provide understanding of the related processes in soil. Concerns regarding the co-existence of beneficial soil biological groups and their importance in nutrient cycling and soil processes with biochar addition needs further consideration. However, biochars differ widely according to the source of organic materials from which they are prepared, and so caution in generalizing about biochar effects is required, and needs to consider fully the origin of the biochar. Biochar needs to be managed in environmental-friendly manner for soil biological, chemical and physical balance and environmental sustainability.

References

Ahiduzzaman, M., & Islam, A. S. (2016). Preparation of porous bio-char and activated carbon from rice husk by leaching ash and chemical activation. *SpringerPlus*, *5*(1), 1248.

Anderson, C. R., Condron, L. M., Clough, T. J., Fiers, M., Stewart, A., Hill, R. A., & Sherlock, R. R. (2011). Biochar induced soil microbial community change: implications for biogeochemical cycling of carbon, nitrogen and phosphorus. *Pedobiologia*, *54*(5), 309-320.

Ascough, P. L., Sturrock, C. J., & Bird, M. I. (2010). Investigation of growth responses in saprophytic fungi to charred biomass. *Isotopes in environmental and health studies*, *46*(1), 64-77.

Atkinson, C. J., Fitzgerald, J. D., & Hipps, N. A. (2010). Potential mechanisms for achieving agricultural benefits from biochar application to temperate soils: a review. *Plant and soil*, *337*(1-2), 1-18.

Beesley, L., & Dickinson, N. (2011). Carbon and trace element fluxes in the pore water of an urban soil following greenwaste compost, woody and biochar amendments, inoculated with the earthworm Lumbricus terrestris. *Soil Biology and Biochemistry*, *43*(1), 188-196.

Bird, M. I., Turney, C. S. M., Fifield, L. K., Jones, R., Ayliffe, L. K., Palmer, A., & Robertson, S. (2002). Radiocarbon analysis of the early archaeological site of Nauwalabila I, Arnhem Land, Australia: implications for sample suitability and stratigraphic integrity. *Quaternary Science Reviews*, *21*(8), 1061-1075.

Blackwell, P., Krull, E., Butler, G., Herbert, A., & Solaiman, Z. (2010). Effect of banded biochar on dryland wheat production and fertiliser use in south-western Australia: an agronomic and economic perspective. *Soil Research*, *48*(7), 531-545.

Blackwell, P., Joseph, S., Munroe, P., Anawar, H. M., Storer, P., Gilkes, R. J., & Solaiman, Z. M. (2015). Influences of biochar and biochar-mineral complex on mycorrhizal colonisation and nutrition of wheat and sorghum. *Pedosphere*, *25*(5), 686-695.

Busch, D., Kammann, C., Grünhage, L., & Müller, C. (2012). Simple biotoxicity tests for evaluation of carbonaceous soil additives: establishment and reproducibility of four test procedures. *Journal of environmental quality*, *41*(4), 1023-1032.

Chan, K. Y., Van Zwieten, L., Meszaros, I., Downie, A., & Joseph, S. (2008). Using poultry litter biochars as soil amendments. *Soil Research*, *46*(5), 437-444.

Cornelissen, G., & Gustafsson, Ö. (2005). Importance of unburned coal carbon, black carbon, and amorphous organic carbon to phenanthrene sorption in sediments. *Environmental science & technology*, *39*(3), 764-769.

Cornelissen, G., Gustafsson, Ö., Bucheli, T. D., Jonker, M. T., Koelmans, A. A., & van Noort, P. C. (2005). Extensive sorption of organic compounds to black carbon, coal, and kerogen in sediments and soils: mechanisms and consequences for distribution, bioaccumulation, and biodegradation. *Environmental Science & Technology*, *39*(18), 6881-6895.

Dempster, D. N., Jones, D. L., & Murphy, D. V. (2012). Clay and biochar amendments decreased inorganic but not dissolved organic nitrogen leaching in soil. *Soil Research*, *50*(3), 216-221.

Ding, Y., Liu, Y. X., Wu, W. X., Shi, D. Z., Yang, M., & Zhong, Z. K. (2010). Evaluation of biochar effects on nitrogen retention and leaching in multi-layered soil columns. *Water, Air, & Soil Pollution*, *213*(1-4), 47-55.

Downie, A., A. Crosky, and P. Munroe, 2009. Physical properties of biochar. In: Lehmann, J. and S. Joseph (Eds.), Biochar for Environmental Management, Science and Technology, Earthscan, London, UK. pp. 13–32.

Domene, X., Mattana, S., Hanley, K., Enders, A., & Lehmann, J. (2014). Medium-term effects of corn biochar addition on soil biota activities and functions in a temperate soil cropped to corn. *Soil Biology and Biochemistry*, *72*, 152-162.

Elmer, W. H., & Pignatello, J. J. (2011). Effect of biochar amendments on mycorrhizal associations and Fusarium crown and root rot of asparagus in replant soils. *Plant Disease*, *95*(8), 960-966.

Foo, K. Y., & Hameed, B. H. (2010). Detoxification of pesticide waste via activated carbon adsorption process. *Journal of hazardous materials*, *175*(1), 1-11.

Frink, D. S. (1992). The chemical variability of carbonized organic matter through time. Archaeology of Eastern North America, 67-79.

Graber, E. R., Harel, Y. M., Kolton, M., Cytryn, E., Silber, A., David, D. R., ... & Elad, Y. (2010). Biochar impact on development and productivity of pepper and tomato grown in fertigated soilless media. *Plant and Soil*, *337*(1-2), 481-496.

Gundale, M. J., & DeLuca, T. H. (2006). Temperature and source material influence ecological attributes of ponderosa pine and Douglas-fir charcoal. *Forest Ecology and Management*, *231*(1), 86-93.

Hale, S. E., Jensen, J., Jakob, L., Oleszczuk, P., Hartnik, T., Henriksen, T., ... & Cornelissen, G. (2013). Short-term effect of the soil amendments activated carbon, biochar, and ferric oxyhydroxide on bacteria and invertebrates. *Environmental science & technology*, *47*(15), 8674-8683

Hammer, E. C., Balogh-Brunstad, Z., Jakobsen, I., Olsson, P. A., Stipp, S. L., & Rillig, M. C. (2014). A mycorrhizal fungus grows on biochar and captures phosphorus from its surfaces. *Soil Biology and Biochemistry*, *77*, 252-260.

Hockaday, W. C., Grannas, A. M., Kim, S., & Hatcher, P. G. (2007). The transformation and mobility of charcoal in a fire-impacted watershed. *Geochimica et Cosmochimica Acta*, *71*(14), 3432-3445.

Husk, B., & Major, J. (2010). Commercial scale agricultural biochar field trial in Québec, Canada over two years: effects of biochar on soil fertility, biology and crop productivity and quality. *Dynamotive Energy Systems, February*.

Ishii, T., & Kadoya, K. (1994). Effects of charcoal as a soil conditioner on citrus growth and vesicular-arbuscular mycorrhizal development. *Journal of the Japanese Society for Horticultural Science*, *63*(3), 529-535.

Jaafar, N. M. (2014). Biochar as a habitat for arbuscular mycorrhizal fungi. In *Mycorrhizal Fungi: Use in Sustainable Agriculture and Land Restoration* (pp. 297-311). Springer Berlin Heidelberg

Jaafar, N. M., Clode, P. L., & Abbott, L. K. (2014). Microscopy observations of habitable space in biochar for colonization by fungal hyphae from soil. *Journal of Integrative Agriculture*, *13*(3), 483-490.

Jaafar, N. M., Clode, P. L., & Abbott, L. K. (2015)a. Soil microbial responses to biochars varying in particle size, surface and pore properties. *Pedosphere*, *25*(5), 770-780.

Jaafar, N. M., Clode, P. L., & Abbott, L. K. (2015)b. Biochar-soil interactions in four agricultural soils. *Pedosphere*, *25*(5), 729-736.

Joseph, S. D., Camps-Arbestain, M., Lin, Y., Munroe, P., Chia, C. H., Hook, J., Van Zwieten, L., Kimber, S., Cowie, A., Singh, B.P. & Lehmann, J. (2010). An investigation into the reactions of biochar in soil. *Soil Research*, *48*(7), 501-515.

Keech, O., Carcaillet, C., & Nilsson, M. C. (2005). Adsorption of allelopathic compounds by wood-derived charcoal: the role of wood porosity. *Plant and Soil*, *272*(1), 291-300.

Kimetu, J. M., & Lehmann, J. (2010). Stability and stabilisation of biochar and green manure in soil with different organic carbon contents. *Soil Research*, *48*(7), 577-585.

Knowles, O. A., Robinson, B. H., Contangelo, A., & Clucas, L. (2011). Biochar for the mitigation of nitrate leaching from soil amended with biosolids. *Science of the Total Environment*, *409*(17), 3206-3210.

Kolb, S. E., Fermanich, K. J., & Dornbush, M. E. (2009). Effect of charcoal quantity on microbial biomass and activity in temperate soils. *Soil Science Society of America Journal*, *73*(4), 1173-1181.

Pignatello, J. J., Kwon, S., & Lu, Y. (2006). Effect of natural organic substances on the surface and adsorptive properties of environmental black carbon (char): attenuation of surface activity by humic and fulvic acids. *Environmental Science & Technology*, *40*(24), 7757-7763.

Leahy, J. G., & Colwell, R. R. (1990). Microbial degradation of hydrocarbons in the environment. *Microbiological reviews*, *54*(3), 305-315.

Lehmann, J., Rillig, M. C., Thies, J., Masiello, C. A., Hockaday, W. C., & Crowley, D. (2011). Biochar effects on soil biota–a review. *Soil biology and biochemistry*, *43*(9), 1812-1836.

Li, D., Hockaday, W. C., Masiello, C. A., & Alvarez, P. J. (2011). Earthworm avoidance of biochar can be mitigated by wetting. *Soil Biology and Biochemistry*, *43*(8), 1732-1737.

Liang, B., Lehmann, J., Solomon, D., Kinyangi, J., Grossman, J., O'neill, B., Skjemstad JO, Thies J, Luizao FJ, Petersen J & Neves, E. G. (2006). Black carbon increases cation exchange capacity in soils. *Soil Science Society of America Journal*, *70*(5), 1719-1730.

Liiri, M., Ilmarinen, K., & Setälä, H. (2007). Variable impacts of enchytraeid worms and ectomycorrhizal fungi on plant growth in raw humus soil treated with wood ash. *Applied soil ecology*, *35*(1), 174-183.

McCormack, S. A., Ostle, N., Bardgett, R. D., Hopkins, D. W., & Vanbergen, A. J. (2013). Biochar in bioenergy cropping systems: impacts on soil faunal communities and linked ecosystem processes. *Gcb Bioenergy*, *5*(2), 81-95.

Nichols, G. J., Cripps, J. A., Collinson, M. E., & Scott, A. C. (2000). Experiments in waterlogging and sedimentology of charcoal: results and implications. *Palaeogeography, Palaeoclimatology, Palaeoecology*, *164*(1), 43-56.

Nieminen, J. K. (2008). Labile carbon alleviates wood ash effects on soil fauna. *Soil Biology and Biochemistry*, *40*(11), 2908-2910.

Pietikäinen, J., Kiikkilä, O., & Fritze, H. (2000). Charcoal as a habitat for microbes and its effect on the microbial community of the underlying humus. *Oikos*, *89*(2), 231-242.

Ponge, J. F., Topoliantz, S., Ballof, S., Rossi, J. P., Lavelle, P., Betsch, J. M., & Gaucher, P. (2006). Ingestion of charcoal by the Amazonian earthworm Pontoscolex corethrurus: A potential for tropical soil fertility. *Soil Biology and Biochemistry*, *38*(7), 2008-2009.

Rillig, M. C., Wagner, M., Salem, M., Antunes, P. M., George, C., Ramke, H.G., Titirici, M.M. & Antonietti, M. (2010). Material derived from hydrothermal carbonization: effects on plant growth and arbuscular mycorrhiza. *Applied Soil Ecology*, *45*(3), 238-242.

Rivera-Utrilla, J., Bautista-Toledo, I., Ferro-García, M. A., & Moreno-Castilla, C. (2001). Activated carbon surface modifications by adsorption of bacteria and their effect on aqueous lead adsorption. *Journal of Chemical Technology and biotechnology*, *76*(12), 1209-1215.

Rutigliano, F. A., Romano, M., Marzaioli, R., Baglivo, I., Baronti, S., Miglietta, F., & Castaldi, S. (2014). Effect of biochar addition on soil microbial community in a wheat crop. *European Journal of Soil Biology*, *60*, 9-15.

Saito, M., & Marumoto, T. (2002). Inoculation with arbuscular mycorrhizal fungi: the status quo in Japan and the future prospects. In *Diversity and Integration in Mycorrhizas* (pp. 273-279). Springer Netherlands.

Sohi, S. P., Krull, E., Lopez-Capel, E., & Bol, R. (2010). A review of biochar and its use and function in soil. *Advances in agronomy*, *105*, 47-82.

Solaiman, Z. M., Blackwell, P., Abbott, L. K., & Storer, P. (2010). Direct and residual effect of biochar application on mycorrhizal root colonisation, growth and nutrition of wheat. *Soil Research*, *48*(7), 546-554.

Steinbeiss, S., Gleixner, G., & Antonietti, M. (2009). Effect of biochar amendment on soil carbon balance and soil microbial activity. *Soil Biology and Biochemistry*, *41*(6), 1301-1310.

Steiner, C., Das, K. C., Garcia, M., Förster, B., & Zech, W. (2008). Charcoal and smoke extract stimulate the soil microbial community in a highly weathered xanthic Ferralsol. *Pedobiologia*, *51*(5), 359-366.

Swift, M. J., Heal, O. W., & Anderson, J. M. (1979). *Decomposition in terrestrial ecosystems* (Vol. 5). Univ of California Press.

Swaine, M., Obrike, R., Clark, J. M., & Shaw, L. J. (2013). Biochar alteration of the sorption of substrates and products in soil enzyme assays. *Applied and Environmental Soil Science*, *2013*.

Tammeorg, P., Parviainen, T., Nuutinen, V., Simojoki, A., Vaara, E., & Helenius, J. (2014). Effects of biochar on earthworms in arable soil: avoidance test and field trial in boreal loamy sand. *Agriculture, ecosystems & environment*, *191*, 150-157.

Thies, J. E., & Rillig, M. C. (2009). Characteristics of biochar: biological properties. *Biochar for environmental management: Science and technology*, 85-105.

Topoliantz, S., & Ponge, J. F. (2005). Charcoal consumption and casting activity by Pontoscolex corethrurus (Glossoscolecidae). *Applied Soil Ecology*, *28*(3), 217-224.

Van Zwieten, L., Kimber, S., Morris, S., Chan, K. Y., Downie, A., Rust, J., ... & Cowie, A. (2010). Effects of biochar from slow pyrolysis of papermill waste on agronomic performance and soil fertility. *Plant and soil*, *327*(1-2), 235-246.

Verheijen, F., Jeffery, S., Bastos, A. C., Van Der Velde, M., Diafas, I., & Parsons, C. (2009). Biochar application to soils: a critical scientific review of effects on soil properties, processes and functions. Joint Research Centre. *Institute for Environment and Sustainability, Ispra, Italy*.

Warnock, D. D., Lehmann, J., Kuyper, T. W., & Rillig, M. C. (2007). Mycorrhizal responses to biochar in soil–concepts and mechanisms. *Plant and soil*, *300*(1-2), 9-20.

Yeboah, E., Ofori, P., Quansah, G. W., Dugan, E., & Sohi, S. P. (2009). Improving soil productivity through biochar amendments to soils. *African Journal of Environmental Science and Technology*, *3*(2), 34-41.

Zackrisson, O., Nilsson, M. C., & Wardle, D. A. (1996). Key ecological function of charcoal from wildfire in the Boreal forest. *Oikos*, 10-19.

Zhang, X.-K., Li, Q., Liang, W.J., Zhang, M., Xue-Lian, B. A. O., & Zu-Bin, X. I. E. (2013). Soil nematode response to biochar addition in a Chinese wheat field. *Pedosphere*, *23*(1), 98-103

4

Roles of Organic Matter in Enhancing Tropical Soil Quality and Plant Growth

Susilawati, K. and Ahmed, O.H.

Introduction

Soil organic matter is a component in soil that has great influence on soil aggregation, water retention or soil water availability (Weil and Magdoff, 2004), promotion of plant growth and supply of micronutrient cation pools (Chen *et al.,* 2004), nutrient cycles or nutrient reservoir (Brady and Weil, 1999), C cycle (Weil and Magdoff, 2004) and so on. The role of soil organic matter in alleviating soil compaction, improving soil workability, enhancing water use efficiency, suppressing plant's pests and diseases, controlling soil erosion whether by wind and water, and controlling surface crusting (Weil and Magdoff, 2004) make the soil organic matter more important in recent years. This chapter highlights the essential of organic matter in sustaining soil quality and promoting optimum plant growth.

Classification and Decomposition

Soil organic matter is classified in the same category as humus. According to Stevenson (1994), humus is the total of organic compounds in soil exclusive of undecayed plant and animal tissues, their 'partial decomposition' products, and the soil biomass. Humification processes convert almost three quarter to about fourth fifth of organic matter into humus. Humus is characterized by high hydrophilicity and sorption ability whereby it considerably participates in the physico-chemical processes in soils. The sorption capacity of humus is 15 to 36 times higher than that of clayey minerals. Humus particles have a negative charge and are surrounded by cations (Hoffman and Hoffman, 2007). In most agricultural soils, the bulk of the organic matter occurs as stable humus (Stevenson, 1994).

The classification of organic matter varies, as it could be based on their decomposition and researcher's understanding. According to Tan (2003), dead fraction (*humus)* is the final product of decomposition where chemical and biological processes take place while live fraction is the materials that take part in the decomposition processes. Stevenson (1994) agreed with this definition except that Stevenson (1994) included microbial biomass and water-soluble organics as parts of soil organic matter.

Generally organic materials go through decomposition processes until they reach stability. Thus, the decomposition stages and rate of organic matter production depends on climate, soil physical, chemical and biological properties factors and vegetation composition. Decomposition of litter inversely relates to C:N and lignin:N ratios (Post, 2006). Thus, C:N and lignin:N ratio could be indicators for decay rate or decomposition stages. The incorporation part of the C into microbial tissue, liberation of CO_2, conversion of organic form of N to NH_3 and utilization of part of N by soil microorganisms for new cell synthesis establish their consistency for C and N (Stevenson, 1994) in composted organic matter and can be used as an indicator for maturity and classification purposes. A C:N ratio of more than 30 slows decomposition processes, beyond this ratio leads to immobilization whilst a ratio of less than 20 releases N (Rice, 2006). The value of 10 to 12 has been proposed by Jimenez and Gracia (1992) for stable and well decomposed organic matter.

Mode of land use has significant effect on soil organic C storage. Shifting of natural vegetation to agriculture results in marked decline in soil organic matter levels due to less C addition from agricultural practices (Weil and Magdoff, 2004; Spaccini *et al.,* 2001). Widespread conversion of natural forests and grasslands to agricultural cultivation has reduced soil organic matter C stocks significantly in a period of 150 years (Wood *et al.,* 2000).

Soil taxonomy uses the same approach in classifying soil organic matter into *fibric, hemic, and sapric* fractions which are least decomposed, partly decomposed, and totally decomposed, respectively (Tan, 2003). However, Aiken *et al.* (1985) classified organic matter based on humic and nonhumic substances. Nonhumic substances refer to the organic compounds that can still be categorized chemically, whilst humic compound contains polyelectrolytic macromolecules complex mixture which come from chemical alteration of nonhumic substances (Stevenson, 1986).

Identification and Quantification

Identification and quantification of organic matter plays a major role in confirming the types and amount of humic substances in soils. With the use of advance technology (e.g. solid-state CPMAAS, ^{13}C-NMR) identification and quantification of major types of functional groups (e.g. alkyl, N-alkyl, O-alkyl, acetal, aromatic, phenolic and carboxylic) existing in organic matter is possible (Guant *et al.,* 2001; Oades, 1995). With these new technologies, recently large amount of glomalin which particulates in organic matter (HA, FA and humin) has been discovered from the recalcitrant portion of soil organic matter (Weil and Magdoff, 2004).

Stevenson (1994) has discussed the use of Nuclear Magnetic Resonance (NMR) spectroscopy and analytical pyrolysis. These techniques have been used in characterizing organic matter in the intact soil without going through the traditional isolation processes. According to Stevenson (1994), these methods provide information on compositional changes in crop residues, peat, and the litter of forest soils during biodegradation and humification. The molecular weight of organic matter could also be determined using different methods. According to Tan (2003) filtration, gel chromatography, viscosimetry, freezing point and x-ray diffraction are the various methods that can be used in determining molecular weight of organic matter. However, due to different procedures and reactions involved in each of the methods used, the values of the molecular weight obtained may widely differ from one to the other. As an example, the molecular weight of HA may vary from 36,000 to 25,000 and 1,390 as determined by viscosimetry, freezing point, and x-ray diffraction techniques, respectively (Tan, 2003).

The use of ultraviolet and visible light spectrophotometry is commonly related to the identification of humic substances colour (Tan, 2003). Two different wavelengths are commonly used [465 (E_4) and 650 (E_6) nm] in measuring the color of humic substances. The ratio of E_4 and E_6 commonly called as colour ratio serves an index for the rate of light absorption in the visible range. High colour ratio (7 to 8 or higher) indicates the presence of FA or low molecular weight HA. Whilst low ratio corresponds to the high molecular weight HA or other compounds (Tan, 2003). In forest soils, ultraviolet spectrophotometry could be used to investigate the substitution of the aromatic C content and related properties of dissolved organic carbon (DOC) (Jaffrain *et al.*, 2007).

Buffering and Exchange Capacity

Organic matter contributes to cation exchange capacity (CEC) of soils. Soil with low clay contents normally depends on soil organic matter for their CEC. For example, a soil with 50% illitic and chloritic clays (500 g/kg) and 5% soil organic matter and pH of 6.5 will have a CEC of ca. 20 cmol (+) kg^{-1} with approximately half originating from the clay and half from the soil organic matter. At this pH, the CEC of soil organic matter mostly comes from carboxyl groups which have lost H^+ and indirectly gained a negative charge. However, when the soil's pH is lower than the pKa values of carboxylic groups or below pH 4, the CEC of soil organic matter is not expressed due to tightly held of H^+ by soil's organic matter acid groups (Weil and Magdoff, 2004). The presence of organic matter increases CEC by 20 to 70% (Stevenson, 1994).

Types of Organic Matter Added into/onto The Soil

There are many types of soil organic matter. They can be divided into two categories; fresh residues or composted organic matter. These aspects are subsequently discussed.

Crop Residues

Surface placement of plant residue can maximize erosion-resistance besides significantly contributing to nutrient cycling functions (Gascho, 2006). Kayuki and Wortmann (2001) demonstrated that application of high quality organic residues increases maize and bean yield. Organic residues also affect water infiltration rate and hydraulic conductivity, processes which control the amount of water entering the soil that can be stored for plant use (Weil and Magdoff, 2004; Whitbread *et al.,* 2000). Polysaccharides exuded from roots and microorganisms adsorb strongly to negatively charged soil particles through cation bridging (Chenu, 1995) and contribute notably to aggregate stability (Martens and Fankenberger, 1992; Angers and Mehuy, 1989; Cheshire *et al.,* 1989) and hydraulic conductivity (Robertson *et al.,* 1991).

Organic Amendments

Organic amendments are one type of organic matter added to the soil. These organic materials improve soil fertility because of their organic matter contribution (Ferreras *et al.,* 2006; Flavel and Murphy, 2006; Kockba *et al.,* 2004). Incorporation of organic amendments alleviate the constraints (e.g. soil acidity) and encourage deep root growth besides being good in improving soil chemical, biological, and physical characteristics (Gopinath *et al.,* 2008; Clark *et al.,* 2007).

Improvement of soil fertility is intimately linked with types and increased levels of organic matter. Soils NH_4^+ ions initially depend on the types of organic amendments added whilst NO_3^- ions are not. Application of chicken manure significantly increased soil pH (maximum up to 8.1) consequently exchangeable NH_4^+ and available NO_3^-. Chicken manures high in C initiate rapid biological activity and improved soil quality (Clark *et al.,* 2007). Organic amendments improve chemical fertility and biological activity in high clay sodic subsoil and contribute to an increase in C content (Clark *et al.,* 2007). The CEC of the soil increased with the addition of organic amendments such as chicken manure (Clark *et al.,* 2007).

Composted and uncomposted animal manures, organic industrial by-products, yard trimmings and other organic residues are used worldwide as organic soil amendments. Amendments characteristics such as availability, cost, ease of application, and chemical and biological properties affect their utility in a specific cropping system. In addition, the problems and opportunities in the cropping systems that might be ameliorated or exacerbated by amendments must be considered. Long-term low-rate annual amendment might be a more economically, agronomically, and environmentally desirable alternative to single-year high-rate application (Stone *et al.,* 2004).

Organic amendments also have the ability to promote biodegradation of certain hazardous chemicals in soils. In the case of herbicide (e.g. Atrazine, trifuralin and so on), application of 0.5% manure, 5% peat, and 5% cornstalk enhanced degradation up to 44% (Moorman *et al.,* 2001). On the other hand, organic amendments of high N content suppress soilborne diseases through the toxic effects of NH_3, nitrous acid, or volatile fatty acids on plant pathogen growth and survival (Stone *et al.,* 2004; Lazarovits, 2001).

Compost

Compost is the most popular organic fertilizer that has been used in agriculture for many years. Compost can be prepared in many ways from different sources of organic waste such as agricultural, sewage sludge, animal, household wastes and so on.

The pH of compost are usually around neutral to slightly alkaline and rarely acidic even though a slight pH drop may occur during the first few days of anaerobic composting due to the production of volatile organic acids. After this period, the pH becomes neutral or slightly alkaline again because these acids are converted to methane and carbon dioxide by the reaction of methane forming bacteria (Ahmed *et al.,* 2004). The CEC of the compost, a measure of the capacity of the compost to hold exchangeable cations, such as K, Ca, Mg and Na to the negatively charged surfaces of the compost (e.g. OH, COOH, when dissociated) also increased from an initial value of 32 to 68.32 cmol kg^{-1}. This high CEC of the compost indicates that the organic material of the compost has become more humified (Sullivan and Miller, 2000).

Application of vermicompost significantly increases yield, enhances transplant growth rate, fresh weight, and the vase life especially for flowers (Nethra *et al.,* 1999). Composted manure increases pH, electrical conductivity (EC), CEC and NO_3^- concentration. It also decreases temperature, moisture and organic matter content, NH_4^+ concentration, and C:N ratio (Gil *et al.,* 2008).

Manure

Animal manures were the first fertilizers used in agriculture (Gascho, 2006). Animal manures normally contain more N compared to P and K. The products of those manures must be odourless or odour-free, pathogen-free and dewatered. The chemical activity of dissolved organic carbon (DOC) fractions can be particularly important in soil systems amended with sludge or manure, in which dissolved matter increases the dissolution of sorbed organic and inorganic elements and facilitate transport through the soil profile (Kaschl *et al.,* 2000).

Benefits and Functions

Organic matter plays a major role in nutrient availability to plant roots in time, space, and form and retention of water in sufficient quantities with appropriate potential energy to be available for root uptake on an almost continuous basis. It provides a network of interconnected pores sufficient to provide pathways of low physical resistance to root growth and meet plant root needs by supplying oxygen and removing carbon dioxide and toxic gases. Plant growth promoting soil organisms, sufficient rooting depth and physical support for optimal plant growth are also some of the indirect functions of organic matter (Weil and Magdoff, 2004). High levels of soil organic matter reduce soil erosion and runoff, enhance soil aggregation and nutrient cycling, besides improving infiltration, movement and retention of soil's water (Greenland and Szabolcs, 1994; Woomer and Swift, 1994).

Total porosity and bulk density also relates directly to soil organic matter content. The relationship between soil organic matter and bulk density is linear in the presence of limited range of soil organic matter contents and curvilinear in the wide range of organic matter. Low particle density due to the presence of organic matter is a major cause for the effect in soil aggregation and bulk density (Franzluebbers *et al.,* 2001). Organic matter increases the capability of holding plant available water in soil, where the difference between the water content at field capacity and that held at the permanent wilting point occurs (Hudson, 1994). According to Stevenson (1994), the presence of organic matter helps in preventing shrinkage and drying of the soils. Besides improving moisture-retaining properties, it also stabilizes soil structure that indirectly affects gas exchange. Organic matter is also involved in solubility of minerals in soil, soil buffering properties thus making soil a better medium for plant growth (Stevenson, 1994).

Soil organic matter serves as both the principal long-term storage medium and as the primary short-term source of N, P and S and other nutrients. Most of the N taken by crops is claimed to come from organic pools that cycle through microbial biomass. Only 10 to 50% of inorganic N fertilizers could be taken up by such N-demanding crop (Omay *et al.,* 1998; Reddy and Reddy, 1993) whilst almost 70% of N could be taken by plant from organic soil pools (Weil and Magdoff, 2004). Crop response to mineral fertilizer inputs is increased in soils in the presence of organic matter (Cassman, 1999; Avnimelech, 1986).

Soil organic matter can also be a source or sink of plant nutrients. Approximately 90 to 95% of the soil N, 40% of the soil P, and 90% of the soil S is associated with soil organic matter. Generally, the C:N:P:S ratio is 100:10:1:1 (Rice, 2006). However, all of these nutrients are in organic forms that need to go through other processes to make it available. As an example, 90 to 95% organic N needs to be mineralized before it is used by plants.

Roles of Humic Substances

Humic substances are known to play significant roles in agriculture. Some of the roles are subsequently discussed.

Nitrogen Loss Control by Humic Acids

Humic acids as part of humic substances reduce ammonia loss from urea. As stated by Susilawati *et al.* (2009), mixture of liquid HA with urea has reduced N loss for about 29% (Table 1). Although urea has been diluted at 4:50 [urea (g):distilled water (mL) / liquid HA / liquid FA / mixture of HA and FA] ratio before it was applied onto the soil surface, the reduction is still significant. Addition other types of CEC sources like zeolite might increase the reduction for about 60% although lower rate (0.75 and 1 g kg^{-1}) of HA was used in combination with zeolite (Ahmed *et al.,* 2006). The ability of HA to retain NH_4^+ (Ahmed *et al.,* 2006) at their negatives charges and the acidic nature of HA might contribute to this significant findings (Siva *et al.,* 1999).

Interaction with Metal Ions

The use of humic substances in chelating metal ions has been reported. The acidic functional groups play a crucial role in the interactions of humic substances with metal ions as well as with some organic pollutants (Janos *et al.,* 2008). The oxygen-containing groups which widely exist in humic substances represent the most important metal binding sites in the humic substances molecules (Janos *et al.,* 2008).

Table 1 Cummulative NH_3 loss for 30 days of incubation

Label	Treatment	N loss (%)	Reduction obtained as compared to T2 (%)
T0	Control	0^{c}	Nd
T1	Urea (liquid)	48.74^{ab}	3.88
T2	Urea (solid)	50.71^{a}	nd
T3	LHA plus urea	35.90^{c}	29.21
T4	LFA plus urea	44.91^{b}	11.44
T5	LHA + LFA (acidified) + urea	46.60^{ab}	8.11
T6	LHA + LFA (unacidified) + urea	48.65^{ab}	4.06
T7	Ammonium sulphate (liquid)	0^{d}	100

Adapted from Susilawati et al. (2009)

Nutrients Availability

In acid soils, there is P deficiency due to strong fixation of inorganic P to iron and aluminum oxide surfaces under low pH conditions. Organic substances from organic matter provide a source of P from mineralization and also reduce the capability of acid soils to lock up P by fixation (Weil and Magdoff, 2004). Addition of HA increases K availability and plant K uptake (Olk, 2006). Humic substances have the ability to store N, P, S and Zn in soils. Thus they can enhanced nutrients uptake and reduce N fertilization in particular (Mayhew, 2004). Use of HA together Zn humate eliminated Zn efficiency symptoms and enhanced dry matter production by 50% in soybean and 120% in wheat. The humate increased soybean dry matter more than $ZnSO_4$ (Ozkutlu *et al.,* 2006).

Enhancement of Plant Growth

Plant growth could be enhanced using organic substances, as an example, HA enhances plant growth. Nitrogen, P, K, Ca, Mg, Fe, Zn and Cu uptake increased because of the presence of HA (Adani *et al.,* 1998; David *et al.,* 1994). The salt of HA such as humates have a great effect on photosynthesis, chlorophyll density and plant root respiration (Chen and Aviad, 1990; Sladky, 1959; Šmídová, 1960). Potassium humate however enhances microbial population of the soil as compared to non humate fertilizer (Bhuma and Selvakumari, 2003).

In case of liquid humic fractions, their uses gave different effect to the plant growth. As stated by Susilawati *et al*. (2011) the efficiency of N, P and K were affected by application of humic fractions but the effect was not the same in roots, stems and leaves (Table 2). Eight treatments namely liquid urea (T1), solid urea (T2), urea + liquid HA (T3), urea + liquid FA (T4), urea + liquid mixture of acidified HA + FA (T5), urea + liquid mixture of unacidified HA + FA (T6), liquid ammonium sulphate [$(NH_4)_2SO_4$] (T7) and control (no fertilizer) (T0) were evaluated in their study.

Table 2 Effect of different treatments on N, P and K uptake efficiency

Plant Part	Treatment	Nutrient efficiency (%)		
		N	P	K
Leaf	T0	Nd	Nd	Nd
	T1	7.18^{b}	4.18^{a}	14.79^{a}
	T2	7.22^{b}	4.03^{a}	14.18^{ab}
	T3	7.26^{b}	3.38^{a}	12.26^{ab}
	T4	9.63^{a}	4.05^{a}	15.39^{a}
	T5	8.56^{ab}	3.74^{a}	10.84^{b}
	T6	8.85^{ab}	3.84^{a}	15.45^{a}
	T7	2.51^{c}	1.02^{b}	4.19^{c}
Stem	T0	Nd	Nd	Nd
	T1	9.01^{b}	4.55^{a}	13.51^{ab}
	T2	7.85^{bc}	4.18^{a}	14.91^{a}
	T3	7.14^{c}	4.45^{a}	14.47^{a}
	T4	9.37^{a}	4.20^{a}	13.84^{ab}
	T5	7.27^{c}	4.23^{a}	11.85^{b}
	T6	6.80^{c}	4.26^{a}	12.34^{b}
	T7	1.69^{d}	0.96^{b}	1.32^{c}
Roots	T0	Nd	Nd	Nd
	T1	3.67^{a}	0.97^{b}	2.73^{abc}
	T2	3.66^{a}	1.45^{a}	3.47^{a}
	T3	2.18^{c}	0.71^{c}	2.16^{c}
	T4	3.46^{a}	1.09^{b}	3.17^{ab}
	T5	3.08^{ab}	1.03^{b}	3.15^{ab}
	T6	2.67^{bc}	0.94^{b}	2.64^{bc}
	T7	0.68^{d}	0.25^{d}	0.14^{d}

Different letters indicate significant difference between means within column using Duncan's New Multiple Range Test (DNMRT) at $p \leq 0.05$.
** Nd – not determine*
Adapted from Susilawati *et al*. (2011)

They found that T4, T5 and T6 significantly improved N used efficiency in leaves by 34, 19 and 23%, respectively (Table 2). Application of HA alone or both (HA and FA) however reduced N used efficiency in stems and the highest N efficiency was shown by FA treatment. The uptake efficiency for P and K in leaves and stems was not affected by humic fractions types. The use of both (HA and FA) has caused the lowest N use efficiency in roots and this has also led to the poor uptake of P and K.

Conclusion

Organic matter is essential in sustaining soil quality or optimum plant growth and development. Many types of fresh or composted organic matter sources are available in our environment that are not yet been utilized optimally to enhance either soil fertility or plant production. This little information could provide some information on the beneficial effect of organic matter or organic substances to the soil and plants.

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References

Adani, F., P. Genevini, P. Zaccheo, and G. Zocchi. 1998. The effect of commercial humic acid on tomato plant growth and mineral nutrition. J. Plant Nutr., 21: 561 – 575.

Ahmed, O.H., M.H.A. Husni, A.R. Anuar, M.M. Hanafi, and E.D.S. Angela, 2004. A modified way of producing humic acid from composted pineapple leaves. Journal of Sustainable Agriculture, 25(1):129 -139.

Ahmed, O.H., H. Aminuddin, and M.H.A. Husni, 2006. Effects of urea, humic acid, and phosphate interactions in fertilizer microsites on ammonia volatilization, and soil ammonium and nitrate contents. International Journal of Agricultural Research, 1(1):25-31.

Aiken, G.R., D.M. McKnight, R.L. Wershaw, and P. MacCarthy. 1985. An introduction to humic substances in soil, sediment, and water. Geochemistry, isolation, and characterization. In: Aiken, G.R., D.M. McKnight, R.L. Wershaw, and P. MacCarthy (Eds.). Wiley-Interscience, New York, NY.

Angers, D.A., and G.R. Mehuys. 1989. Effects of cropping on carbohydrate content and water stable aggregation of a clay soil. Can. J. Soil Sci., 69: 373 – 380.

Avnimelech, Y. 1986. Organic residues in modern agriculture. In: The role of organic matter in modern agriculture. Chen, Y., and Y. Avnimelech (Eds.). Martinus Nijhoff, Dordrecht, pp: 1 – 10.

Bhuma, M. and G. Selvakumari. 2003. Studies on the effect of potassium humate on the biological properties of the soil with Green gram. Madras Agric. J., 90 (10 – 12): 736 – 738.

Brady, N.C., and R.R. Weil. 1999. The nature and properties of soils. 12th edition, Prentice-Hall, Upper Saddle River, NJ.

Cassman, K.G. 1999. Ecological intensification of cereal production systems: Yield potential, soil quality, and precision agriculture. Proceedings of the National Academy of Science, USA, p: 96.

Chen, Y., M.D. Nobili, and T. Aviad. 2004. Stimulatory effects of humic substances on plant growth. In: Magdoff, F. and R.R. Weil. Soil organic matter in sustainable agriculture. CRC Press, New York, Washington D.C. US.

Chen, Y. and T. Aviad. 1990. Effects of humic substance on plant growth. In: MacCarthy, P., C.E. Clapp, R.L. Malcolm, and P.R. Bloom (eds), Humic substances in soil and crop sciences: selected readings. Soil Science Society of America, Madison, Wisconsin, pp: 161 – 186.

Chenu, C. 1995. Extracellular polysaccharides: An interface between microorganisms and soil constituents. In: P.M. Huang (Ed.). Environmental impact of soil component interactions. CRC Press, Boca Raton, FL. p. 217.

Cheshire, M.V., A. Lomax, and C.M. Mundie. 1989. Structure of soil carbohydrates resistant to periodate oxidation. J. Soil Sci., 40: 865 – 872.

Clark, G.J., N. Dodgshun, P.W.G. Sale, and C. Tang. 2007. Changes in chemical and biological properties of a sodic clay subsoil with addition of organic amendments. Soil Biology & Biochemistry, 39: 2806 – 2817.

David, P.P., P.V. Nelson, and D.C. Sanders. 1994. A humic acid improves growth of tomato seedling in solution culture. J. Plant Nutr., 17: 173 – 184.

Ferreras, L., E. Gomez, S. Toresani, I. Firpo, and R. Rotondo. 2006. Effect of organic amendments on some physical, chemical, and biological properties in a horticultural soil. Bioresource Technology, 97: 635 – 640.

Flavel, T.C. and D.V. Murphy. 2006. Carbon and nitrogen mineralization rates after application of organic amendments to soil. Journal of Environmental Quality, 35: 183 – 193.

Franzluebbers A.J., J.A. Stuedemann and S.R. Wilkinson. 2001. Bermudagrass Management in the Southern Piedmont USA.I. Soil and Surface Residue Carbon and Sulfur. Soil Sci. Soc. Am. J., 65: 834 – 841.Gasho, G.J. 2006. Fertilizers: Organic. In: Lal, R. (ed.). Encyclopedia of Soil Science. Vol. 1. Taylor & Francis, New York, pp: 698 – 700.

Gil, M.V., M.T. Carballo, and L.F. Calvo. 2008. Fertilization of maize with compost from cattle manure supplemented with additional mineral nutrients. Waste Management, 28: 1432-1440.

Greenland, D.J., and I. Szabolcs. 1994. Soil resilience and sustainable land use. CAB International, Wallingford, U.K.

Gopinath, K.A., S. Saha, B.L. Mina, S. Kundu, H. Pande, and H.S. Gupta. 2008. Impact of organic amendments on growth, yield and quality of wheat and on soil properties during transition to organic production. Nutr. Cycl. Agroecosyst., 82:51-60.

Guant, J.L., S.P. Sohi, H. Yang, N. Mahieu, and J.R.M. Arah. 2001. a procedure for isolating organic matter fractions suitable for modelling. In: Rees, R.M. *et al*., (Eds.). Sustainable management of soil organic matter. CABI Publishers, New York, pp: 90 – 95.

Hoffmann, K., and J. Hoffmann. 2007. The utilization of peat, lignite, and industrial wastes in the production of mineral-organic fertilizers. American Journal of Agricultural and Biological Sciences, 2(4): 254 – 259.

Hudson, B.D. 1994. Soil organic matter and available water capacity. J. Soil Wat. Con., 49(2): 189 – 194.

Jaffrain, J., F. Gérard, M. Meyer, and J. Ranger. 2007. Assessing the quality of dissolved organic matter in forest soils using ultraviolet absorption spectrophotometry. Soil Sci. Soc. Am. J., 71: 1851 – 1858.

Janos, P., S. Kříženecká and L. Madronová. 2008. Acid-base titration curves od solid humic acids. Reactive & Funtional Polymers, 68: 242 – 247.

Kaschl, A., V. Romheld, and Y. Chen. 2000. The influence of soluble organic matter from municipal solid waste compost on trace metal leaching in calcareous soils. Sci. Tot. Environ., 291: 45 – 57.

Kayuki, K.C. and C.S. Wortmann. 2001. Plant materials for soil fertility management in subhumid tropical areas. Agron. J., 93: 929 – 935.

Kockba, M., G. Ritvo, and Y. Avinimeleck. 2004. The effect of municipal solid waste compost (MSW) on the replacement of sodium in sodic soil models. Soil Sci., 169: 567 – 572.

Lazarovits, G. 2001. Management of soil-borne plant pathogens with organic soil amendments: A disease control strategy salvaged from the past. Canadian Journal of Plant Pathology, 23: 1 – 7.

Martens, D.A. and W.T. Frankenberger. 1992. Decomposition of bacterial polymers in soil and their influence on structure. Biol. Fertil. Soils, 13: 65 – 73.

Mayhew, L. 2004. Humic substances in biological agriculture. Acres-USA, 34 (1).

Moorman, T.B., J. K. Cowan, E.L. Arthur, and J.R. Coats. 2001. Organic amendments to enhance herbicide biodegradation in contaminated soils. Biol. Fertil. Soils, 33: 541-545.

Nethra, N.N., K.V. Jayaprasad, and R.D. Kale. 1999. China aster [Callistephus chinensis (L.) Ness] cultivation using vermicomposts as organic amendment. Crop Res. Hissar, 17: 209 – 215.

Oades, J. M. 1995. Recent advances in agronomineral Interactions: Implications for carbon cycling and soil structure. In: Huang, P.M. (Eds.). Environmental impact of soil component interactions, Vol. 1: Natural and Antropogenic Organics, Lewis Publisher, Boca Raton, FL, pp: 119 – 134.

Olk, D.C. 2006. A chemical fractionation for structure-function relations of soil organic matter in nutrient cycling. Soil Sci. Soc. A. J., 70: 1013 – 1022.Siva, K.B., 1997. Use of palm oil mill effluent (POME) and peat to reduce ammonia volatilization from fertilizer urea. Masters Thesis. Serdang, Malaysia, Universiti Putra Malaysia. pp: 41.

Omay, A.B., C.W. Rice, L.D. Maddux, and W.B. Gordon. 1998. Corn yield and nitrogen uptake in monoculture and in rotation with soybean. Soil Sci. Soc. Am J., 62: 1596 – 1603.

Ozkutlu, F., B. Torun, and I. Cakmak. 2006. Effect of zinc humate on growth of soybean and wheat in zinc-deficient calcareous soil. International Symposium on Soil and Plant Analysis, 37 (15 – 20): 891 – 901.

Post, W.M. 2006. Organic matter: Global distribution in world ecosystems. In: Lal, R. (Ed.). Encyclopedia of Soil Science. Vol. 2. Taylor & Francis, New York, pp: 1216 – 1221.

Reddy, G.B., and K.R. Reddy. 1993. Fate of nitrogen-15 enriched ammonium nitrate applied to corn. Soil Sci. Soc. Am. J., 57: 111 – 115.

Rice, C.W. 2006. Organic matter and nutrient dynamics. In: Lal, R. (Ed.). Encyclopedia of Soil Science. Vol. 2. Taylor & Francis, New York, pp: 1180 – 1183.

Robertson, E.B., S. Shlomo, and M.K. Firestone. 1991. Cover crop management of polysaccharide-mediated aggregation in an orchard soil. Soil Sci. Soc. Am. J., 55: 734 – 739.

Sladky, Z. 1959. The effect of extracted humus substances on the growth of tomato plants. Biol. Plant. (Prague), 1: 142 – 150.

Šmídová, M. 1960. The influence of humic acid on the respiration of plant roots. Biol. Plant. (Prague), 2: 152 – 164.

Spaccini, R., A. Piccolo, G. Haberhauer, M. Stemmer, and M.H. Gerzabek. 2001. Decomposition of maize straw in three European soils as revealed by DRIFT spectra of soil particle fractions. Geoderma, 99: 245 – 260.

Stevenson, F.J. 1986. Cycles of soil carbon, nitrogen, phosphorus, sulphur, micronutrients. John Wiley Wiley & Sons, New York, NY.

Stevenson, F.J., 1994. Humus chemistry: genesis, composition, reactions, 2nd edition. Wiley, New York, pp: 378 - 486.

Stone, A.G., S.J. Scheuerell, and H.M. Darby. 2004. Suppression of soilborne diseases in field agricultural systems: organic matter management, cover cropping, and other cultural practices. In: Magdoff, F., and R.R. Weil (ed.). Soil organic matter in sustainable agriculture. CRC Press LLC, Boca Raton, FL. pp: 131 – 177.

Sullivan, D.M. and R.O. Miller. 2000. Compost quality attributes, measurements, and variability. In: Stoffella, P.J., B.A. Kahn (Eds.). Compost utlization in horticultural cropping systems, Boca Raton, USA, Lewis Publishers, pp: 95 – 120.

Susilawanti Kasim, Osumanu H. Ahmed, Nik Muhamad A. Majid, Mohd K. Yusop and Mohamadu B. Jalloh. 2009. Reduction of Ammonia Loss by Mixing Urea with Liquid Humic and Fulvic Acids Isolated from Tropical Peat Soil. American Journal of Agricultural and Biological Sciences 4 (1): 18-23.

Susilawati, K., O.H. Ahmed, A.B. Nik Muhamad, and M. Y. Khanif, 2011. Effectiveness of liquid organic-nitrogen fertilizer in enhancing nutrients uptake and use efficiency in corn (Zea mays). American Journal of Biotechnology, 10 (12): 2274 – 2281.

Tan, K.H., 2003. Humic matter in soil and the environment: Principles and controversies. Marcel Dekker, Inc., New York, pp:34-71.

Weil, R.R. and F. Magdoff. 2004. Significance of soil organic matter to soil quality and health. In: Soil organic matter in sustainable agriculture. Weil, R.R. and F. Magdoff (Ed.). CRC Press, USA.

Whitebread, A.M., G.J. Blair, and R.D.B. Lefroy. 2000. Managing legume leys, residues and fertilizers to enhance sustainability of wheat cropping systems in Australia. 2. Soil physical fertility and carbon. Soil Tillage Res., 54: 77 – 89.

Wood, S., K. Sebastian, and S.J. Scherr. 2000. Pilot analysis of global agroecysytems. International Food Policy and World Resource Institute. http://www.wri.org/publication/pilot-analysis-global-ecosystems-agroecosystems.

Woomer, P.L. and M.J. Swift. 1994. The biological management of tropical soil fertility. Tropical Soil Biology and Fertility Programme and Sayce Publishing, John Wiley & Sons, New York.

5

Utilizing Palm Oil Mill Effluent Sludge as an Organic Amendment

Mohd Nizar, K., Isharudin, M.I., Abd. Jamil, Z., Hamdan, J., Syahrizan, S., Hazandy, A.H., Zolhalim, A.S., Wan Mohd Nazri, W.A.R. and Muhammad Esyam, A.

Introduction

The increasing demand of palm oil was driven by the increasing consumption of vegetable oil due to the growing human population (Corley, 2009). Due to this condition, palm oil production had grown rapidly and led to a growing concern about its environmental and ecological impacts. Increasing global demand for palm oil was mainly supplied by the increasing production in Malaysia (Lee, 2009) and it was likely to increase in coming years (Wahid *et al.*, 2008). The competition for palm oil usage between food, the feedstock for chemicals and biodiesel had put palm oil in the limelight and resulted in a controversial global debate (Reijnder & Huijbregts, 2008). The debate had revealed the importance of considering sustainability for palm oil production (Tan *et al.*, 2010).

In Malaysia, there were 4.69 million hectares of land planted with oil palm with a total annual processing capacity of 92.78 million tonnes of oil palm fresh fruit (MPOB, 2009; Tan *et al.*, 2010; Wan Razali *et al.*, 2012). According to Tan *et al.*, (2010), there were 406 currently active palm oil mills in Malaysia, which produced palm oil effluent mill (POME). POME was the most polluted organic residue generated from palm oil mill (Baharuddin *et al.*, 2010). POME was consisting high organic content mainly oil and fatty acids and able to support bacterial growth to reduce its pollution strength. Indeed, the anaerobic process was the most suitable approach for its treatment (Mumtaz *et al.*, 2008). Unfortunately, in Malaysia, the most popular method for POME treatment was the open pond system utilized by more than 85% of the mills due to low cost and economic value (Baharuddin *et al.*, 2010).

Current Status of Palm Oil Mills and Processes

Palm oil mill was a place where the oil is extracted. Significantly, every commercial plantation had its own mill to extract oil and released waste such as empty fruit bunches (EFB) and palm oil mill effluent (POME). A large area was needed for operating the mills starting from grading until producing the oil product and storage. Crude palm oil was produced from palm fresh fruit bunches (FFB) through major operational processes such as steaming and squeezing (Lam and Lee, 2011). It had been reported that processing/milling of one ton of FFB requires 1.5 m^3 of water, and 50% of the water used during milling was released as liquid waste known as palm oil mill effluent (POME) while the rest was lost as steam in the boiler blowdown, wash waters and leakage (MPOB, 2008). According to Lorestani (2006) reported that the oil palm mill in Malaysia is discharged about 50 million m^3 of POME each year. In general, the POME production mainly was low in pH and high concentration of grease, oil, protein, lipids, and minerals, which important for microbial activity for the decomposition process (Rupani *et al.*, 2010). This process was crucial for converting POME into an environmental friendly waste.

Palm Oil Mill Effluent (POME) Sludge

Palm oil mill effluent (POME) consists of suspended solids and dissolves solids which were left in the mills and discharged into the treatment ponds commonly named as palm oil mill effluent sludge. Therefore, the amount of sludge was increased significantly due to the large quantity of POME production each year (Khairuddin *et al.*,2016). POME sludge was consisting high nutrient value (Zakaria *et al.*, 1994). The contents of treated POME sludge was identified suitable and in accordance with the standard of WHO-ML for human consumption and safety in term of heavy metal elements and microorganism (Table 1). However, the sludge was having a bad odor and considered as a source of surface and ground pollution. Therefore, the industries player were focussed on cost-effective and sustainable technologies to dispose this industrial sludge. According to Khairuddin *et al*. 2016 in the recent studies showed that proper management and treatment processes of POME sludge in the ponds might utilize this abundant materials into the beneficial organic matter for plant consumption. Due to that, it is possible in the upcoming years equipped with proper treatment techniques and methodologies the abundant

POME sludge issue would be resolved as it was possible to utilize as an organic fertilizer. Comparison between final compost and POME mixing with empty fruit bunch (Wan Razali *et al.*, 2012), POME anaerobic sludge (Baharuddin *et al.*, 2010) and POME sludge (Khairuddin *et al.*,2016) were identified that the nutrients and heavy metals showed that there were insignificant difference. Nevertheless, the content of nitrogen, totally solid, volatile solid, ferrum, and calcium of POME sludge was higher rather than final compost and anaerobic POME sludge. In addition, the C/N ratio value showed that maturity stage of decomposition was below 15 which was already stable.

Table 1 Physicochemical analysis of POME sludge

Parameter	POME Sludge (Khairuddin *et al.*, 2016)	Final compost (Wan Razali *et al.*, 2012)	POME anaerobic sludge (Baharuddin *et al.*, 2010)
Carbon (%)	25.53	38.5	31.5
Nitrogen (%)	4.21	2.7	4.7
C/N ratio	6.35	13.8	6.7
Moisture content (%)	68.46	49.3	95.0
pH value	7.40	7.5	7.40
Total solid (%)	32.40	-	-
Volatile solid (%)	89.43	-	-
Heavy metal elements	POME Sludge (Khairuddin *et al.*, 2016)	Final compost (Wan Razali *et al*, 2010)	WHO-ML Standard
Copper (mg/kg)	45.05 $\pm$ 2.87	70.40 $\pm$ 21.60	75.00
Chromium (mg/kg)	27.86 $\pm$ 0.55	9.30 $\pm$ 0.20	150.00
Cadmium (mg/kg)	0.41 $\pm$ 0.01	4.10 $\pm$ 0.50	1.90
Zinc (mg/kg)	130.11 $\pm$ 3.49	90.70 $\pm$ 10.00	140.00
Plumbum (mg/kg)	0.38 $\pm$ 0.10	4.20 $\pm$ 1.60	0.30
Nickel (mg/kg)	10.77 $\pm$ 0.15	n.d	67.00
Manganese (mg/kg)	422.56 $\pm$ 12.04	250.40 $\pm$ 25.10	500.00

cont'd Table 1

Nutrient elements	POME Sludge (Khairuddin *et al.*, 2016)	Final compost (Wan Razali *et al*, 2010)
Ferrum (%)	2.24 ± 0.02	1.20 ± 0.30
Potassium (%)	0.03 ± 0.01	0.03 ± 0.20
Calcium (%)	1.67 ± 0.04	0.70 ± 0.20
Magnesium (%)	0.55 ± 0.02	1.00 ± 0.10
Phosphorus (%)	0.08 ± 0.01	1.30 ± 0.20
Sulfur (%)	0.30 ± 0.01	1.20 ± 0.40

WHO-ML; Codex Alimentarius Commission (FAO/WHO). Food additives and contaminants. Joint FAO/WHO Food (2001)

According to Chan and Chooi (1984), palm oil mill effluent sludge was possible to be dried and used as a fertilizer due to it's high nutrients content. Drying was conducted in the open ponds, but during the raining seasons, this process was disrupted due to the slow rate of drying and overflow problem. . In the present study, the treated POME sludge was used as soil amendment for maize growth due to it's high nutrients content after adopting some procedures in the open treatment ponds before it was applied as soil amendment.

Potential Usage of POME Sludge in Agriculture

Usage of organic waste on land was necessary for sustainable agriculture (Silva, *et al.*, 2010). Furthermore, utilizing POME sludge might increase soil quality and environment as well as economic performances. On the other hand, application of organic matter based on oil palm waste such as empty fruit bunch (EFB), decanter cake and POME in agriculture had been well utilized (Embrandiri, *et al.*, 2013). However, the actual practices was different where the POME sludge was still left abundantly and underutilized by the industrial players. Proper management and treatments of POME sludge might become possible for the plantation to utilize it as fertilizer. According to Aroujo *et al.* (2009), the main intention was finding a solution in an ecological way to use POME sludge without environmental risks and recycling it as an organic waste. Meanwhile, the potential of organic waste with high content of organic matter and nutrients could be used as fertilizer for the plants (Singh, *et al.*, 2015; Melo, *et al.*, 2007). Nevertheless, the important characteristics of sludge was its nutrient quality, C/N ratio, trace elements, pH, moisture, and heavy metals. Indeed, these could give significant effect on soil nutrients and plant performances (Singh *et al.*, 2015). In the recent studies by Khairuddin *et al.*, (2016), proper management and treatment techniques the POME sludge was able to be converted into an organic amendment (treated POME sludge) for plant establishment. Significant results were produced on the effect of soil properties and plant growth after application of treated POME sludge (Khairuddin *et al.*, 2017).

Effects Of POME Sludge on Physicochemical and Soil Properties

Introduction

According to Gasim *et al.* (2011) recorded that soil was an important resource in term of land use for sustainable crops. Indeed, it's determined the healthy ecosystem promoted by the soil quality indicators (Arshad and Martin, 2002; Gasim *et al.*, 2011). The deficiency in soil might also contribute to less production of crops.

Physico-chemical Properties

POME sludge contains 11.1% crude protein, 12% crude lipids (ether extract), 17% crude fiber, 9% ash and 50.5% carbohydrates (N-free extract) on a dry matter basis, making it became useful (Ahmed, 2009). Apart from the organic composition, POME was also rich in mineral content, particularly phosphorus, potassium, magnesium, and calcium (Ahmad *et al.*, 2003, Okwute & Isu, 2007). Thus, most of the dewatered POME (sludge) would be recycled or returned to the plantation as fertilizer, however, the application must be done correctly to avoid overdose which might result in nutrient imbalance and led to an undesirable chemical reactions in the soil. Underutilized of POME might cause the accumulation of magnesium and inhibited the availability of potassium in the soil. It was evidence indicating that utilization of POME was limited by the potassium (or magnesium) value (Kittikun *et al.*, 2000).

Generally, the POME was found high in heavy metals but in this study, heavy metal content was important to determine for the safety used of human consumption, especially as fertilizer. Furthermore, the value of the heavy metals in treatment pond system was found below the WHO-ML standard (Khairuddin *et al.* 2017). The heavy metals present in the POME were cadmium, plumbum, mercury, chromium, nickel, and manganese. According to Bigdeli and Seilsepour (2008), cadmium was a highly mobile metal, easily absorbed by the plants through root surface and moved to wood tissue and transferred to the upper parts of the plants. The main source of cadmium in the air was from the burning fossil fuels such as coal or oil which was also similar to the incineration process by the municipal waste (EPA, 2000).The treated POME sludge was contained heavy metals below the WHO-ML standard and safe to use as fertilizer (Khairuddin *et al*. 2017).

Soil chemical properties

According to Embrandiri *et al*. (2013) reported that palm oil mill effluent (POME), palm kernel cake, decanter cake, empty fruit bunch and palm kernel shell were potential to use and processed as an organic amendment for land application. Recently, some studies were using decanter cake with lady's finger (Embrandiri *et al*., 2013), tannery sludge with capsicum plants (Silva *et al*., 2010), cowpea (Silva *et al*., 2013) and olive mill waste with lettuce and chicory (Kelepsei and Tzortzakis, 2009) as their organic matter materials for soil amendments. Unfortunately, only a few studies were utilized the POME sludge as a soil amendment for the plant growth. According to Khairuddin *et al*. (2016), the treated POME sludge was identified equally balance with C, N, and macronutrients (P, K, Ca, Mg, S, Na). Mostly, these elements were vital in soil fertility and able to increase productivity of the crops. Moreover, heavy metals content in POME sludge were determined in accordance with the standard of WHO/FAO that safe for human consumption. POME sludge increased organic matter content, decreased bulk density and provided more nutrients to the soil. According to Hao *et al*. (2008), stated that the application of organic amendments generally elevates the soil organic matter and microbial biomass contents to a much greater extent. Indirectly, it could also assist in producing high yield as well as the better growth of the plant.

Soil physical properties

Application treated POME sludge might enhance soil physical properties and improved fertility status of the soil (Khairuddin *et al*. 2016). According to Dexter (1988), organic matter content reduced the soil bulk density and soil compaction. El-Shakweer *et al*. (2008) was also reported that the organic matter elevated soil porosity and infiltration rate. Hati and Bandyoopadhay (2011) was also reported that the addition of organic matter increased aggregation and pore volume and showed a positive effect on the saturated hydraulic conductivity of the soil. Improvement of soil physical and hydraulic properties would increase crop yield and quality. The plant available water (PAW) was determined by the different of water content between field capacity and permanent wilting point in the soil. In addition, the plant available water might assist the efficiency of nutrients uptake in the soil by the plant (Karlen *et al*., 1994).

Effects of POME Sludge Application on Plant Growth

Introduction

Growth morphology and performance was important to determine the potential biomass of the plant that affected the used of various fertilizer applications (Silva *et al*, 2010). Indeed, a similar effect was shown by organic fertilizer in the growth media (Singh and Agrawal, 2009; Silva *et al*, 2010). Some studies were shown the significant growth of shoot, root and total biomass of plants such as capsicum (Silva *et al*., 2010), lady finger (Singh and Agrawal, 2009), Chinese cabbage (Wong, 1996), sesame plant (Gupta and Sinha, 2006) and beetroot (Aggarwal and Goyal, 2007). Gas exchanges properties were important to identify the needs of plant growth (Box, 1996). Moreover, there was important to observe the interaction between physicochemical needs with the gas exchange parameter including net photosynthesis (A_{net}), stomatal conductance (G_s), internal CO_2 concentration (C_i), and transpiration rate (E_L).

Growth morphology and performance

The effect of treated POME sludge on the morphology and growth parameter was vital to observe. Morphology and growth parameter was observed and measured such as plant height, diameter, leaf number, leaf area, root density, ears length, ears diameter, ears biomass, leaf biomass, shoot biomass and root biomass. Morphological analysis revealed that the performance of organic amendment positively affects the plant growth.

Organic amendment based on oil palm had shown as the suitable substitute of inorganic fertilizers (Embrandiri *et al*., 2013). The treated POME sludge showed a positive effect on the crop, due to the high nutrients content in the organic matter. All the measurements were directly influenced the physical growth of maize (Khairuddin *et al*. 2016).

Plant biomass was derived from the plant material that included leaves, brunches, barks, woods, and roots. Dry matter was the basic fundamental aspect of identifying the importance of biomass for the plant. These could be observed from the previous research publication on plant physiology. According to Silva *et al*. (2010) reported that total biomass was able to justify the effectiveness of the plant growth in any treatments in the plant trial. According to Hunt (1982), the growth parameter was formulated by correction of leaf area ratio (LAR) (1), specific leaf area (SLA) (2), leaf weight ratio (LWR) (3) and the root-shoot ratio (RSR) (4). Below are the details:

$$\text{LAR (cm2 g-1)} = \frac{\text{Leaf area}}{\text{Total biomass}} \qquad \text{Eq. 1}$$

$$\text{SLA (cm2 g-1)} = \frac{\text{Leaf area}}{\text{Leaf biomass}} \qquad \text{Eq. 2}$$

$$\text{LWR (cm2 g-1)} = \frac{\text{Leaf biomass}}{\text{Total biomass}} \qquad \text{Eq. 3}$$

$$\text{RSR (cm2 g-1)} = \frac{\text{Root biomass}}{\text{Shoot biomass}} \qquad \text{Eq. 4}$$

Gas exchange properties

According to Abdul-Hamid *et al*. (2011) the measurement of net photosynthesis (A_{net}), stomatal conductance (G_s), internal CO_2 concentration (C_i), and transpiration rate (E_L) were used to measure the respond of plants towards the environment. According to Silva *et al*. (2010), gas exchange content indicates the effectiveness of plant could obtain nutrients for the growth. An organic fertilizer from tannery sludge (Silva *et al*. 2010), POME decanter cake (Embtandiri *et al*., 2013), and POME sludge – humic and fulvic acids (Mia and Shamsuddin, 2010) reported an increase in gas exchange and chlorophyll formation at growing stage.

Additionally, chlorophyll content had also increased the pigment content in the plant. Meanwhile, there were no negative effects on plant growth after application of organic sludge (Silva *et al*., 2010).

According to Abdul-Hamid *et al*. (2011), there was a positive linear between photosynthesis clean (A_{net}) and stomatal conductance (G_s) was obviously related to the environment and leaves morphology. In addition, the correlation could be modulated in the long term with the growth conditions such as nitrogen, light environment, and high CO_2 concentrations (Wong *et al*., 1979). The combination pattern of stomata-photosynthesis model was the cause of the changes in physiological capacities of photosynthesis and patchy stomatal closure (Uddling *et al*., 2005) and low leaf water potential (Kramer and Boyer, 1995 and Abdul-Hamid *et al*., 2011). Furthermore, the abundance of CEC in soil affected the stomata conductivity for the gas exchanges on the guard cells and epidermal turgor in the cell (Franks *et al*., 2001).

Effect of POME Sludge Application on Root Length Density and Shear Strength

Plant root system is important in the interaction with the soil that contributes more than shoot system. According to Hedley *et al*. (2010) reported that evaluations of plant root systems are particular limits with the soil matrix. However, analysis on the root length density (RLD) could possibly determine the potential production of the plants. The root length density (RLD) highlighted the reliability to sequester certain element such as carbon in the soil properties (Hedley *et al*., 2010). Indeed, the RLD had the potential to increase water use efficiency by optimizing the used of subsoil water and recapturing the nitrate (Besharat *et al*., 2010).

Soil strength was derived from direct shear test method which the frictional resistance met by the soil constituent particles when they were forced to slide over one another or to move out of interlocking positions (Khairuddin *et al*., 2017). The shear strength extent to which stresses or forces were absorbed by solid-to-solid contact among the particles, cohesive forces related to chemical bonding of clay minerals, and surface tension forces within the moisture films in unsaturated soils (Isa *et al*., 2017). The presence of roots resulted in an overall increase of the soil strength. According to Khairuddin *et al*. (2017), the treated POME sludge was significantly increased soil shear strength with low cohesion value and high angle of internal friction contributed by the root densities. This arises from the combined effects of soil reinforcement by a mass of roots (Mohamad Nordin *et al.*, 2011) and soil moisture depletion by evapotranspiration (Ali and Osman, 2008). Other factors, which affected the soil strength was soil moisture content, particle size distribution (soil texture), and mineralogical content of the different soil series.

Generally, the graph (Fig. 1) showed that the shear strength was obtained after the application of normal stress at 100 kN m^{-2}, 200 kN m^{-2} and 300 kN m^{-2} in all samples. The peak strength or shear strength was plotted for the different POME sludge treatments that were used in the experiment. The different types of POME sludge were differentiate based on maturity stage of decomposition in the treatment ponds. Relatively, the dumping pond (DP) treatment was low cohesion, c, while the angle of internal friction, ϕ, increased and the shear stress at failure was obtained from the shear strength equation. The maximum shear strength in DP treatment was 500.72 kN m^{-2}, ϕ was 30^{O} and the root length density (RLD) was observed at 334.16 g m^{-3} (Table 2). The treatment with high root length density (RLD) in algae pond (ALP) with 271.7 g m^{-3} showed the decreased of cohesion value to 135 and ϕ value was slightly increased. The maize treated with DP treatment indicated high shear strength compared to all the treatments (Table 2). According to Ali and Osman (2008) reported that roots system and roots length density (RLD) increased the shear strength of the soil. The cohesion value in control treatment was slightly lower than the other treatments. While the cohesion value (c) was low, the ϕ value tends to increase.

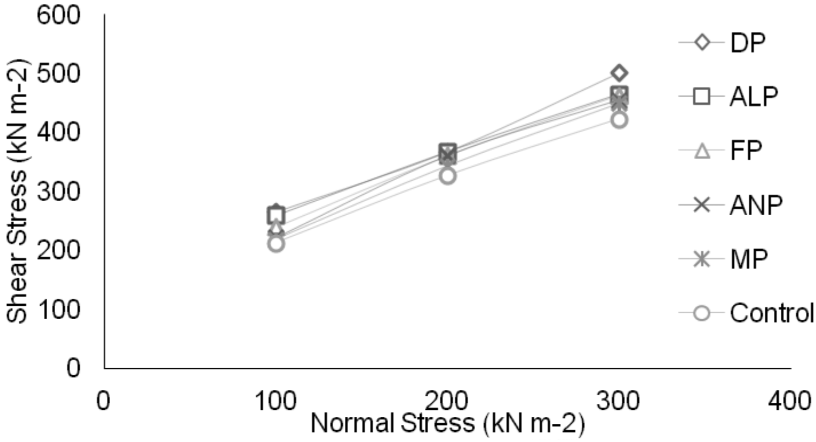

Figure 1 Relationship of shear stress and normal stress for POME sludge treatment (control, MP, ANP, FP, ALP, and DP) (Khairuddin *et al.*, 2017)

Table 2 Soil and POME sludge treatment indicated root density and shear strength parameter of Entisols (Khairuddin *et al.*, 2017)

Treatment	Root density (cm cm^{-3})	Cohesion (c)	Angle of Internal Friction (ϕ)	Shear strength (kN m^{-2}) (τ)
Control	75.53	140	25^{O}	422.95
MP	143.69	135	26^{O}	449.67
ANP	203.02	130	27^{O}	454.39
FP	258.24	120	28^{O}	462.83
ALP	271.7	100	29^{O}	464.14
DP	334.16	80	30^{O}	500.72

Note: MP sludge (Mixing ponds sludge), ANP sludge (Anaerobic ponds sludge), FP sludge (Facultative ponds sludge), ALP sludge (Algae ponds sludge) and DP sludge (Dumping ponds sludge)

Effect of POME Sludge on Microbial Present

Microbial in POME sludge was vital in the digestion process. This study had identified the present of anaerobic bacteria such as methanogenic bacteria (A), acetogenic bacteria (B) and acidogenic bacteria (C) (Fig. 2 and Fig. 3). According to Mata-Alvarez (2003) that these bacterial activities were important in the decomposition process. According to Kankal, 2012, the anaerobic breakdown or digestion was assisted by the methanogenic bacteria.

(a) (b)

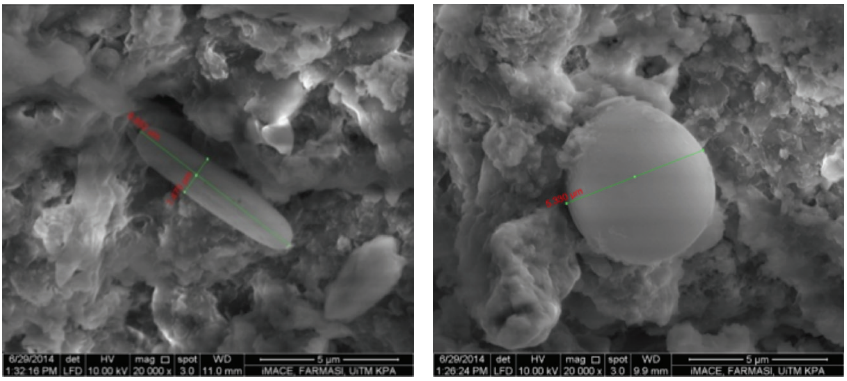

Figure 2 FESEM micrograph (Magnification 20000x) treated POME sludge structure with acetogenic and methanogenic bacteria in dumping pond. (Khairuddin *et al*., 2016)

(c)

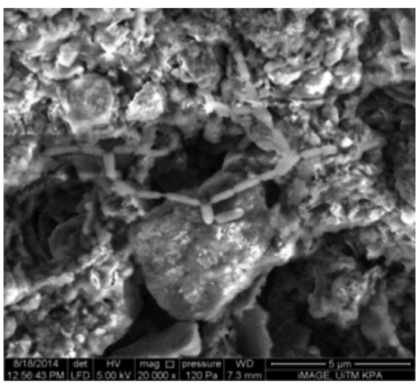

Figure 3 FESEM micrograph (Magnification 20000x) detected colonization of acidogenic bacteria in treated POME sludge. (Khairuddin *et al*., 2016)

Conclusion

Treated POME sludge was potential and safe to use for the plants as an alternative of soil organic amendments. The results had proven that treated POME sludge was low heavy metals and rich in nutrients content required by the plant growth. Adopting the appropriate management and treatment processes in the mill treatment pond was vital in order to produce good quality of organic amendments.

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References

Abdul-Hamid, H., Abdu, A., Ismail, M. K., & Senin, A. L. (2011). Gas Exchange of Three Dipterocarp Species in a Reciprocal Planting. Asian Journal of Plant Sciences.

Aggarwal, H., & Goyal, D. (2007). Phytoremediation of some heavy metals by agronomic crops. Developments in environmental science, 5, 79-98.

Ahmad, A., Ismail, S. and Bhatia, S. (2003).Water recycling from palm oil mill effluent (POME) using membrane technology.Desalination, 157(87): 95-101.

Ali, F. H., and N. Osman. (2008). Shear strength of a soil containing vegetation roots. Soils and Foundations, 48(4): 587-596.

Araujo, A.S.F., Melo, W.J., Santos, J.A. and Melo, V.P. (2009). Use of municipal and in-dustrial sludge in agricultural soil: responses of soil microbial and soil enzymes. In: Samuelson JP (Org.). Industrial waste: Environmental Impact, Disposal and Treatment. 1ed. New York: Nova Science: 349-367.

Arshad, M.A. and Martin, S. (2002). Identifying critical limits for soil quality indicators in agroecosystem. Agric. Ecosyst. Environment, 88: 153-160.

Baharuddin, A.S., Lim, S.H., Md Yusof, M.Z., Abdul Rahman, N.A., Md. Shah,U.K., Hassan, M.A., Wakisaka,M., Sakai, K. and Shirai, Y. (2010). Effect of palm oil mill ef-fluent (POME) anaerobic sludge based compost using fourier transform infrared (FTIR) and nuclear magnetic resonance (NMR) analysis. African Journal Biotechnolo-gy, 10(41): 2427-2436.

Besharat,S., Nazemi,AH. and Sadraddini, AA. (2010). Parametric Modeling of Root Length Density and Root Water Uptake in Unsaturated Soil. Turk Journal Agriculture For;34, 439-449.

Bigdeli, M., and Seilsepour, M. (2008). Investigation of Metals Accumulation in Some Vegetables Irrigated with Waste Water in Share Rey-Iran and Toxicological Implications.American-Eurasian J. Agric. & Environ. Sci., 4 (1): 86-92.

Box, E. O. (1996). Plant functional types and climate at the global scale. Journal of Vegetation Science, 7(3), 309-320.

Chan, K.S. and Chooi, C.F. (1984). Ponding system for palm oil mill effluent treatment.In: Proceedings of the Regional Workshop on Palm Oil Mill Technology & Efflu-ent Treatment: 185-192.

Codex Alimentarius Commission (FAO/WHO). (2001). Food additives and contaminants. Joint FAO/WHO Food, Standards Program; ALINORM 01/12A: 1-289.

Corley, R.H. (2009). How much palm oil do we need? Environment Science Policy 12(2):134-136.

Dexter, A. R. (1988). Advances in characterization of soil structure. Soil and tillage research, 11(3-4), 199-238.

El-Shakweer, M. H. A., El-Sayad, E. A., & Ewees, M. S. A. (2008). Soil and plant analysis as a guide for interpretation of the improvement efficiency of organic conditioners added to different soil in Egypt. Common. Soil Sci. Plant Anal., 29(11-14), 2067-2088.

Embrandiri,A., Singh, R.P. and Ibrahim, M.H. (2013). Biochemical, morphological, and yield responses of lady's finger plants to varying ratios of plam oil mill waste (decanter cake) application as a bio-fertilizer. International Journal of Recycling of Organic Waste in Agriculture, 2: 1-7.

Environment Protection Agency (EPA). (2000). Wastewater Technology Fact Sheet: Anaerobic Lagoons. Reviewed on 20 April 2017 at http://www.epa.gov/owm/mtb/mtbfact.htm

Franks, P. J., Buckley, T. N., Shope, J. C., & Mott, K. A. (2001). Guard cell volume and pressure measured concurrently by confocal microscopy and the cell pressure probe. Plant Physiology, 125(4), 1577-1584.

Gasim, M. B., Ismail, B. S., Mir, S. I., Rahim, S. A., & Toriman, M. E. (2011). The physi-co-chemical properties of four soil series in Tasik Chini, Pahang, Malaysia. Asian Jour-nal of Earth Sciences, 4(2): 75.

Hao, X. H., Liu, S. L., Wu, J. S., Hu, R. G., Tong, C. L., & Su, Y. Y. (2008). Effect of long-term application of inorganic fertilizer and organic amendments on soil organic matter and microbial biomass in three subtropical paddy soils. Nutrient Cycling in Agroecosystems, 81(1): 17-24.

Hati, K., & Bandyoopadhay, K. (2011). Fertilizers (mineral, organic), effect on soil physical properties. Encyclopedia of Agrophysics (pp. 296-299). Springer Netherlands.

Hedley, M., Kusumo,B., Hedley, C. and Tuohy, M. (2010). Field Measurement of Root Density and Soil Organic Carbon Content Using Soil Spectral Reflectance. 19th World Congress of Soil Science, Soil Solutions for a Changing World. Brisbane, Australia.

Hunt, R. (1982). Plant growth curves, the functional approach to plant growth analysis. London, Edward Arnold: 248.

Isa, I. Shamshuddin, J. and Khairuddin, M.N. 2017. Effects of mix vegetation and root shear strength grown on carbonaceous shale. Asian J. Applied Sci., 11: 1-13.

Kankal, N.C., Dhadse, S. and Kumari, B. (2012). Study of diverse methanogenic and non-methanogenic bacteria used for the enhancement of biogas production. Int.Journal Life Science Biotechnology& Pharma Research, 1(2).

Karlen, D. L., & Stott, D. E. (1994). A framework for evaluating physical and chemical indicators of soil quality. Defining soil quality for a sustainable environment (definingsoilqua):53-72.

Kelepesi, S., and Tzortzakis, N. G. (2009). Olive mill wastes—a growing medium component for seedling and crop production of lettuce and chicory. International journal of vegetable science, 15(4), 325-339.

Khairuddin, M. N., Isharudin M. I., Abd Jamil, Z., Hamdan, J. and Syahrizan, S. (2017). Shear Strength and Root Length Density Analyses of Entisols Treated with Palm Oil Mill Effluent (POME) Sludge. Soil & Environment 36(1): 90-114.

Khairuddin, M. N., Zakaria, A. J., Isa, I. M., Jol, H., Rahman, W. M. N. W. A., & Salleh, M. K. S. (2016). The Potential of Treated Palm Oil Mill Effluent (Pome) Sludge as an Organic Fertilizer. AGRIVITA, Journal of Agricultural Science, 38(2), 142-154.

Kramer, P. J., & Boyer, J. S. (1995). Water relations of plants and soils. Academic press.

Lam, M. K., & Lee, K. T. (2011). Renewable and sustainable bioenergies production from palm oil mill effluent (POME): win–win strategies toward better environmental protection. Biotechnology Advances, 29(1), 124-141.

Lee, K.T. (2009). Life cycle assessment of palm oil diesel.Application Energy 8 (6): 189

Lorestani, A. A. Z. (2006). Biological treatment of palm oil mill effluent (POME) using an up-flow anaerobic sludge fixed film (UASFF) bioreactor (Doctoral dissertation, USM).

Mata-Alvarez, J. 2003. Biomethanization of the organic fraction of municipal solid wastes. IW Publication.

Melo, W.J., Aguiar, O.S., and Melo, V.P. (2007). Nickel in a tropical soil treated with sewage sludge and cropped with maize in a long-term field study. Soil Biol Biochem ;39: 1341- 1347.

Mohamad Nordin, A., O., Normaniza, and H.A., Faisal, 2011. Soil-root shear strength properties of some slope plants. Sains Malaysiana 40(10): 1065–1073.

Malaysian Palm Oil Board (MPOB). (2009). Malaysia oil palm statistic. Reviewed on 27 Feb 2017 at http://www.econ.mpob.gov.my/economy/annual/stat2009/area1_1.pdf

Malaysian Palm Oil Board (MPOB). (2008). Bio-fertilizer from palm oil biomass and POME solids by mobile composter.MPOB TT 381. Reviewed on 29 Jan 2017 at http://palmoilis.mpob.gov.my/publications/TOT/TT-381.pdf

Mia, M. B., & Shamsuddin, Z. H. (2010). Rhizobium as a crop enhancer and biofertilizer for increased cereal production. African journal of Biotechnology, 9(37), 6001-6009.

Mumtaz, T., Abd-aziz, S., Abdul Rahman, N.A., Phang, L.Y., Shirai,Y. and Hassan, M.A. (2008). Pilot-scale of recovery of low molecularweight organic acids from anaer-obically treated palm oil mill effluent (POME) with energy integrated system. African Journal Biotechnology, 21: 3900-3905.

Okwute, L.O. and Isu, N.R. (2007).The environmental impact of palm oil mill effluent (POME) on some physicochemical parameters and total aerobic bioload of soil at a dump Site in Anyigba, Kogi State, Nigeria. African Journal of Agricultural Research, 2 (12): 656-662.

Reijnders, L., and Huijbregts, M. A. J. (2008). Palm oil and the emission of carbon-based greenhouse gases. Journal of cleaner production, 16(4): 477-482.

Rupani, P.F., Singh, R.P., Ibrahim, M.H. and Esa, N. (2010). Review of current palm oil mill effluent (POME) treatment methods, vermicomposting as a sustainable practice. World Applied Sciences Journal 11(1): 70–81.

Silva, M.D., Ferreira Araújo, A. S., Pinheiro Leal Nunes, L. A., de Melo, W. J., & Pratap Singh, R. (2013). Heavy metals in cowpea (Vigna unguiculata L.) after tannery sludge compost amendment. Chilean journal of agricultural research, 73(3), 282-287.

Silva, J. D., Leal, T. T., Araújo, A. S., Araujo, R. M., Gomes, R. L., Melo, W. J., and Singh, R. P. (2010). Effect of different tannery sludge compost amendment rates on growth, biomass accumulation and yield responses of Capsicum plants. Waste Management, 30(10), 1976-1980.

Singh,R.P, Sarkar,A. Sengupta, C., Singh,P, Miranda, A.R.L, Nunes, L.A.P.L., Ferreira de Araujo, A.S. and José de Melo, W. (2015). Effect of utilization of organic waste as agricultural amendment on soil microbial biomass. Annual Research & Review in Biol-ogy 7(3): 155-162

Singh, R.P., and Agrawal, M. (2009). Use of sewage sludge as fertilizer supplement for abelmoschus esculentus plants: physiological, biochemical and growth responses. Inter-national Journal of Environmental Waste Management 3: 91–106.

Tan, K.T., Lee, K.T., Mohamed, A.R., and Bhatia, S. (2010). Palm oil: addressing issues and towards sustainable development. Renewable Sustainable Energy Review 13(2): 420.

Uddling, J., Hall, M., Wallin, G., & Karlsson, P. E. (2005). Measuring and modelling stomatal conductance and photosynthesis in mature birch in Sweden. Agricultural and Forest Meteorology, 132(1), 115-131.

Wahid, M.B.S., Balu, N., and Ismail N.(2008). EU's renewable energy directive: possible implications on Malaysian palm oil trade. Oil Palm Ind. Economic Journal (8): 1.

Wan Razali, W.A., Baharuddin, A.S., Talib, A.T., Sulaiman, A., Naim, M.N., Hassan, M.A. and Shirai, Y.,(2012). Degradation of oil palm empty bunches (OPEFB) fibre during composting process using in-vessel composter. BioResources 7(4): 4786-4805.

Wong, J.W., Li, G. and Wong, M.H. (1996). The growth of Brassica chinensis in heavymetal contaminated sewage sludge compost from Hong Kong. Bioresource Technology 58: 309–313.

Wong, S. C., Cowan, I. R., & Farquhar, G. D., 1979. Stomatal Conductance Correlates with Photosynthetic Capacity. Nature 282

Zakaria, Z.Z., Khalid,H., and Hamdan, A.B. (1994). Guidelines on land application of palm oil mill effluent (POME). PORIM Bulletin Palm Oil Research Institute Malaysia: 28.

6

Greenhouse Gas Emission from Pineapple Cultivation on Drained Tropical Peat Soils

Liza Nuriati Lim Kim Choo, Osumanu Haruna Ahmed, Mohamadu Boyie Jalloh and Hamdan Jol

Summary

Pineapple (*Ananas comosus* (L.) Merr.) cultivation on tropical peats could affect the release of carbon dioxide, methane, and nitrous oxide emissions into the atmosphere once the peat is drained. The contribution of soil greenhouse gases from pineapple cultivation is important as pineapples are mainly grown in the peat soils of Malaysia. Pineapple is one of the main export commodities besides oil palm that contributes to the nations' economic growth in the agriculture sector. It is essential to quantify soil greenhouse gas emissions before deciding whether pineapple cultivated peats are net sink or net source of atmospheric greenhouse gases. At present, information on soil greenhouse gas emission in pineapple cultivation on tropical peat soils is scarce. This chapter provides general perspective on tropical peatlands that includes the occurrence of tropical peat soils, its classification, characteristics, and cultivation trend. Factors influencing carbon dioxide, methane and nitrous oxide emissions from pineapple cultivated peats are also discussed. Soil greenhouse gas emission measurement approach and the need to partition soil respiration from cultivated peat soils are also included in this chapter.

Keywords: Greenhouse gases, organic soils management, land degradation, peatland, pineapple

Introduction

Peat soils in Malaysia are rapidly developed for large-scale agriculture namely oil palm and pineapple due to their positive contribution to the nation's engine of growth in the agriculture sector. However, drainage that causes water level drawdown of peat soil following land clearing for agriculture accelerates peat organic matter decomposition (Berglund and Berglund, 2011). This leads to substantial loss of carbon reserve and release of greenhouse gases namely carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) into the atmosphere (Kløve *et al.,* 2010). Carbon dioxide, N_2O, and CH_4 from peat soil are emitted through aerobic and anaerobic microbial respiration, root respiration, peat oxidation, nitrification, and denitrification. The emission of peat greenhouse gases is said to be influenced by land use type (Ismail, 2010), peat type (Kechavarzi *et al.,* 2010), temperature (Jauhiainen *et al.,* 2012), fertilization (Hadi *et al.,* 2005), water table depth (Chimner and Cooper, 2003), and photosynthetic activities (Mäkiranta *et al.,* 2008). Although attempts have been made to measure greenhouse gases from cultivated tropical peats, such studies are limited to forest ecosystem, oil palm, paddy, rice-soybean, sago, and pulpwood (Inubushi *et al.,* 2003; Hadi *et al.,* 2005; Melling *et al.,* 2005; Watanabe *et al.,* 2009; Jauhiainen *et al.,* 2012).

At present, there is dearth of information on greenhouse gas emission from pineapple cultivation on drained peat soils. The understanding and contribution of pineapple cultivation on peats to greenhouse gases is important, as 90% of pineapples are grown on peat soils in Malaysia (Raziah and Alam, 2010). Malaysia is one of the top fifteen leading fresh pineapple exporter in the world (Raziah, 2009) with an export value for canned pineapples worth at RM 37.2 million in 2011 (MPIB, 2015). From 2006 to 2010, the pineapple industry in Malaysia has been reviving. This export earning has been improving after a lapse during the 8th Malaysia Plan (2001 to 2005). The lapse was because of the decline in crop yield that was caused by attacks from nematode *Paratylenchus* species (Nik Masdek, 2007). Pineapple is unique as it is classified as C3 and C4 plant or Crassulacean Acid Metabolism (CAM) plant (Ritchie and Bunthawin, 2010) that may well give a different trend of greenhouse gas emission compared with crops such as oil palm that is widely planted and well studied on tropical peat soils. Pineapple absorbs CO_2 at night and keeps them in the form of acid in the leaves while emitting the gas during the day for it to be processed into carbohydrates to increase water efficiency. It is able to absorb CO_2

more efficiently with high temperature differences between day and night (Mohammed Selamat, 1996).

With the growing concern about the effects of greenhouse gases on environmental quality coupled with the need to achieve sustainable agriculture, there is a need to account for greenhouse gases from pineapple cultivation on peat soils to provide a basis for future emission factors. Failure to account for greenhouse gas emissions from pineapple cultivation on tropical peatlands could underestimate future rates of increase in atmospheric greenhouse gases and their effects on global environmental change processes.

General Perspective on Tropical Peatlands

Occurrence of tropical peat soils

Tropical peatlands occur in East Asia, Southeast Asia, Central America, South America, and Southern Africa (Rieley, 2008). In Southeast Asia, there are approximately 27.1 million hectares of peat soil (Hoojier *et al.,* 2010) that constitute about 6% of the total global peatland area (Jauhiainen *et al.,* 2012). In Malaysia, peatlands cover about 2.6 million hectares of the total land, and they are located mostly at the low-lying coastal areas with excess moisture and having a high annual rainfall (Safford and Maltby, 1998; Ritzema *et al.,* 2004; Ismail and Jamaludin, 2007).

Tropical peat soils are generally defined as soils formed by the accumulation of partially decayed woody plant materials under waterlogged condition. Peats are high in organic material with a minimum of 50 to 65% organic matter or 35% mineral content (Farmer *et al.,* 2011; Huat *et al.,* 2011), with thickness of 40 cm within the first 100 cm of the soil surface (Couwenberg, 2011), but may differ in location due to factors such as climate, temperature, organic content, and degree of decomposition (Andriesse, 1988; Mutalib *et al.,* 1991; Huat *et al.,* 2011).

Peat classification and characteristics

Peat soils can be classified according to their degree of decomposition and fibre content using the Von Posts Scale (H1 to H10). Based on the American Society for Testing and Materials (ASTM) standard, peat is classified into three categories namely fibric, hemic, and sapric peat (Melling, 2005; Huat *et al.,* 2011). Fibric peat is the least decomposed (scale of H1 to H4) with a fibre content of over 66%. Hemic peat is partially decomposed with a scale of H5 to H7, and fibre content ranging from 33 to 66%. The sapric is a highly decomposed peat with a Von Post Scale of H8 to H9, and less than 33% fibre content.

In terms of physical properties, tropical peat soils are characterized by low bulk density, low bearing capacity, high porosity, and high moisture retention capacity but with a tendency to subside once the soil is drained (Andriesse, 1988; Ma and Tobin, 2004; Huat *et al.,* 2011). They have a surface area that is greater than 200 m^2/g. Peat soils are also chemically defined as a naturally polyelectrolyte where their major components comprises of humic acids, lignin and carbohydrate (Helal Udin *et al.,* 2003; Allen *et al.,* 2004). Peat soils are generally acidic ranging from pH 3 to 4.5. They are also low in macro and micronutrients (Safford and Maltby, 1998). Peat soils have high carbon to nitrogen (C:N) ratio but exhibit high cation exchange capacity (CEC) which is essential for nutrient storage (Andriesse, 1988; MARDI, 1996; Kasimir-Klemedtsson *et al.,* 1997).

Cultivation of peat soils

Tropical peat soil is considered as a unique natural resource that is utilizable for a variety of purpose. Peat soils have been used for centuries for productive purposes especially agriculture and forestry, besides playing a vital role in ecosystem services including biodiversity conservation, carbon sequestration, and water regulation. Due to the increasing demand for land development, peat soils have been encroached, deforested, and drained for large-scale plantations such as oil palm, sago, acacia, and pineapple (Ismail, 2008). The cultivation of these plantation crops has the strategic potential to increase a nation's economic growth through the production of agricultural commodities for export. Peat soils are also used for small-scale agriculture such as vegetables and fruits to increase the livelihood of rural people (Ismail and Jamaludin, 2007). For these purpose, peatlands need to be drained as they are highly waterlogged, and are commonly limed due to their acidic nature. They are also fertilized due to the low availability of nutrients and deficiency in micronutrients (MARDI, 1996; Safford and Maltby, 1998; Watanabe *et al.,* 2009).

Presently, 5.1 to 12.9 million hectares of peat in Southeast Asia (Hoojier *et al.,* 2010; Jauhiainen *et al.,* 2012) have been drained for large-scale agriculture. The area of peat soils under agriculture is expected to increase in Malaysia and Indonesia where agriculture is one of their engines of economic growth. In Malaysia, approximately 600, 000 hectares of peats are cultivated with oil palm, pineapple, rubber, and sago (Ismail, 2008). In 2015, the Malaysian government announced the expansion plan to increase pineapple acreage in the state of Sarawak to 3500 hectares for MD2 pineapple cultivars in Kota Samarahan, Saratok, Mukah, Tanjung Manis, and Bau-Lundu region (Reukeith, 2015). This rapid agricultural development on peat soils however, is currently gaining widespread criticism from both the scientific community and policy makers. This is because tropical peats in Southeast Asia are considered as an important buffer for climate change as they are good in storing carbon at approximately 68.5 G tonnes of carbon (Page *et al.,* 2011).

Deforestation and drainage of peats change their physical, chemical, and biological properties. Peat decomposition is the breakdown of parent materials by microorganisms in aerobic condition that eventually leads to carbon loss and contributes to greenhouse gas emission namely CO_2, CH_4, and N_2O into the atmosphere (Hadi *et al.,* 2000; Maljanen *et al.,* 2004; Kløve *et al.,* 2010). In addition, clearing and drainage of peat soils for agricultural development normally involves slash and burn (Ismail and Jamaludin, 2007; Ismail, 2010). This method is cheap, fast, and reliable that helps to naturally fertilize peat soils from the ashes of burned plant materials. Problems arise not only from the degradation of the peat materials over time but partly due to the risks of peat fires. Burning leads to CO_2 and other toxic gases into the atmosphere (Zulkefli *et al.,* 2010). Greenhouse gas emissions from agricultural activities on drained tropical peats had since gained the interest of scientific community back in the late 1990s' due to the occurrence of the major peat fires that caused heavy haze in Southeast Asia in 1997 to 1998. The peat fires were due to large scale burning of peat areas in Indonesia especially in Kalimantan and Sumatra for large-scale plantations. This disrupted not only economic activities in the region but posed environmental pollution and health risks, in which it was estimated to contribute approximately 0.6 to 3.8 G tonnes of carbon to the atmosphere (Ismail, 2010).

Greenhouse gas emissions from tropical peatlands

In their natural states, peats emit CH_4 but sequester CO_2. Peatlands are waterlogged most time of the year and requires drainage to make the soil suitable for agriculture. Although peats store large amount of organic carbon, peat soils drained for agriculture in particular accelerate their decomposition rates (Maljanen *et al.,* 2004; Kløve *et al.,* 2010). Biogeochemical processes lead to release of CO_2, CH_4, and N_2O into the atmosphere (Berglund and Berglund, 2011). In addition, the biogeochemical processes cause surface subsidence and flooding especially in low-lying areas (Safford and Maltby, 1998; Kechavarzi *et al.,* 2010). In recent times, concerns about how this anthropogenic activity from tropical peats affects the balance of greenhouse gases in the atmosphere and their role in environmental degradation *vis a vis* climate change have been expressed (Hoojier *et al.,* 2010; Couwenberg, 2011).

Factors such as temperature, water table, and hydrology which affect GHG emissions in both temperate and boreal peatlands have been well studied (Waddington and McNeil, 2002; Biasi *et al.,* 2008; Kechavarzi *et al.,* 2010; Kløve *et al.,* 2010). However, studies on greenhouse gas emissions from drained tropical peatlands are limited. Greenhouse gas measurements are mostly carried out in the forest ecosystem, oil palm plantation, paddy field, rice-soybean, sago, and pulpwood (Inubushi *et al.,* 2003; Hadi *et al.,* 2005; Melling *et al.,* 2005; Watanabe *et. al.*, 2009; Jauhiainen *et al.,* 2012). Additionally, factors influencing greenhouse gas emissions on drained peatlands are poorly understood. At the moment, greenhouse gas emission studies are focused on the palm oil industry, as it is the major crop that is widely planted on drained peats in Indonesia and Malaysia. However, current studies have failed to address greenhouse gas emissions from drained peatlands cultivated with pineapples as pineapples are widely grown in peat soils of Malaysia.

Carbon Dioxide and Factors Affecting Its Emission

Carbon dioxide is emitted from peatlands when peat materials are decomposed because of accelerated oxidation of peat materials (Hoojier *et al.,* 2010; Jauhiainen *et al.,* 2012). Carbon dioxide may also be emitted from peat soils through burning by wildfires, root respiration, and physical oxidation (Kasimir-Klemedtsson, 1997; Kuzyakov, 2006). Carbon dioxide emission is influenced by depth of water table (Chimner and Cooper, 2003; Berglund and Berglund, 2011) and temperature (Berglund *et al.,* 2010; Jauhiainen *et al.,* 2012).

It is generally believed that higher CO_2 emission occurs by lowering the water table of peats because the process leads to exposure of the peat to oxidation through microbial processes. However, there are conflicting findings on the effect of water table fluctuation on CO_2 emission. Chimner and Cooper (2003) observed that lowering of water level within microcosm (fen peat) doubled the rate of CO_2 emission only at certain depth (e.g. when the water table dropped from just above to just below the soil surface) but had little effect on the emission rate upon further lowering of water table. Zulkefli *et al.* (2008) reported that soil CO_2 flux under different agricultural systems in tropical peatland varied with crop types but no clear evidence was found for the effect of water table depth. Among the explanations include stable soil thermal conductivity and labile carbon pool (Parmentier *et al.,* 2009; Berglund and Berglund, 2011). However, findings by Jauhiainen *et al.* (2012) show that soil moisture content in the top soil above the water table affect CO_2 emission by influencing oxygen availability in the peat pore space (aerobic condition) and not water table depth. Hence, there is a need to take into account soil moisture when evaluating greenhouse gas emission. Furthermore, Kechavarzi *et al.* (2010) stated that soil moisture content may not directly affect CO_2 respiration in well decomposed peat but observed higher emission rate for fibrous peat at higher moisture content in which CO_2 emission is influenced by peat type. Other studies that monitored the relationship between water table and CO_2 emission from drained peat by estimating CO_2 flux based on soil subsidence, carbon content, and bulk density showed that CO_2 emission increased approximately 9.1 t CO_2 ha^{-1} yr^{-1} when water table was lowered every 10 cm depth by assuming the carbon content as 60 kg m^{-3} (Hoojier *et al.,* 2010).

The other factor that affects CO_2 emission from peat decomposition is temperature. Temperature affects CO_2 emission by increasing peat oxidation through microbial activities. Higher soil temperature (25°C to 30°C) favours microbial activity but higher temperature (> 30°C) above the optimal activity may inhibit microbial respiration because of inactivation of biological oxidation system (Petterson, 2004; Pietikäinen *et al.,* 2005; Zulkefli *et al.,* 2010). Most studies on greenhouse gas measurements have revealed good correlation between temperature and CO_2 emission (Berglund *et al.,* 2010; Kechavarzi *et al.,* 2010; Zulkefli *et al.,* 2010; Jauhiainen *et al.,* 2012).

Carbon dioxide emission from agricultural activities on drained peatlands is currently gaining momentum and attention from the scientific community because of the direct CO_2 emission from decomposition of organic carbon in soils. This is because tropical peat store large amount of carbon (11 to 14% of the global peat carbon) (Jauhiainen *et al.,* 2012). Peats in the tropics are rapidly being depleted mostly for agricultural development. Hoojier *et al.* (2010) estimated that CO_2 emission from decomposition of drained peats in Southeast Asia contributed to 1.3 to 3.1% of the present global CO_2 emission which is said to be equivalent to the combustion of industrial fossil fuel. Current review by Couwenberg (2011) showed that estimates for CO_2 emission from cultivated drained peats is based on subsidence data (assuming that 40% of surface height loss is due to decomposition). However, this assumption might not be accurate as carbon loss measurement based on subsidence causes high variation taking into consideration the heterogeneity of the organic material and peat type under different climates. Furthermore, subsidence based measurements for CO_2 emission must only be made after an equilibrium state had been achieved. This is to ensure that the emission solely comes from microbial derived decomposition and not from the physical oxidation through shrinkage and consolidation of the peat after drainage (Kasimir-Klemedtsson *et al.,* 1997). Hence, there is a need for direct CO_2 measurements from drained and cultivated peats to ensure that emissions are neither underestimated nor overestimated.

Carbon Dioxide Emission in Pineapple Cultivation on Tropical Peat Soil

Peat soils cultivated with Moris pineapple in Malaysia released 179.6 t CO_2 ha/yr on a yearly basis and emissions were affected by seasonal variation (Lim Kim Choo and Ahmed, 2014). Soil CO_2 emission in the wet season was reported to be influenced by the heterogeneity of the soil organic matter and the diversity of microbial structure in peat (Kechavarzi *et al.,* 2010). Although soil temperature regulates soil CO_2 emission, emissions in the dry season were reported to be low. This findings suggested that higher soil temperature may have partially inhibited microbial respiration through the inactivation of biological oxidation system (Zulkefli *et al.,* 2010) and emissions across time rather depends on the moderate fluctuation in soil temperature (0.2 and 1.6°C) of the tropics (Lim Kim Choo and Ahmed, 2014).

Peat soils cultivated with pineapple was regulated by photosynthetic activity, heterotrophic respiration, and decomposition of root exudates at the rhizosphere (Kuzyakov, 2006; Mäkiranta *et al.,* 2008). Carbon dioxide emissions were affected by time of sampling. Soil CO_2 emissions increased during night time but were lower during day time. These observations were found to be consistent with the significant negative correlation between soil CO_2 emission and soil temperature reported by Lim Kim Choo and Ahmed (2014). These findings further suggested that CO_2 emission increased with decreasing temperature. Luta *et al.* (2017) reported that peat soil water table fluctuation does not significantly affect the emission of CO_2 in pineapple cultivation under field lysimeter study but emissions under low water table during the dry season was higher compared than that of high water table during the wet season. The findings by Luta *et al.* (2017) corroborates that of Jauhiainen *et al.* (2012) who confirmed that water table depth does not affect CO_2 emission but soil moisture content in the top soil above the water table affect CO_2 emission by influencing oxygen availability in the peat pore space.

Methane and Factors Affecting Its Emission

In their natural state, tropical peats emit methane (CH_4). Both disturb and undisturbed peatlands are sources of atmospheric CH_4. Methane is produced during microbial decomposition of organic matter under anaerobic condition (IAEA, 1992). Tropical peat soils are waterlogged under natural condition. This condition restricts diffusion of atmospheric oxygen into peats. This inhibits microbial decomposition of organic materials (Chimner & Cooper, 2003). Hence, microorganism are not able to metabolize organic matter into CO_2 instead, anaerobic degradation of carbon is carried out by methanogens to produce CH_4 (Parmentier *et al.,* 2009).

Methane production in the anaerobic layer occurs in the form of dissolved species or as bubbles. Methane is released into the atmosphere through: molecular diffusion (peat pore spaces), ebullition (a process where bubbles in gas are formed from a dissolved state in water and channel their way to surface to the atmosphere) at the water table interface, and vascular plants with aerenchymous porous tissues which transport gases through plant roots into the atmosphere (Dinsmore *et al.,* 2009; Watanabe *et al.,* 2009; Farmer *et al.,* 2011; Zhang and Jiang, 2014). Methane can also be emitted through non-microbial CH_4 production from soil organic matter under favorable conditions such as high temperature, UV radiation, and reactive oxygen species (Wang *et al.,* 2013).

Methane emission from tropical peatlands seems to be influenced by water table (Moore and Dalva, 1993) and soil temperature (Nyakanen *et al.,* 1995; Hargreaves and Fowler, 1998). Drained tropical peatlands emit low methane through microbial processes in which CH_4 produced at the saturated layer (at the interface or beneath the water table) is oxidized at the unsaturated soil into CO_2 by methanotrophs (Parmentier *et al.,* 2009). However, methane could be influenced by fertilization (Watanabe *et al.,* 2009). Methane emitted through vascular plant mediated transport could as well be substantial on drained peats that are cultivated with crops (Watanabe *et al.,* 2009). In addition, methane emission from drained peatlands is low and negligible but emissions from available ditches and during flooding could be substantial (Couwenberg, 2011). Couwenberg (2011) reported that study on CH_4 emission from paddy ecosystem on peats was within uncertainty range of the Intergovernmental Panel on Climate Change (IPCC) CH_4 default emission factor.

Methane Emission in Pineapple Cultivation on Tropical Peat Soil

Lim Kim Choo and Ahmed (2017a) have reported that peat soils cultivated with pineapple contributed to low CH_4 emission (0.65 t CH_4 ha/yr) on a yearly basis and emissions were not affected by seasonal variation and time of sampling. Soil CH_4 emissions for pineapple cultivated peats were further reported to be affected by nitrate-based fertilization through nitrate electron acceptors by inhibiting CH_4 production (Jassal *et al.,* 2011; Sirin and Laine, 2012) but emissions were neither affected by soil temperature nor by soil moisture. The findings by Lim Kim Choo and Ahmed (2017a) on the insignificant correlation between soil temperature and CH_4 emissions corroborates that of Jeffary *et al.* (2016) who suggested that the factor controlling CH_4 emission was mainly related to the fluctuation of water table at the soil-water interface. It was also possible that CH_4 emissions were emitted from non-microbial production of CH_4 sources such as lignin and humic acids (Wang *et al.,* 2013). This might have occurred under moderate temperature fluctuation of the tropics as peats are high in organic matter besides being a natural polyelectrolyte with substances such as humic acids, fulvic acids, humin, lignin, and carbohydrate (Helal Udin *et* *al.,* 2003; Allen *et al.,* 2004; Zulfikar *et al.,* 2013). Jeffary *et al.* (2016) reported that horizontal emissions of CH_4 in pineapple cultivated peats were higher than that of vertical emissions with increased emissions particularly in the wet seasons due to the increase in water table. Nevertheless, the lower CH_4 emissions from the reported findings are consistent with the general believe that CH_4 emission from drained cultivated tropical peat is lower than those of anaerobic or water logged peat soils.

Nitrous Oxide and Factors Affecting Its Emission

Nitrous oxide is produced from two main processes, firstly through denitrification. Denitrification is defined as the reduction of nitrate to N_2O and consequently to dinitrogen. The second process nitrification, whereby ammonium is oxidized to nitrate. These processes are regulated by microbial activities which are influenced by soil nitrogen and nitrogen fertilization (IAEA, 1992; Saggar *et al.,* 2013; Uchida *et al.,* 2013). Nitrous oxide is considered as one of the greenhouse gases that must be accounted for under the United Nation Framework Convention on Climate Change (UNFCCC) due to its ozone depleting properties and effect on global warming (Hadi *et al.,* 2000; Jassal *et al.,* 2011; Chen *et al.,* 2014). The lifespan of N_2O is approximately 120 years compared to other greenhouse gases and the global warming potential of N_2O is 310 times greater than CO_2 (Reth *et al.,* 2008).

Nitrous oxide from peatlands is derived from both nitrification and denitrification processes (Maljanen *et al.,* 2007; Jauhiainen *et al,* 2012). Nitrification occurs in aerobic condition in drained and fertilized peat soils (van Beek *et al.,* 2010). This is because decomposition of organic nitrogen which in turn accelerates soil mineralization, thus increasing inorganic form of nitrogen which is commonly accompanied by the release of N_2O into the atmosphere. Anaerobic condition in peats favours N_2O emission through nitrifying bacteria which use nitrate for their metabolic processes. This ends up releasing N_2O as a by product to the atmosphere.

Both aerobic and anaerobic conditions cause N_2O emission, but the release is influenced by nitrogen fertilization in particular because N_2O emission increases through mineralization (Couwenberg, 2011; Saggar *et al.,* 2013; Uchida *et al.,* 2013). However, N_2O emission is reported to be regulated by soil moisture as the emission of this gas is high at intermediate soil moisture content (Kasimir-Klemedtsson *et al.,* 1997). Fluctuation in water table also affects N_2O emission (Hadi *et al.,* 2000; Kløve *et al.,* 2010; Berglund and Berglund, 2011). At the moment, there are limited studies on N_2O emission from pineapple cultivation on drained peat soils. Nitrous oxide emission studies from drained tropical peats are limited to paddy, cassava, and rice-soybean fields (Hadi *et al.,* 2000; Inubushi *et al.,* 2003; Hadi *et al.,* 2005). Existing reported data on N_2O emission from drained tropical peatlands is based on similar studies in the temperate grasslands and temperate croplands (Couwenberg, 2011). Hence, there is a need for direct N_2O measurement from tropical drained cultivated peats to provide basis for future emission factors under different land use.

Nitrous Oxide Emission in Pineapple Cultivation on Tropical Peat Soil

Cultivation of Moris pineapple on drained peat soils was reported to release 15.7 t N_2O ha/yr on a yearly basis and N_2O emissions were influenced by seasonal variation (Lim Kim Choo and Ahmed, 2017b). Seasonal variation in soil N_2O emissions could be ascribed to fertilization management in pineapple cultivation in which nitrogen based fertilization may have contributed to N_2O emission through mineralization (Kasimir-Klemedtsson *et al.,* 1997; Couwenberg, 2011; Jassal *et al.,* 2011). It is also possible that soil N_2O emissions were influenced by root exudates at the rhizosphere. These root exudates are low in nitrate due to plant nitrogen uptake (Saggar *et al.,* 2013) that contributes to a different rate of N_2O emission. Although fertilization regulates N_2O emission, soil N_2O emission from pineapple cultivated peat was reported not to be affected by soil temperature nor by soil moisture (Lim Kim Choo and Ahmed, 2017b). Due to the limited reports available on N_2O emissions from pineapple cultivated peat soils, further research is needed to assess N_2O emissions from cultivated peats as N_2O emissions from drained peats seems to be influenced by nitrogen fertilization, diversity of microbial structure, and the heterogeneity of the soil organic matter.

Soil Greenhouse Gas Flux Measurement Approach on Cultivated Peat Soils

Generally, there are three established methods to quantify soil greenhouse gas emissions on peat soils. The methods are soil subsidence based estimation, eddy covariance and chamber method. A simplified schematic diagram showing the principle approach and disadvantage for each method for greenhouse gas flux measurement is illustrated in Figure 1.

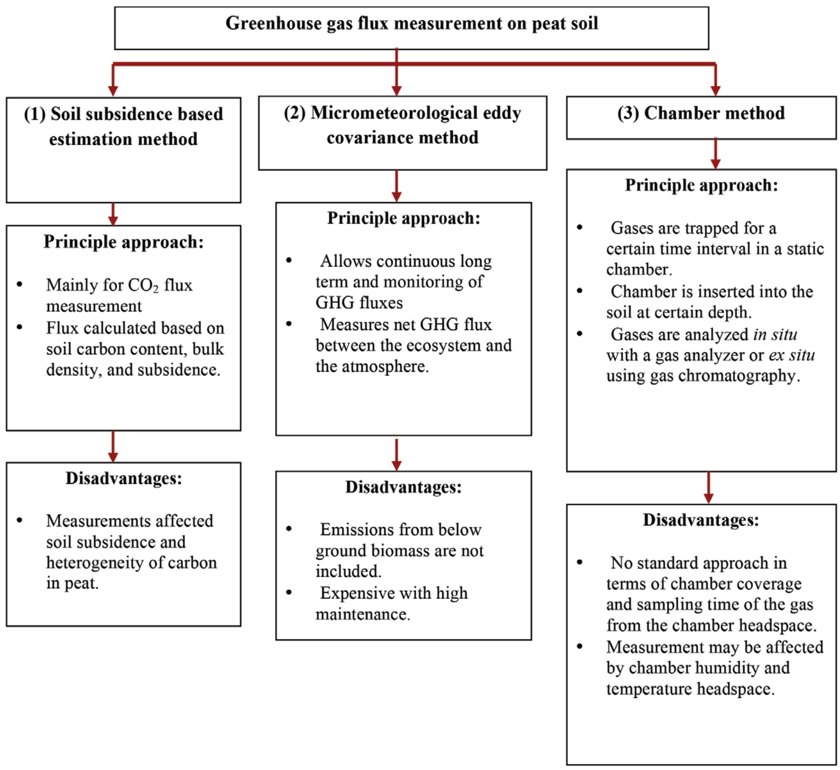

Figure 1 Methods to quantify greenhouse gas emissions on peat soil.

Quantification of greenhouse gas flux based on soil subsidence estimation is applicable to CO_2 emission only. The principle of this approach is by calculating CO_2 emission based on the carbon content of the soil, bulk density, and subsidence. This is a direct approach, simple, and clearly defined when all required data is complete. However, problems arise by taking into consideration the carbon content of the soil as flux measurement from tropical peat utilizing this method assumes carbon content of the tropical peat to be 60 kg m^{-3} or 60% of organic matter (Hoojier *et al.*, 2012). Carbon varies because of the heterogeneity of peat materials. It is also influenced by degree of decomposition, type of peat, and location in response to changing environmental processes under different climate (Kløve *et al.,* 2010; Berglund and Berglund, 2011; Sirin and Laine, 2012). In addition, peat soil subsidence is affected by microbial decomposition of organic matter and physical oxidation of shrinkage and consolidation due to the loss of water volume after peat drainage. To accurately estimate CO_2 emission, flux quantification must only be taken into account after an equilibrium state of peat subsidence has been achieved. This is because subsidence due to physical oxidation is continuous and may take several years (three to five years) after the peat is drained (Kasimir-Klemedtsson *et al.,* 1997; Berglund and Berglund, 2011; Couwenberg, 2011).

The second method that is also used to measure greenhouse gas flux from tropical peats is the micrometeorological eddy covariance method. This approach allows the continuous and long term monitoring of greenhouse gas fluxes. It measures the net greenhouse gas flux between the ecosystem and the atmosphere including tall vegetation (Couwenberg, 2011). However, the limitation of this method is that it is unable to directly quantify the emission coming from below ground biomass including litter stocks. Furthermore, this approach can only quantify total soil respiration but cannot separate its individual respiration components and normally, closed chamber method is applied together to measure the emission coming from the soil (Kuzyakov, 2006). In addition, by combining the micrometeorological eddy covariance and chamber method, the net carbon and nitrogen cycle and the associated sequestration can be obtained. However, eddy covariance method is relatively expensive with high maintenance costs (Kuzyakov, 2006).

The widely used method to measure greenhouse gas fluxes is the direct measurement approach using the closed chamber method (Kløve *et al.,* 2010; Berglund and Berglund, 2011; Couwenberg, 2011). The principle of this method is based on trapping of gasses for a certain time interval in a static chamber that is inserted into the soil at certain depths in which the trapped gases is either analyzed *in situ* with a gas analyzer or collected and analyzed *ex situ* using gas chromatography (IAEA, 1992). This method quantifies total soil respiration without separating the individual components of soil respiration into autotrophic and microbial respiration (Kuzyakov, 2006).

Although the chamber method is well established and widely used for greenhouse gas flux measurement in tropical peat, questions arise in terms of coverage of the chamber, sampling time of the gas from the chamber headspace and limited number of sampling cycles carried out under different land use. This is because the limited studies available on greenhouse gas measurements using closed chamber method on drained tropical peat has failed to provide standard approach. This has resulted in controversial findings where most measurements are made in small plots (few cm^2) and duration (few minutes) (Inubushi *et al.,* 2003; Hadi *et al.,* 2005; Zulkefli *et al.,* 2008). The coverage of the chamber must take into account the inherent heterogeneity of the peats. Furthermore, limited number of time interval during sampling (for a few minutes) will not be able to represent the emission of gases from the soil. This is because emission may be influenced by gas diffusion through pores into the air. However, if the sampling time interval is extended excessively (for example 30 minutes), gas fluxes may not be accurately measured as it could be affected by the chamber humidity and temperature headspace (Couwenberg, 2011). In addition, because greenhouse gas studies on drained tropical peats are carried out during day time (Inubushi *et al.,* 2003; Hadi *et al.,* 2005; Zulkefli *et al.,* 2010), the chamber method fails to account for emissions during the night. This is because greenhouse gas fluxes are regulated by temperature (Berglund *et al.,* 2010; Kechavarzi *et al.,* 2010; Jauhiainen *et al.,* 2012). Furthermore, differences between day and night temperatures could show different trends of greenhouse gas emissions under different land uses.

At present, there are no specific guidelines on the size of the chamber and interval sampling time at the chamber headspace to allow for accurate measurements of greenhouse gas fluxes from drained cultivated peat. Therefore, measurement of greenhouse gas fluxes must take into consideration the appropriate size of the chamber which relates to the type of peat and land use management at the experimental site. Furthermore, intermediate sampling time interval must be considered to avoid high humidity in the chamber and temperature headspace. This allows for accurate measurements for greenhouse gas emission from the soil. Moreover, increasing sampling periods to include not only day but also night measurements could clarify how greenhouse gas fluxes respond to certain environmental factors such as temperature and moisture. Generally, the chamber method seems to provide a direct measurement of soil greenhouse gas emission which is reliable, relatively inexpensive and applicable for most ecosystem and land use.

Partitioning Soil Respiration From Cultivated Peat Soils

Most greenhouse gas studies on cultivated drained tropical peats only quantified total soil respiration (mainly for CO_2 emission) without partitioning greenhouse gas emission into individual components that contribute to the total soil respiration (Inubushi *et al.,* 2003; Hadi *et al.,* 2005; Melling *et al.,* 2005; Vasender and Jauhiainen, 2008; Zulkefli *et al.,* 2008). Sources of soil respiration can be divided into autotrophic respiration by plant and heterotrophic respiration by microorganisms. Partitioning soil respiration is gaining research interest in the tropics. This is because total soil greenhouse gas emission does not account for which respiration component (plant or microbial respiration) responds to changing environmental condition such as temperature, soil moisture, water table, land use management, and intensity of different weather patterns that clarifies whether the soil is a sink or source to atmospheric greenhouse gases (Mäkiranta *et al.,* 2008; Kuzyakov, 2006; Kløve *et al.,* 2010).

Components of soil respiration from plant derived respiration can be divided into root respiration, rhizo-microbial respiration, and microbial respiration of dead plant residues, or, in other words, respiration that occurs in the rhizosphere (Baggs, 2006; Kuzyakov, 2006; Mäkiranta *et al.,* 2008). On the other hand, soil microbial respiration is divided into microbial decomposition of soil organic matter in root free soil and microbial decomposition of soil organic matter with plant residues. It also includes microbial respiration of dead plant residues and rhizo-microbial respiration where both respirations fall under plant derived respiration at the rhizosphere (Kuzyakov, 2006). A simplified schematic diagram showing the individual components which contribute to the total soil respiration comprising plant and microbial derived respiration is illustrated in Figure 2.

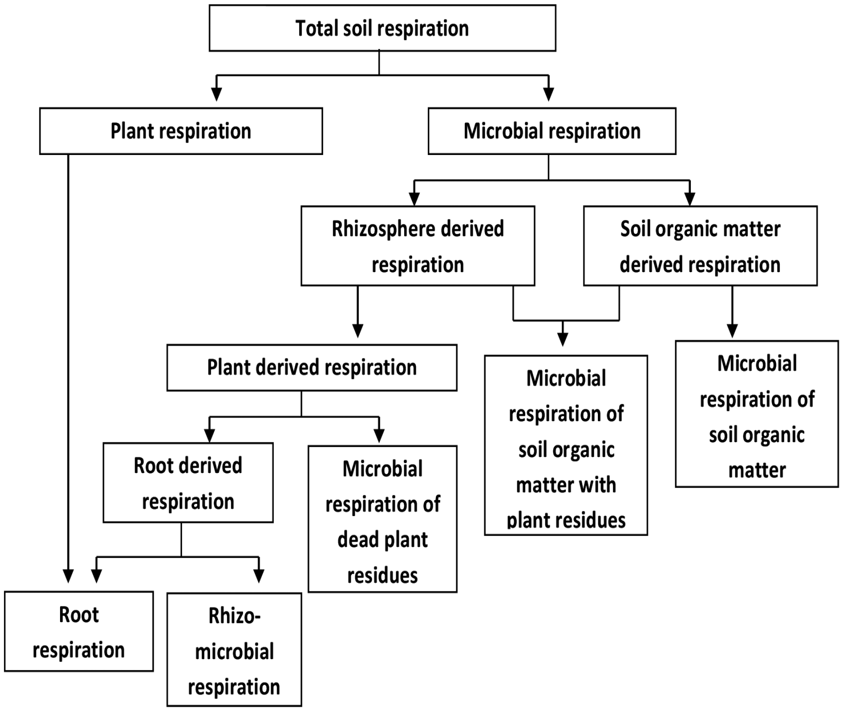

Figure 2 Partitioning of soil respiration into its individual respiration sources (Adapted from Kuzyakov, 2006)

Soil respiration is commonly divided into autotrophic and microbial respiration using non-isotopic method. The most commonly used methods are trenching and component integration methods. The trenching method measures differences in soil respiration from planted soil with crops (by severing roots to prevent further respiration) and without trenched roots while the component integration method measures soil respiration by manually separating respiration sources. The two approaches are simple, experimentally easy to carry out and implement, inexpensive, and applicable to various ecosystems (Baggs, 2006; Kuzyakov, 2006). Apart from these methods, another approach is the isotopic method that commonly uses isotopic ^{14}C and ^{13}C labeling. This method allows *in situ* separation between root and soil respiration by eliminating soil-plant disturbance. This isotopic technique is expensive, experimentally complex and it requires special equipment as well as maintenance (Baggs, 2006; Kuzyakov, 2006).

At the moment, partitioning of soil CO_2 emission from drained tropical peats cultivated with pulpwood in Sumatra, Indonesia has been reported by Jauhiainen *et al.* (2012). In this study, both trenching and component integration methods were used to separate autotrophic and microbial respiration. Jauhiainen *et al.* (2012) reported that root respiration contributed to 21% of the total soil CO_2 emission while the remaining 79% was from heterotrophic microbial respiration. Partitioning of soil CO_2 emissions into root respiration, microbial respiration, and oxidative peat decomposition of a drained peatland cultivated with pineapple in Malaysia has been reported by Lim Kim Choo and Ahmed (2014). In this study, partitioning of soil CO_2 emission was achieved using component integration method based on the differences between planted soil and unplanted soil utilizing the lysimeter approach. The authors reported that microbial respiration were largely responsible for CO_2 emission that contributed to 36% (218.8 t CO_2 ha/yr) of the total soil CO_2 emission on a yearly basis, followed by oxidative peat decomposition (34% at 205 t CO_2 ha/yr) while root respiration contributed to a lower CO_2 emission (30% at 179.6 t CO_2 ha/yr). Soil CO_2 emissions reported by Lim Kim Choo and Ahmed (2014) were higher than those reported by Jauhiainen *et al*. (2012). The difference in the CO_2 emission rate can be ascribed to several reasons including different type of pulpwood plant physiology, plant density, and canopy coverage which affect solar radiation. Furthermore, the heterogeneity of the peat type which is mainly fibric to hemic in the study by Jauhiainen *et al.,* (2012) compared with sapric peat in the study by Lim Kim Choo and Ahmed (2014) may have accelerated peat decomposition process characterized by higher CO_2 emission (Berglund *et al.,* 2010). In addition, the study of Jauhiainen *et al.* (2012) was limited to day time CO_2 emission measurement compared with the 24 hour monitoring period in the study by Lim Kim Choo and Ahmed (2014) where differences in day and night soil temperatures may have affected CO_2 emission.

Conclusion

Peat soils cultivated with pineapple contributed to the release of CO_2 and N_2O emissions into the atmosphere but were a negligible source of CH_4 under drained condition. Soil greenhouse gas emissions were influenced by pineapple photosynthetic activity, heterotrophic respiration, decomposition of root exudates at the rhizosphere, heterogeneity of soil organic matter, and nitrogen based fertilization for pineapple. Soil greenhouse gas emissions were neither affected by soil temperature nor soil moisture but the emissions seemed to be controlled by moderate soil temperature fluctuation throughout the wet and dry seasons. Flux measurement during soil greenhouse gas monitoring using the closed chamber method may have been influenced by gas diffusion through peat pores. Thus, greenhouse gas emission studies should take into consideration the lateral and vertical gas diffusion movement as the transport mechanism of gas fluxes may be affected by peat macro and micropores at different soil depths. Further research is also needed on partitioning soil respiration at the rhizosphere. This is important as most of the greenhouse gas emissions from drained peat were contributed by heterotrophic and autotrophic respiration at the rhizosphere. This study will pose a challenge as rhizosphere derived greenhouse gas emission include respiration from rhizomicrobial respiration, microbial respiration of dead plant residues, microbial decomposition of soil organic matter in root affected soil, and root respiration. Different technique of flux measurement may have to be considered to partition soil rhizosphere derived respiration including isotopic method. Mitigation measures from drained cultivated peat are needed to control greenhouse gas emissions from pineapple cultivation. Identification of beneficial microorganisms that reduce peat decomposition may help to minimize CO_2 emission from cultivated peats. Quantification and identification of the ratio of active methanogenic and methanotrophic communities in peat and the possibility of pineapples as plant-mediated CH_4 transport may help to further verify whether drained peats are net sources or sinks of CH_4. Further research is needed to assess non-microbial CH_4 production in peat as this soil contains high organic matter with lignin and humic acids as its major components. Application of potential natural materials to replace nitrogen based fertilization for pineapple is needed to minimize N_2O emissions.

References

Allen, S.J., McKay, G. and Porter, J.F. (2004). Adsorption isotherm models for basic dye adsorption by peat in single and binary component systems. *Colloid and Interface Science* 280: 322-333.

Andriesse, J.P. (1988). The main characteristics of tropical peats. In: *Nature and Management of Tropical Peat Soils*. FAO Soils Bulletin 59. Rome: FAO.

Baggs, E.M. (2006). Partitioning the components of soil respiration: a research challenge. *Plant Soil* 284: 1-5.

Berglund, Ö, Berglund, K. and Klemedtsson, L. (2010). A lysimeter study on the effect of temperature on CO_2 emission from cultivated peat soils. *Geoderma* 154 (3-4): 211-218.

Berglund, Ö and Berglund, K. (2011). Influence of water table level and soil properties on emissions of greenhouse gases from cultivated peat soil. *Soil Biology & Biochemistry* 43: 923-931.

Biasi, C., Lind, S.E., Pekkarinen, N.M., Huttunen, J.T., Shurpali, N.J., Hyvönen, N.P., Repo, M.E. and Martikainen, P.J. (2008). Direct experimental evidence for the contribution of lime to CO_2 release from managed peat soil. *Soil Biology & Biochemistry* 40: 2660-2669.

Chen, H., Mothapo, N.V. and Shi, W. (2014). The significant contribution of fungi to soil N_2O production across diverse ecosystems. *Applied Soil Ecology* 73: 70-77.

Chimner, R.A. and Cooper, D.J. (2003). Influence of water table levels on CO_2 emissions in a Colorado subalpine fen: an insitu microcosm study. *Soil Biology & Biochemistry* 35: 345-351.

Couwenberg, J. (2011). Greenhouse gas emissions from managed peat soils: is the IPCC reporting guidance realistic?. *Mires and Peat* 8 (2): 1-10.

Dinsmore, K.J., Skiba, U.M., Billett, M.F. and Rees, R.M. (2009). Effect of water table on greenhouse gas emissions from peatland mesocosms. *Plant Soil* 318: 229-242.

Farmer, J., Matthews, R., Smith, J.U., Smith, P. and Singh, B.K. (2011). Assessing existing peatland models for their applicability for modelling greenhouse gas emissions from tropical peat soils. *Current Opinion in Environmental Sustainability* 3: 339-349.

Hadi, A., Inubushi, K., Purnomo, E., Razie, F., Yamakawa, K and Tsuruta, H. (2000). Effect of land-use change on nitrous oxide (N_2O) emission from tropical peatlands. *Chemosphere – Global Change Science* 2: 347-358.

Hadi, A., Kazuyuki, I., Yuichiro, F., Erry, P., Muhammad, R. and Haruo, T. (2005). Greenhouse gas emissions from tropical peatlands of Kalimantan, Indonesia. *Nutrient Cycling in Agrosystems* 71: 73-80.

Hargreaves, K.J. and Fowler, D. (1998). Quantifying the effects of water tables and soil temperature on the emission of methane from peat wetland at the field scale. *Atmospheric Environment* 32 (19): 3275-3282.

Helal Udin, A.B.M., Ahmad Sujari, A.M. and Mohd. Nawi, M.A. (2003). Effectiveness of peat coagulant for the removal of textile dyes from aqueous solution and textile wastewater. *Malaysian Journal of Chemistry* 5 (1): 034-043.

Hoojier, A., Page, S., Canadell, J.G., Silvius, M., Kwadijk, J., Wösten, H. and Jauhiainen, J. (2010). Current and future CO_2 emissions from drained peatlands in Southeast Asia. *Biogeoscience* 7: 1505-1514.

Hoojier, A., Page, S., Jauhiainen, J., Lee, W.A., Lu, X.X., Idris, A. and Anshari, G. (2012). Subsidence and carbon loss in drained tropical peatlands. *Biogeoscience 9*: 1053-1071.

Huat, B.B.K., Kazemian, S., Prasad, A. and Barghchi, M. (2011). State of an art review of peat: General perspective. *International Journal of the Physical Sciences* 6 (8): 1988-1996.

IAEA (1992). Manual on measurement of methane and nitrous oxide emissions from agriculture. In *Sampling techniques and sample handling*. IAEA-TECDOC-674. Vienna, Austria: IAEA.

Inubushi, K., Furukawa, Y., Hadi, A., Purnomo, E. and Tsuruta, H. (2003). Seasonal changes of CO_2, CH_4 and N_2O fluxes in relation to land-use change in tropical peatlands located in coastal area of South Kalimantan. *Chemosphere 52*: 603-608.

Ismail, A.B. and Jamaludin, J. (2007). Land Clearing Techniques Employed at MARDI Peat Research Station, Sessang, Sarawak, and their Immediate Impacts. In *A case study at MARDI Peat Research Station, Sessang, Sarawak, Malaysia*, eds. A.B. Ismail, H.K. Ong, M.J. Mohamad Hanif and M.S. Umi Kalsom, pp. 1-8. Malaysia: MARDI.

Ismail, A.B. (2008). Towards wise use of tropical peatland: from agriculture perspective. *Proceedings of the International Symposium and Workshop on Tropical Peatland,* Kuching, Sarawak, Malaysia, 19-22 August, pp. 129-141.

Ismail, A.B. (2010). Farm management practices for mitigation of carbon dioxide emission in peatland agrosystems. In *Proceedings of the International Conference on Balanced Nutrient Management for Tropical Agriculture*. Kuantan, Pahang, 12-16 April, pp. 72-76.

Jauhiainen, J., Hoojier, A. and Page, S.E. (2012). Carbon dioxide emissions from an Acacia plantation on peatland in Sumatra, Indonesia. *Biogeoscience* 9: 617-630.

Jassal, R.S., Black, T.A., Roy, R. and Ethier, G. (2011). Effect of nitrogen fertilization on soil CH_4 and N_2O fluxes, and soil and bole respiration. *Geoderma* 162: 182-186.

Jeffary, A.V., Ahmed, O.H., Heng, R.K.J., and Choo, L.N.L.K. (2016). Horizontal and vertical emissions of methane from a drained tropical peat soil cultivated with pineapple (Ananas comosus (L.) Merr.). *International Journal of Biosciences* 9(5): 10-18.

Kasimir-Klemedtsson, A., Klemedtsson, L., Berglund, K, Martikainen, P., Silvola, J. and Oenema, O. (1997). Greenhouse gas emissions from farmed organic soils: a review. *Soil Use and Management* 13: 245-250.

Kechavarzi, C., Dawson, Q., Bartlett, M. and Leeds-Harrison, P.B. (2010). The role of soil moisture, temperature and nutrient amendment on CO_2 efflux from agricultural peat soil microcosms. *Geoderma* 154: 203-210.

Kløve, B., Sveistrup, T.E. and Hauge, A. (2010). Leaching of nutrients and emission of greenhouse gases from peatland cultivation at Bodin, Northern Norway. *Geoderma* 154: 219-232.

Kuzyakov, Y. (2006). Sources of CO_2 efflux from soil and review of partitioning methods. *Soil Biology & Biochemistry* 38: 425-448.

Lim Kim Choo, L.N. and Ahmed, O.H. (2014). Partitioning carbon dioxide emission and assessing dissolved organic carbon leaching of a drained peatland cultivated with pineapple at Saratok, Malaysia. *Scientific World Journal*, vol. 2014, Article ID 906021, 9 pages. http://dx.doi.org/10.1155/2014/906021

Lim Kim Choo, L.N. and Ahmed, O.H. (2017a). Methane emission from pineapple cultivation on a tropical peatland at Saratok, Malaysia. Sustainable Agriculture Research: In Press (Vol. 6 No. 3)

Lim Kim Choo, L.N. and Ahmed O.H. (2017b). Nitrous oxide emission of a tropical peat soil grown with pineapple at Saratok, Malaysia. Sustainable Agriculture Research: In Press (Vol. 6 No. 3)

Luta, W., Ahmed, O.H., Heng, R.K.J., and Choo, L.K.N. (2017). Water table fluctuation and carbon dioxide emission from a tropical peat soil cultivated with pineapples (Ananas comosus L. Merr.). *International Journal of Biosciences* 10(1): 172-178.

Ma, W. and Tobin, J.M. (2004). Determination and modeling effects of pH on peat biosorption of chromium, copper and cadmium. *Biochemical Engineering* 18: 33-40.

Mäkiranta, P., Minkkinen, K., Hytönen, J. and Laine, J. (2008). Factors causing temporal and spatial variation in heterotrophic and rhizospheric components of soil respiration in afforested organic soil croplands in Finland. *Soil Biology & Biochemistry* 40: 1592-1600.

Maljanen, M., Komulainen, V.M., Hytönen, J., Martikainen, P.J. and Laine, J. (2004). Carbon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics. *Soil Biology & Biochemistry* 36: 1801-1808.

Maljanen, M., Martikkala, M., Koponen, H.T.Virkajärvi, P. & Martikainen, P.J. (2007). Fluxes of nitrous oxide and nitric oxide from experimental excreta patches in boreal agricultural soil. *Soil Biology and Biochemistry, 39,* 914-920.

MARDI (1996). *Master Plan for Malaysian Agricultural Research and Development Institute Sessang Peat Research Station*, pp. 34 – 67. Malaysia: MARDI.

Melling, L. (2005). Soil of the study area, physical properties. In *Dalat and Mukah Sago Plantation Peat Soil Study Final Report*, pp. 23-32. Malaysia: Department of Agriculture Sarawak.

Melling, L., Hatano, R. and Goh, K.J. (2005). Soil CO_2 flux from three ecosystems in tropical peatland of Sarawak, Malaysia. *Tellus B* 57:1-11

Mohammed Selamat, M. (1996). Biologi tanaman dan keperluan persekitaran. In *Penanaman Nanas - Nanas makan segar dan nanas kaleng,* ed. M. Mohammed Selamat, pp. 7-16. Malaysia: MARDI.

Moore, T.R. and Dalva, M. (1993). The influence of temperature and water table position on carbon dioxide and methane from laboratory columns of peatland soils. *Journal of Soil Science* 44: 651-644

MPIB (2011). Perangkaan Asas Nanas – Lembaga Perindustrian Nanas Malaysia.

Mutalib, A.A., Lim, J.S., Wong, M.H. and Koonvai, L. (1991). Characterization, distribution and utilization of peat in Malaysia. In *Proceedings of the International Symposium on Tropical Peatland*. Kuching, Sarawak, Malaysia, 6-10 May, pp. 7-16.

Nik Masdek, N.H. (2007). Paratylenchus sp. Associated with pineapple yield decline. *Journal of Tropical Agriculture and Food Science* 35 (1): 191-199.

Nyakanen, H., Alm, J., Lang, K., Silvola, T. and Martikainen, P.J. (1995). Emission of CH_4, N_2O and CO_2 from a virgin fen drained for grassland in Finland. *Journal of Biogeography* 22: 351-357.

Page, S.E., Rieley, J.O. and Banks, C.J. (2011). Global and regional importance of the tropical peatland carbon pool. *Global Change Biology* 17: 798-818.

Parmentier, F.J.W., van der Molen, M.K., de Jeu, R.A.M., Hendriks, D.M.D. and Dolman, A.J. (2009). CO_2 fluxes and evaporation on a peatland in the Netherlands appear not affected by water table fluctuations. *Agricultural and Forest Meteorology* 149: 1201-1208.

Petterson, M. (2004). *Factors Affecting Rates of Change in Soil Bacterial Communities.* (Doctoral thesis). Lund University, Sweden, http://www.lub.lu.se/luft/diss/sci_649/sci_649.pdf (accessed February 2014).

Pietikäinen, J., Petterson, M. and Bååth, E. (2005). Comparison of temperature effects on soil respiration and bacterial and fungal growth rates. *FEMS Microbial Ecology* 52 (1): 49-58.

Raziah, M.L. (2009). Scenario and prospect of Malaysia's pineapple industry. *Economic and Technology Management Review* 4: 11 – 24.

Raziah, M.L. and Alam, A.R. (2010). Status and impact of pineapple technology on mineral soil. *Economic and Technology Management Review* 5: 11-19.

Reth, S, Graf, W., Gefke, O., Schilling, R., Seidlitz, H.K. and Munch, J.C. (2008). Whole-year round observation of N_2O profiles in soil: a lysimeter study. *Water Air Soil Pollution: Focus* 8: 129-137.

Reukeith, G.O. (2015). Sarawak to increase pineapple production. In: The Borneo Post (Newspaper article: 30 November 2015).

Rieley, J.O. (2008). Tropical peatland – The amazing dual ecosystem: Co-existence and mutual benefit. Paper presented at the 12th International Peat Congress, International Peat Society, Tullamore, Ireland, 8-13 June.

Ritchie, R.J. & Bunthawin, S. (2010). Photosynthesis in pineapple (Ananas comosus comosus [L.] Merr) measured using PAM (Pulse Amplitude Modulation Fluorometry). *Tropical Plant Biology, 3(4),* 193-203.

Ritzema, H., Veltman, D. and Wösten, H. (2004). A system to support decision making for peatland management in the humid tropics. In: Proceedings of the 12th International Peat Congress, June 2004, Tampere, Finland, pp. 720-725.

Safford, L. and Maltby, E. (1998). Chapter 2 – Management and Chapter 3 – Nature and distribution of tropical peatlands. In *Guidelines for Integrated Planning and Management of Tropical Lowland Peatlands with Special Reference to Southeast Asia*, eds. L. Safford and E. Maltby, pp. 7, 37-41. Gland, Switzerland and Cambridge: IUCN Publications.

Saggar, S., Jha, N., Deslippe, J., Bolan, N.S., Luo, J., Giltrap, D.L., Kim, D.G., Zaman, M. and Tillman, R.W. (2013). Denitrification and N_2O: N_2 production in temperate grasslands: Processes, measurements, modeling and mitigating negative impacts. *Science of the Total Environment* 465: 173-195.

Sirin, A. and Laine, J., Chapter 7: Peatlands and greenhouse gases, 2012, Wetlands International, http://www.wetlands.or.id/PDF/chapter_7-9.pdf (accessed July 2012).

Uchida, Y., Von Rein, I., Akiyama, H. and Yagi, K. (2013). Contribution of nitrification and denitrification to nitrous oxide emissions in Andosol and from Fluvisol after coated urea application. *Soil Science and Plant Nutrition* 59: 46-55.

van Beek, C.L., Pleijter, M. & Kuikman, P.J. (2010). Nitrous oxide emissions from fertilized and unfertilized grasslands on peat soil. *Nutrient Cycling in Agroecosystems, 89(3),* 453-461.

Vasander, H. and Jauhiainen, J. (2008). Land use change in tropical peatlands and current uncertainties in greenhouse gas emissions, In *Proceedings of the*

International Symposium and Workshop on Tropical Peatland: Peatland development – Wise Use and Impact Management. Kuching, Sarawak, Malaysia, August 19-22, pp. 143-148.

Waddington, J.M. and McNeil, P. (2002). Peat oxidation in an abandoned cutover peatland. *Canadian Journal of Soil Science* 82: 279-286.

Wang, B., Hou, L., Liu, W. & Wang, Z. (2013). Non-microbial methane emissions from soils. *Atmospheric Environment, 80*, 290-298.

Watanabe, A., Purwanto, B.H., Ando, H., Kakuda, K. and Jong, F.S. (2009). Methane and CO_2 fluxes from an Indonesian peatland used for sago palm (*Metroxylon sagu* Rottb.) cultivation: Effects of fertilizer and groundwater level management. *Agriculture, Ecosystems and Environment* 134: 14-18.

Zhang, X. & Jiang, H. (2014). Spatial variation in methane emissions from natural wetlands in China. *International Journal of Environmental Science and Technology, 11*, 77-86.

Zulfikar, M.A., Novita, E., Hertadi, R. & Djajanti, S.D. (2013). Removal of humic acid from peat water using untreated powdered eggshell as a low cost adsorbent. *International Journal of Environmental Science and Technology, 10*, 1357-1366.

Zulkefli, M., Liza Nuriati, L.K.C., Ismail, A.B. and Jamaludin, J. (2008). Soil carbon loss under different land clearing techniques and agriculture systems on tropical peatland. In *Proceedings of the International Symposium and Workshop on Tropical Peatland: Peat Development – Wise Use and Impact Management*. Kuching, Sarawak, Malaysia, August 19-22, pp. 376-381.

Zulkefli, M., Liza Nuriati, L.K.C. and Ismail, A.B. (2010). Soil CO_2 flux from tropical peatland under different land clearing techniques. *Journal of Tropical Agriculture and Food Science* 38 (1): 131-137.

7

Distribution of Microbial Biomass, Carbon and Nitrogen in a Forest Reserves of Perak, Malaysia

Daljit Singh Karam, Keeren Sundara Rajoo, Farrah Melissa Muharam and Dzarifah Dzulperi

Introduction

Soil microorganisms are an important aspect in the evaluation of soil quality. This includes determining the soil microorganisms' abundance, biodiversity, food web structure and community. This is due to the sensitive nature and rapid response of soil microorganisms to environmental changes (Karam *et al*., 2016). Furthermore, soil microorganisms are correlated with important soil ecosystem functions, such as moisture storage, decomposition of forest debris or litter, nutrient cycling, detoxification of toxicants and suppression of noxious and pathogenic organisms. Therefore, soil microorganisms give a clear picture of the cause and effect chain that involves land use management decisions and an approach to optimize plant and animal productivity. Biological parameters that are widely recommended to quantify changes due to land conversion are microbial population counts, enzymatic activity and biomass (Briones *et al*., 2010; Karam *et al*., 2013b). Many studies of soil properties neglect the importance of biological elements, due to the lack of adequate and appropriate equipment; preparations for microbial studies require precision and caution (Müller *et al*., 2009).

Microbial biomass is the mass of living microorganisms in a particular ecosystem at a given time. Microbial biomass responds rapidly to different land use management techniques, and is also very dependent on the climatic conditions (Raubuch and Joergensen, 2002). Karlen *et al*. (1997) stated that microbial biomass should be included as a biological indicator because it reflects land use history and management. Nsabimana *et al*.

(2004) stated that soil microbial community's biodiversity are essential for soil function sustainability. Kwabiah *et al*. (2003) found that microbial biomass enhanced the process of transforming organic compound into an inorganic form that would be made available for plant uptake. Groffman *et al*. (2001) and Salamanca *et al*. (2006) in a study on soil microbial biomass and activity in tropical forests found that in disturbed forests left for natural regeneration growth, microbial activity was low, but there was a faster growth rate for vegetation that supplied soil microbial biomass with a considerable and adequate amount of organic matter. This finding indicates that microbial biomass is an important constituent of soil organic matter, and the availability of soil microbes in organic matter shows the recovery process of rehabilitated and natural regenerated forests. Moreover, through deforestation and the establishment of forest plantation projects, land managers can quantify and relate the structure and function of soil microbial activities at particular forest sites to the current soil properties (Karam *et al*., 2012; Karam *et al*., 2014; Karam *et al*., 2015). Hence, the objective of this study is to determine whether there's a difference in soil microbial activities between natural forests and rehabilitated forests. By comparing these two forests types, we would be able to determine whether enrichment is able to replicate natural forest's microbial activity over time.

Forest Rehabilitation

Forest rehabilitations are established to nurture or restore degraded forest areas to its original state. Most of the forest lands degrade due to lack of forest treatments, which is essential to restore the trees' growth. Furthermore, forest plantation establishment is carried in order to curtail the overexploitation of natural forest resources (Trockenbodt *et al*., 2002; Malik *et al*., 2015; Mohd-Aizat *et al*., 2015). Through forest plantation, the demand for specific tree species can be countered according to the need whether the planting would be mono-cultured or mix planted. The planted forest seedlings used can be from indigenous or exotic species, depending on the survival rates and demand for the wood production. For example, *A. mangium* plantations are important plantation estates, especially in Sabah, Malaysia. An introduced species, *Acacia mangium* is a fast growing hardwood tree that has high demand for wood chips and fiber production.

Another approach of forest rehabilitation through the introduction of valuable tree species to degraded forest areas without the elimination of valuable individual trees that already exist is known as enrichment planting (Montagnini *et al.*, 1997; Karam *et al.*, 2011b; Karam *et al.*, 2014). Enrichment planting is more effective in lowering negative impacts on soil quality compared to monoculture techniques because it results in less susceptibility to pests and diseases, higher biodiversity, better water conservation and less soil erosion. Many dipterocarps or timber tree species are shade tolerant; thus, shade trees or canopies protect seedlings from direct sunlight. As for multi-storied forest management, the technique of forest rehabilitation in which high quality timber trees are employed to create two or more layers of canopies. The upper canopies are secondary forest or planted *Acacia mangium*, while the lower canopy consisting of planted or introduced dipterocarps trees species.

In Malaysia, Multi-Storied Forest Management Project started from 1991 to 1999 and was divided into two distinctive phases. Multi-storied planting technique is a method of replanting trees by using fast growing trees to act as a canopy sheltering shade tolerant tree species. The Forestry Department Peninsular Malaysia (2002) stated that multi-storied forest management has gained significant attention as an ideal forest management technique for conserving biodiversity, preserving the environment and producing timber. Chikus and Bukit Kinta Forest Reserves in Perak have been subjected to this planting technique. *Acacia mangium* and indigenous high quality timber species including Shorea and Hopea were planted in the early stages of the project.

Chikus and Tapah Hill Forest Reserves

Sampling was carried out at a natural forest (N 04.10076˚ E101.19411˚, ± 28 m a.s.l) and an 18-year-old stand of *Shorea leprosula* (N 04.09197˚ E 101.19499˚, ± 28 m a.s.l) plots from 21st to 23rd July 2010. The area has an average annual precipitation of 3 223 mm and a mean temperature of 27.7°C. *S. leprosula* was planted in 1992 through collaborative work of the Multi-Storied Forest Management System involving the Forestry Department of Peninsular Malaysia, Perak State Forestry Department and Japan International Cooperation Agency (JICA). The planting distance for each tree was 10 m x 3 m. The study was also carried out in planted forest (F1) (N 04.179394˚ E 101.31998˚, ± 46 m) and secondary forest (F2)

(N 04.17336˚ E 101.31974˚, ± 32 m) at Tapah Hill Forest Reserve, Perak from 21st to 23rd July 2010 in which both selected plots were adjacent to each other, respectively. All of the tree species planting was done on 2nd February 1968 and the trees were 42 years old. The adjacent secondary forest had been left idle to undergo natural regeneration without any forest treatment. Twenty six thousand five hundred and forty four saplings were planted with 304 saplings per hectare and the rates of survival recorded in 1970 found that only 9,158 trees managed to grow well and survive with 105 saplings per hectare, respectively. *Shorea leprosula*, *S. parvifolia*, *S. bracteolata*, and *S. macroptera* were the main species of Dipterocarpaceae planted in the compartment 13 enrichment planting plot. The distance between each tree was 10 m x 3 m.

Soil Microbial Biomass C and N Distribution

Differences in the MBC/MBN ratio between natural forest and planted forest plots indicate that qualitative changes take place in microbial biomass; these changes are believed to be enhanced by the cycling and turnover process of microflora in the soil (Behera and Sahani, 2003). The MBC/MBN ratios found at 0-15 cm and 15-30 cm depths in the natural forest were within the optimum ranges of 5-8 (Figure 1). However, the higher MBC/MBN ratio in the planted forest plot compared to natural forest could be due to high amount of microbial biomass N compared to microbial biomass N (Joergensen *et al.*, 1995; Karam *et al.*, 2011a). Furthermore, immobilization of nutrients as a result of tremendous high amounts of microbial biomass C compared to microbial biomass N contributes to the higher MBC/MBN ratio observed in the soil of the planted forest plot; the higher MBC/MBN ratio in the lower soil layer of both the natural forest and planted forest plots indicates a low level of accumulation of organic matter and fine roots (Bremer and van Kessel, 1992; Maithani *et al.*, 1996; Karam *et al.*, 2011b). Longer period of time after rehabilitation of degraded forest needed to be given so that soil biological properties could present with a better results due to formation of more organic matter that could act as an essential medium for microbial growth which normally takes long period of time to form (Arunachalam and Pandey, 2003; Arifin *et al.*, 2012). Dinesh *et al.* (2003) studied the effects of deforestation and cultivation of wet tropical forest on the soil microbial indices found that the range of MBC in the moist deciduous forest with less human interferences possess 623 ± 11 to 674 ± 14 μg g^{-1} soil that is near to natural forest of Chikus Forest Reserve. As for planted forest of *Tectona grandis* and *Pterocarpus dalbergioides* in the previous study by Dinesh *et al.* (2003), the range of MBC was 143 ± 6 up to 154 ± 8 μg g^{-1} soil, but lesser than the amount of MBC possessed by planted forest of Chikus Forest Reserve. These results showed that multi-storied forest management enhances the MBC and MBN of soil at rehabilitated forests. The range of MBN for moist deciduous forest was between 50 ± 5 and 54 ± 5 μg g^{-1} soil. As for planted forest of *P. dalbergioides* and *T. grandis*, the range was between 11 ± 3 and 14 ± 4 μg g^{-1} soil. Planted forest of *P. dalbergioides* and *T. grandis* was established for more than 41 and 53 years, respectively (Dinesh *et al.,* 2003). However, the amount of MBC and MBN did not reach an equal or higher level as compared to planted forest of Chikus Forest Reserve which proves technique of forest rehabilitation gives a significant impact on the changes of soil microbial. As for correlation analysis carried out between soil microbial biomass C with organic matter and also between soil microbial biomass N with total N, results showed that the availability of the current amount of soil microbial biomass C and N did not show any positive contribution towards the availability of organic matter and total N in the natural and planted forests. (Karam *et al*., 2013b; Singh *et al*., 2013). Availability of organic matter also showed that it does not have positive contribution on MBC/MBN ratio.

Table 1 Selected soil physico-chemical properties Chikus and Tapah Hill Forest Reserves

Parameters	Natural	Planted	Enrichment	Secondary
		0-15 cm depth		
Organic matter (%)	12.31 ± 1.47a	8.52 ± 0.55b	16.99 ± 0.84a	12.12 ± 0.35b
Organic carbon (%)	7.14 ± 0.85a	4.94 ± 0.32b	9.86 ± 0.49a	7.03 ± 0.47b
Total nitrogen (%)	1.66 ± 0.20a	1.10 ± 0.15b	1.55 ± 0.09a	1.11 ± 0.09b
pH-H_2O	4.16 ± 0.08a	4.22 ± 0.03a	4.36 ± 0.11a	4.19 ± 0.05a
Bulk density (g cm^{-3})	1.21 ± 0.05a	1.36 ± 0.02b	1.16 ± 0.01a	1.24 ± 0.02b
Moisture content (%)	52.83 ± 2.91a	17.67 ± 1.33b	26.33 ± 0.61a	20.50 ± 1.91b

cont'd Table 1

	15-30 cm depth			
Organic matter (%)	9.63± 1.62a	8.07 ± 0.45b	14.29 ± 0.35a	11.27 ± 0.78c
Organic carbon (%)	5.58 ± 0.94a	4.68 ± 0.26b	8.29 ± 0.20a	6.54 ± 0.45c
Total nitrogen (%)	1.26 ± 0.25a	0.87 ± 0.06a	0.81 ± 0.05a	0.77 ± 0.10a
pH-H_2O	4.65 ± 0.10a	4.30 ± 0.04b	4.42 ± 0.10a	4.23 ± 0.08b
Bulk density (g cm^{-3})	1.35 ± 0.06a	1.39 ± 0.02a	1.22 ± 0.01a	1.26 ± 0.02a
Moisture content (%)	50.50 ± 2.67b	19.50 ± 1.78b	23.33 ± 0.49b	19.17 ± 2.60b

Note: Different letters in each row indicate significant differences between the means of soil properties at between forest plot using a *t*-test (p<0.05).

Table 2 Microbial biomass carbon (C) and nitrogen in Chikus and Tapah Hill Forest Reserves, Perak

		Chikus		Tapah	
Parameter	Depth (cm)	Natural	Planted	Enrichment	Secondary
MBC	0-15	824 ± 34a	542 ± 329b	465 ± 105b	158 ± 66c
	15-30	524 ± 115a	337 ± 233b	325 ± 58b	124 ± 35c
MBN	0-15	149 ± 20a	37 ± 4c	239 ± 8a	162 ± 18b
	15-30	113 ± 20a	9 ± 4c	134 ± 12a	78 ± 11b

Note: Different letters in each row indicate significant differences between the means of soil properties at between forest plot using a *t*-test (p<0.05).

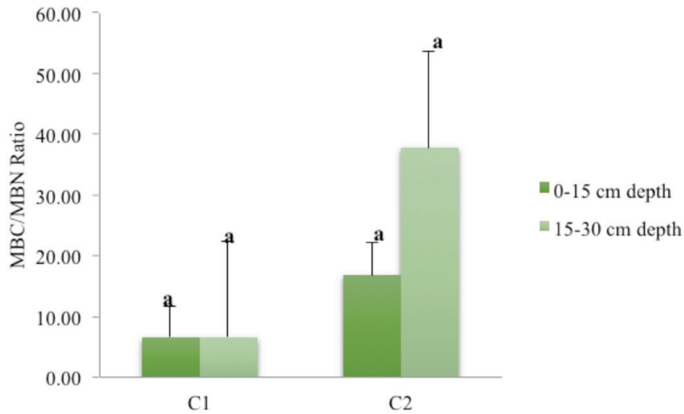

Figure 1 Total means of MBC/MBN ratio N of C1 and C2 plots. Different letters indicate significant differences between means for the same soil depths in natural forest (C1) compared to planted forest (C2) plots, using a Student's *t*-test (p<0.05). (Bars are means, whiskers indicate standard error of five soil replicates)

A greater amount of organic matter in planted forest is a valuable indication of greater amount of MBC. Islam and Weil (2000) suggested that abundance and thickness of the layer of litter as observed on the forest floor promote high decomposing processes by soil microorganisms. In addition, Powlson *et al.* (1987) claimed that the sensitivity posed by labile C is proportional to the limitation of soil microbial biomass and this affects organic C aggradation. MBN is greater in planted forest than in secondary forest for both soil depths (Figure 2). Higher MBN could be due to the higher total N availability possessed by planted forest compared to secondary forest. Kandeler *et al.* (2006) found that an increase in microbial N might be reflected by the competition between microorganism and plants in limited N ecosystem condition. Hence, these results proved that changes in N whether it increases or decreases catalysed the level of MBN as what we can observe in planted forest and secondary forest, respectively. Variation of the MBC/MBN ratio between planted forest and secondary forest showed the qualitative changes occurring in the soil biological composition as an effect of rehabilitation activities. The level of microbial biomass C was high for both planted and secondary forests compared to its microbial biomass N which indicated that mineralization of soil nutrients are higher compared to the rate of immobilization. Barbhuiya *et al.* (2004) explained that reasonably high soil organic substrate and low total N compared to organic C at both sites are believed to be the catalyst for MBC/MBN ratio. Comparing the amount of MBC found in the planted forest of *T. grandis* and *P. dalbergioides* is studied by Dinesh *et al.* (2003) showed that the amount of MBC (143 ± 6 to 154 ± 8 µg g^{-1} soil) was lower compared to planted forest (325 ± 58 to 465 ± 105 µg g^{-1} soil) of Tapah Hill Forest Reserve. The MBN levels of planted forest in Tapah Hill Forest Reserve (134 ± 12 to 239 ± 8 µg g^{-1} soil) also higher compared to planted forest of *T. grandis* and *P. dalbergioides* (11 ± 3 to 14 ± 4 µg g^{-1} soil) indicates enrichment planting technique gave better results of soil microbial compared to monoculture planting technique. Correlation analysis done between soil microbial biomass C with organic matter, soil microbial biomass N with total N and organic matter with MBC/MBN ratio results showed that the availability of the soil microbial biomass does not have any positive contributions towards organic matter constituents for planted and secondary forests.

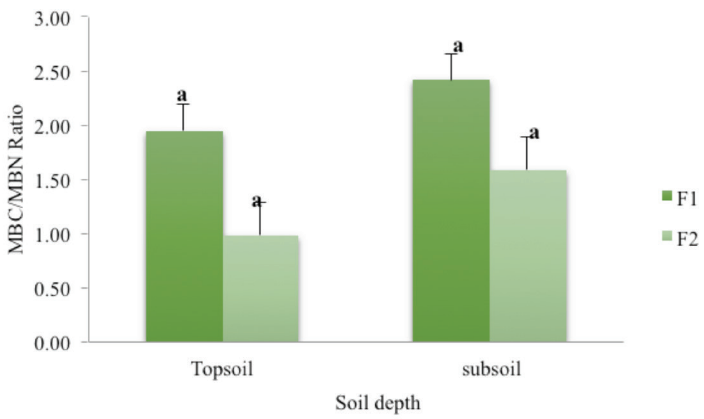

Figure 2 Ratio of microbial biomass C to microbial biomass N (MBC/MBN) between the same soil depths in planted forest (F1) and secondary forest (F2) plots. Different letters indicate significant differences between means for the same soil depths comparing planted forest (F1) to secondary forest (F2) plots, using a Student's *t*-test (p<0.05). (Bars are means, whiskers indicate standard error of five soil replicates).

Conclusion

Studies on forest soil fertility typically focus only on the physical and chemical properties, while the biological properties are always neglected. However, this study has shown that biological properties like microbial biomass carbon and nitrogen has the potential to provide faster indicators on current changes in the soil environment like deforestation and also rehabilitation. Microbial biomass carbon and nitrogen in the natural forest were found to be higher compared to the rehabilitated forests studied. However, the level of microbial biomass in planted forest of Chikus and Tapah Hill forests showed better results compared to secondary forest that underwent natural regeneration without any forest silviculture. Besides that, longer periods of time need to be given for forest soil to rehabilitate. As the study has proven, planted forests are able to reach the level of microbial biomass that is similar to a natural forest.

References

Arunachalam, A. and Pandey, H.N. 2003. Ecosystem restoration of jhum fallows in northeast India: microbial C and N along altitudinal and successional gradients. *Restor. Ecol.*, 11: 168-173.

Barbhuiya, A.R., Arunachalam, A., Pandey, H.N., Arunachalam, K., Khan, M.L. and Nath, P.C. 2004. Dynamics of soil microbial biomass C, N and P in disturbed stands of a tropical wet-evergreen forest. *European Journal of Soil Biology*, 40: 113-121.

Blaser, J., Sarre, A., Poore, D. and Johnson, S. 2011. Status of Tropical Forest Management, 2011. ITTO Technical Series No. 38.International Tropical Timber Organization, Yokohama, Japan.

Bremer, S.M. and van Kessel, C., 1992. Seasonal and microbial biomass dynamics after addition of lentil and wheat residues. *Soil Sci. Soc. Am. J.*, 56: 1141-1146.

Briones, M.J.I., Garnett, M.H. and Inesori, H. 2010. Soil biology and warming play a key role in the release of 'old C' from organic soils. *Soil Biology and Biochemistry*, 42: 960-967.

Brookes, P.C., Landman, A., Pruden, G. and Jenkinson, D.S. 1985. Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method to measure microbial biomass nitrogen in soil. *Soil Biol. Biochem.*, 17: 837-842.

Dinesh, R., Chandhuri, S.G., Ganeshamurthy, A.N. and Dey, C. 2003. Changes in soil microbial indices and their relationships following deforestation and cultivation in wet tropical forests. *Applied Soil Ecology*, 24: 17-26.

Forest Department of Peninsular Malaysia, 2003. Forest Plantation for the Future - A Record of The Multi-Storied Forest Management Project in Malaysia and The Small Scale Fast-Growing Forest Plantation Project in Malaysia.

Groffmann, P.M., McDowell, W.H., Myers, J.C. and Merriam, J.L. 2001. Soil microbial biomass and activity in tropical riparian forests. *Soil Biology and Biochemistry*, 33: 1339-1348.

Islam, K.R. and Weil, R.R. 2000. Land use effects on soil quality in a tropical forest ecosystem of Bangladesh. *Agriculture, Ecosystem and Environment*, 79: 9-16.

Jia, G., Cao, J., Wang, C. and Wang, G. 2004. Microbial biomass and nutrients in soil at the different stages of secondary forest succession in Ziwulin, northwest China. *Forest Ecology and Management*, 217: 117-125.

Joergensen, R.G., Anderson, T.H. and Wolters, T. 1995. Carbon and nitrogen relationship in the microbial biomass of soils in beech (*Fagus sylvatica*) forests. *Biol. Fertil. Soils*, 19: 141-147.

Kandeler, E., Mosier, A.R., Morgan, J.A., Milchunas, D.G., King, J.Y., Rudolph, S. and Tscherko, D. 2006. Response of soil microbial biomass and enzyme activities to the transient elevation of carbon dioxide in a semi-arid grassland. *Soil Biology and Biochemistry*, 38: 2448-2460.

Karam, D.S., Abdu, A., Radziah, O., Shamshuddin, J., Abdul-Hamid, H., Majid, N.M., Panicker, M., Halim, N.H.A., 2011. Assessing soil biological properties of natural and planted forests in the Malaysian tropical lowland dipterocarp forests. American Journal of Applied Sciences, 8(9): 854-859.

Karam, D.S., Arifin, A., Radziah, O., Shamshuddin, J., Majid, N.M., Hazandy, A.H., Zahari, I., Nor-Halizah, A.H., Rui, T.X., 2011. Impact of long-term forest enrichment planting on the biological status of soil in a deforested hill dipterocarp forest in Tapah, Perak, Malaysia. *The Scientific World Journal*.

Arifin. A., Karam, D.S., Shamshuddin, J., Majid, N.M., Radziah, O., Hazandy, A.H., Zahari, I., 2012. Proposing a suitable soil quality index for natural, secondary and rehabilitated forests in Malaysia. *African Journal of Biotechnology*, 11(14): 3297-3309.

Karam, D.S., Arifin, A.,Othman, R., Jusop, S., Abdul-Hamid, H., Seema, T., 2013. Changes in the physico-chemical properties of soils under rehabilitated lowland dipterocarps forest at Chikus Forest Reserve, Perak, Malaysia. T*he Journal of Macrotrends in Applied Science,* 1(1): 42-57.

Karam, D.S., Arifin, A., Shamshuddin, J., Radziah, O., Majid, N.M., Hazandy, A.H, Carole, L., 2012. Properties of soils in natural, secondary and rehabilitated forests in Chikus and Tapah Hill Forest Reserves, Peninsular Malaysia. *African Journal of Biotechnology*, (ISSN 1684-5315).

Karam, D.S., Arifin, A., Radziah, O., Shamshuddin, J., Abdul-Hamid, Majid, N.M., Zahari, I., Halim, N.H.A., Yen, C.K., 2013. Evaluation of soil biological properties of 9- and 15-year-old stands in the oil palm plantation in Perak, Malaysia. *African Journal of Agricultural,* 8 (29), 3904-3910.

Singh, K.S.D., Arifin, A., Radziah, O., Shamshuddin, J., Hazandy, A.H., Majid, N.M., 2013. Status of soil microbial population, enzymatic activity and biomass of selected natural, secondary and rehabilitated forests. *American Journal of Environmental Sciences*, 4(6): 301-309.

Karam, D.S., Abdu, A., Othman, R., Jusop, S., Hanif, A.H.M., 2014. Impact of enrichment planting activity on soil physic-chemical properties of degraded forestland. *International Journal of Environmental Sciences*, 5(2): 407-422.

Karam, D.S., Jamil, M.N.C., Jaafar, N., Rajoo, K.S., Arifin, A., 2015. Effects of Different Incubation Periods on Microbial Biomass Carbon (MBC) in Two Soil Series. *Journal of Environmental Science and Pollution Research*, 1(1): 20-22.

Karam, D.S., Abdu, A., Rajoo, K.S., Muharam, F.M., Kasim, M.R.M and Hamid, H.A., 2016. Proposed soil quality index for rehabilitated tropical forest sites in Malaysia. *American Journal of Applied Sciences*, DOI:10.3844/ajassp.

Karlen, D.L., Mausbach, M.J., Doran, J.W., Cline, R.G., Harris, R.F. and Schuman, G.E. 1997. Soil quality: a concept, definition, and framework for evaluation. *Soil Sci. Soc. Am. J.*, 61: 4-10.

Kwabiah, A.B., Palm, C.A., Stoskopf, N.C. and Voroney, R.P. 2003. Response of soil microbial biomass dynamics to quality of plant materials with emphasis on P availability. *Soil Biology and Biochemistry*, 35: 207-216.

Maithani, K., Arunachalam, A., Tripathi, R.S. and Pandey, H.N. 1996. Seasonal dynamics of microbial biomass C, N, and P during regrowth of a disturbed subtropical humid forest in northeast India. *Forest Ecology and Management,* 4: 31-37.

Malik, M.N.A., Abdu, A., Karam, D.S., Jusop, S., Hamid, H.A., Jamaluddin, A.S., Basri, M.H.A., Mohd, N.P., 2015. The fertility status of soils at rehabilitated degraded land in Universiti Putra Malaysia Planted with *Pinus caribaea* and *Swietenia macrophylla*. American Journal of Applied Sciences, 12(10): 752-758.

Mohd-Aizat, A., Mohamad-Roslan, M.K., Sulaiman, W.N.A., Karam, D.S., 2014. The relationship between soil pH and selected soil properties in 48 years logged-over forest. International Journal of Environmental Sciences, 4(6): 1129-1140

Montagnini, F., Eibl, B., Grance, L., Maiocco, D. and Nozzi, D. 1997. Enrichment planting in overexploited subtropical forests of the Paranaense region of Misiones, Argentina. *Forest Ecology and Management*, 99: 237-246.

Moura-Costa, P.H. 1993. Large scale enrichment planting with dipterocarps, methods and preliminary results. In Procedings of the Yogyakarta Workshop, BIO/REFOR/IUFRO/SPDC: Bio-reforestation in Asia-Pacific Region. Suzuki, K., Sakurai, S. and Ishii, K. (Eds.) pp. 72-77.

Müller, E., Wildhagen H., Quinstern, M., Heβ, J., Wichern, F. and Joergensen, R.G. 2009. Spatial patterns of soil biological and physical properties in a ridge tilled and a ploughed Luvisol. *Soil and Tillage Research*, 105: 88-95.

Nsabimana, D., Haynes, R.J. and Wallis, F.M. 2004. Size, activity and catabolic diversity of the soilmicrobial biomass as affected by land use. *Applied Soil Ecology*, 26: 81-92.

Powlson, D.S., Brookes, P.C. and Christensen, B.T. 1987. Measurement of soil microbial biomass provides an early indication of changes in the total soil organic matter due to straw incorporation. *Soil Biol. Biochem.*, 19: 159-164.

Raubuch, M. and Joergensen, R.G. 2002. C and net N mineralization in a coniferous forest soil: the contribution of the temporal variability of microbial biomass C and N. *Soil Biology and Biochemistry*, 34: 841-849.

Salamanca, E.F., Raubuch, M. and Joergensen, R.G. 2006. Microbial reaction of secondary tropical forest soils to the addition of leaf litter. *Applied Soil Ecology*, 31: 53-61.

Trockenbedt, M., Imiyabir, Z. and Josue, J. 2002. Hollow logs and logging residues from Deramakot Forest Reserve, Sabah, Malaysia. *Forest Ecology and Management*, 165: 141-150.

Vance, E.D., Brookes, P.C. and Jenkinson, D.S. 1987. An extraction method for soil measuring soil microbial biomass C. *Soil. Biol. Biochem.*, 19: 703-707.

Vásquez-Murrieta, M.S., Govaerts, B. and Dendooven, L. 2007. Microbial biomass C measurements in soil of the central highlands of Mexico. *Applied Soil Ecology*, 35: 432-440.

Witt, C., Gaunt, J.L., Galicia, C.C., Ottow, J.C.G. and Neue, H.-U. 2000. A rapid chloroform-fumigation extraction method for measuring soil microbial carbon and nitrogen in flooded rice soils. *Biol. Fertil. Soils*, 30: 510-519.

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Selected Liming Materials for Improvement of Rice Production on an Acid Sulfate Soils of Merbok Paddy Granary Area in Peninsular Malaysia

Elisa Azura Azman, Seishi Ninomiya, Shamshuddin Jusop, Tasuku Kato, Uraiwan Tongkaemkaew, Hamdan Jol, Mohd Izuan Effendi Halmi and Roslan Ismail

Introduction

Soils in Malaysia are highly weathered soils classified as Ultisols and Oxisols under the USDA Soil Taxonomy classification. These soils are known to be moderately acidic to highly acidic in nature with soil pH range from pH 2.9 to 5, low in essential macro-nutrients such as N, P, K, Ca and Mg, plus, low in micronutrients such as Mn, Zn and Fe. The prevailing condition of Ultisols and Oxisols are not suitable for sustainable rice production. Besides that, these soils are also known to be deficient in available Si, arisen from extensive leaching processes (Crooks and Prentica, 2012) . However, in Malaysia, part of these soils known as an acid sulphate soils have been used for rice production with an average yields of 3.7 t ha^{-1} $season^{-1}$ in year 1995 (Najim *et al*., 2007).

Rice is a staple food in Malaysia, and the Malaysian Government acknowledges the need to achieve 100% self-sufficiency level in paddy production by year 2020. Thus, the need to increase rice production is paramount. Following that, rice production can be increased in two possible way: 1) physically increase the rice production area, and/or 2) increase the rice yield per unit area. In the past 30 years or so, with no rice area expansion and decline in rice yield per unit area, rice production has fallen below the Malaysia market demand (Shamshuddin, 2006).

Land is valuable comodity, open new area for agriculture (especially to increase rice production is not feasible), thus one good alternative is the use of marginal land. One of the marginal land with some potential is an acid sulfate soils like in Merbok paddy granary area (Fig. 1) of Peninsular

Malaysia. Mainly, acid sulfate soils is not suitable for crop production without proper soil amelioration (i.e., liming the soil). As such, liming is a common approach to improve acidic soil prior to crop production.

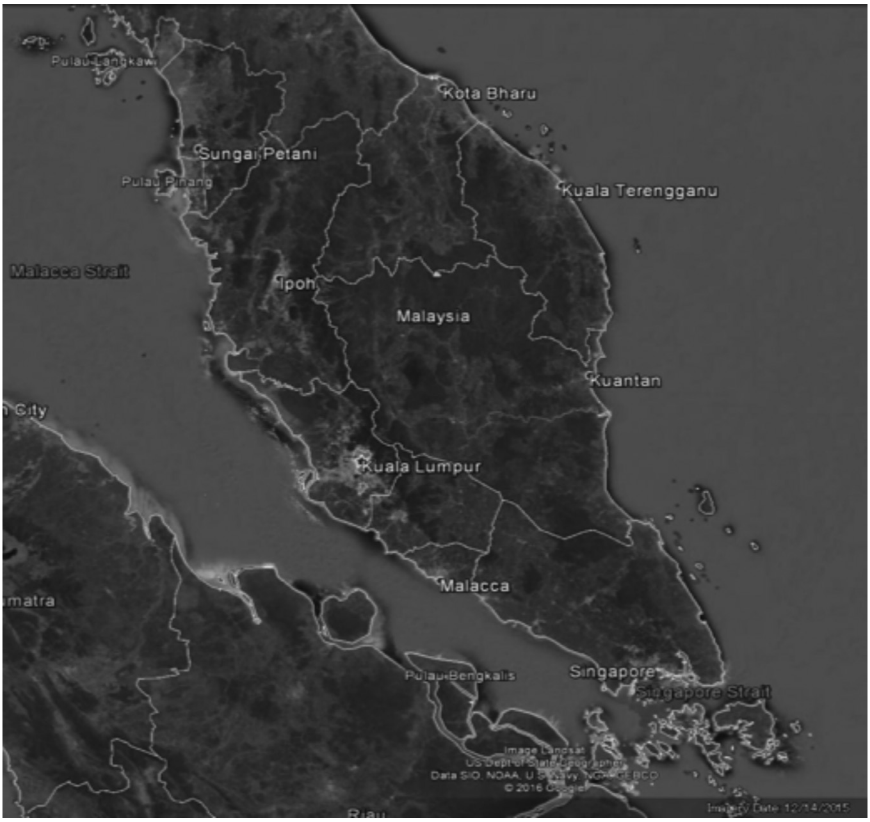


Fig. 1 Satellite photo of Peninsular Malaysia. The Merbok paddy granary area at northern region of Peninsular Malaysia (red circle). Source: Google Earth.

In Malaysia, various liming materials are available locally, such as ground magnesium limestone (GML), hydrated lime ($Ca(OH)_2$) and liquid lime. Britestone Sdn. Bhd is one example of a local liming material company. This company mines limestone from the limestone hills of Ipoh, Perak, west coast of Peninsular Malaysia, and the product called as ground magnesium limestone (referred as GML) after finely ground. Liming practices improves soil pH and subsequently precipitates Al as inert Al-hydroxides, thereby reducing Al toxicity (Shamshuddin, *et al.*, 2010) in an acid sulfate soils. Besides that, GML also supply significant amount of Ca and Mg into the soil system for cop uptake.

Rice yield can be increase to 4.5 t ha^{-1} $season^{-1}$ with annual GML application of 2 t ha^{-1} (Ting *et al.*, 1993). Whereas, application of 4 t ha^{-1} GML produce yield of 3.53 and 4.21 t ha^{-1} in Kedah and Kelantan areas, respectively (Elisa *et al.*, 2014; Shazana *et al.*, 2014). In some instance, application of 4 t ha^{-1} GML showed a beneficial effect that lasted for eight years with the effect being comparable to application of 1 t ha^{-1} GML anually (Shamshuddin *et al.*, 1998).

Secondly is hydrated lime ($Ca(OH)_2$) referred as HL. According to the rice farmers, hydrated lime shows immediate effect upon application to the soil; therefore, they apply hydrated lime every season prior to rice planting.

Third is the liquid lime referred as LL was introduced by Humibox (M) Sdn. Bhd., a private company. The company claims that the application of lime in a liquid form is more effective in correcting soil acidity compared with the powder form such as GML and HL (http://www.humibox.com/products/soil-acidity-correctors/liquid-lime). The field trials were conducted in Seberang Perak area and the rice yield increased by 28.5% after application of liquid lime in 2004 compared to 2003 (unpublished).

A fact to note that, liming material (GML, HL and/or LL) application is able improve the fertility of marginal acidic soil for crop production. Therefore, the objectives were to access the efficiency and cost-effectiveness of GML, HL and LL as selected liming materials to increase rice yield under an acid sulfate soils granary area in Peninsular Malaysia.

Soil Chemical Properties Prior to Rice Planting

Initial soil chemical characteristics for an acid sulfate soils prior rice planting are given in Table 1. Soil pH for topsoil (0–15 cm) indicated high acidity with a pH < 3, whereas exchangeable Al (6.7 $cmol_c$ kg^{-1}) exceeded the critical Al level for rice growth (2 $cmol_c$ kg^{-1}) according to Dobermann and Fairhurst (2000) and Hiradate and Matsumoto (2007). Soil pH decreased with soil depth, whereas exchangeable Al increased with soil depth. Soil pH values clearly indicated high soil acidity within the studied areas. Exchangeable calcium (Ca) was relatively low compared to the minimal requirement of Ca for rice growth at 2 $cmol_c$ kg^{-1} (Table 1), according to Palhares *et al.* (2001).

Table 1 Soil chemical characteristics for an acid sulfate soil at different depths prior to rice sowing.

Depth (cm)	pH water (1:2.5)	EC (dS m^{-1})	Exchangeable cations ($cmol_c$ kg^{-1})					Fe (mg kg^{-1})	CEC ($cmol_c$ kg^{-1})	Total Carbon (%)	Total Nitrogen (%)	C:N ratio	Available P (mg kg^{-1})	Al saturation (%)
			K	Ca	Mg	Na	Al							
0-15	2.63	0.71	0.46	1.72	1.45	0.49	6.70	525.00	11.14	2.78	0.19	14.63	12.18	61.92
15-30	2.60	1.54	0.15	1.53	1.89	0.52	8.63	284.70	11.35	1.82	0.10	18.20	10.22	67.84
30-45	2.56	1.85	0.18	1.89	2.14	0.55	8.12	316.40	11.93	1.89	0.10	18.90	10.08	63.04
45-60	2.58	2.41	0.18	1.77	2.69	0.73	8.45	307.50	13.35	2.30	0.10	23.00	12.46	61.14
60-75	2.89	5.00	0.19	2.12	3.35	1.07	9.86	560.55	15.57	3.54	0.12	29.50	15.54	59.43

Soil pH was determined in water at the soil to solution ratio of 1:2.5 using a pH meter (Sartorius pH meter PB-11).
EC was determined in water at the soil to solution ratio of 1:5 using a Radiometer CDM 210 instrument.
The basic cations (K, Ca, Mg, Na) in the 1 M NH_4OAc solution were determined by atomic absorption spectroscopy (AAS) (Perkin Elmer, model 1100B)
Cation exchange capacity was determined by 1 M NH_4OAc, which was buffered at pH 7
Exchangeable Al was extracted by 1 M KCl, and the Al in the extract was determined by AAS (Perkin Elmer, model 1100B)
Extractable Fe (Dilute Double Acid method) was also determined by AAS (Perkin Elmer, model 1100B).
Total carbon and nitrogen were determined by the CNS Analyzer Leco RC-412 (Leco Corporation, St. Joseph, MI).Available P was determined using Bray II extracting reagent (0.1 N HCl with 0.03 N NH_4F) and was measured by auto-analyser (AA) (Lachat QuickChem® 8000 Series FIA + Syatem; Lanchat Instruments, Loveland, USA).

In contrast, magnesium (Mg) met the Mg requirement of 1 $cmol_c$ kg^{-1} for rice plants (Dobermann and Fairhurst, 2000). It was clear that both Ca and Mg increased with soil depth. Total nitrogen (TN) was low. Total carbon (TC) was above 2%, thereby providing a C:N ratio of between 14.6–32.5, sufficient to support microbial activity in the root-rhizosphere system. The higher carbon content of these soils can be attributed to the residual effect of *in-situ* rice straw decomposition after each harvest. Available P was noted to be sufficient, within the 7–20 mg kg^{-1} P for rice growth (Dobermann and Fairhurst, 2000).

Field Condition of An Acid Sulfate Soils

Besides, the physical-chemical limitation of cultivating rice in an acid sulfate soils, generally, in Peninsular Malaysia, several other challenges are present, such as rice blast (Figure 2), dry condition (Figure 3a), and excess water (Figure 4) in the paddy field. Rice blast is caused by the *Magnaporthe grisea* fungus. It is also known as rice blast fungus, rice rotten neck, rice seedling blight, blast of rice, oval leaf spot of graminea, pitting disease, ryegrass blast, and Johnson spot. It is a plant-pathogenic fungus that causes a serious disease affecting rice, thus decrease rice yield within 3-5 days after infection.

In some season, the changes in the Peninsular Malaysia climatic condition makes the soil dry and hard, this scenario often makes the soil hard to till, thus water have to be pumped from the nearby water source (Figure 3b). This all add up to the rice cultivation cost.

Peninsular Malaysia is classified as $A_{f,}$ under Köppen climatic classification. Under such scenario it represent tropical rainforest climate marked with average precipitation of at least 60 mm (2.4 in) in every month. This indicate high amount of rainfall, thus, flooding hazard is another important factor in rice cultivation (Figure 4) in Peninsular Malaysia.

Fig. 2 Rice field condition after being attacked by rice blast at 90 days after sowing (DAS).

(a)

(b)

Fig. 3 Dry conditions the first week after sowing (a) and water was pumped in from a drainage canal (b) (for 1st season).

Fig. 4 The experimental field condition during the second season with excess water (flooding hazard).

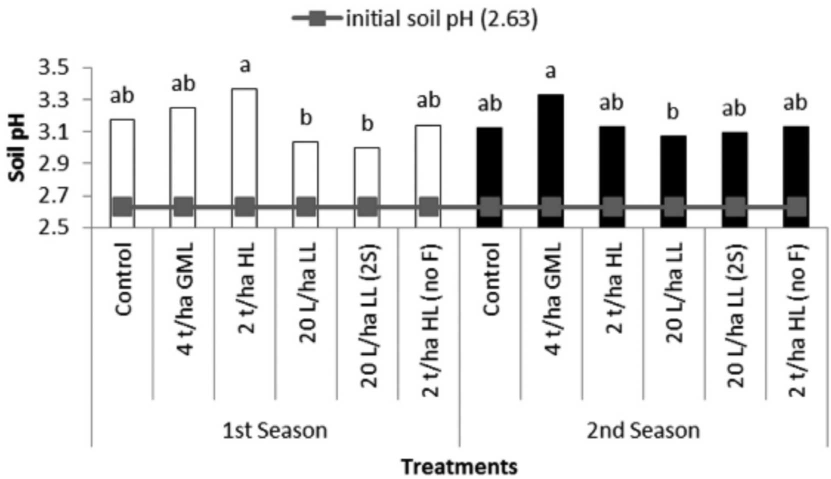
Effect of Various Liming Materials Treatment on pH, Al and Fe Content of The Acid Sulfate Soils

Soil pH of the topsoil before liming materials application was pH 2.63. Figure 5 shows soil pH improvement between 2.99 to 3.36 with application of various liming materials as shown in Table 2. Furthermore, Table 3 shows the elemental composition of the selected liming materials used for the rice cultivation, with GML used in the powder form contains about 30-35% of available Ca (in the form of CaO) compared to LL with about 99% of available Ca (in the form of $CaCO_3$).

In the first season, the soil treated with 2 t ha^{-1} HL showed a significant increment of soil pH compared to soil treated with 20 L ha^{-1} LL. Meanwhile, for the second season, the soil pH of the soil treated with 4 t ha^{-1} increase significantly compared to soil treated with 20 L ha^{-1} LL.

No significant effect for exchangeable Al was noted among treatments after the first and second seasons of rice cultivation. However, the exchangeable Al after the second season harvest decreased compared to the first season. The reductions in exchangeable Al were between 7.43% to 21.55% for all treatments, except for the soil treated with 2 t ha^{-1} HL without fertilizer application. In addition, there was no significant effect among treatments for Fe, available P, total C, and total N after the first and second seasons, as shown in Table 4.

The effects of treatments on the pH, Al and Fe content in water during the first and second seasons are shown in Figure 6. During the first season, the water samples were analyzed every week from 14 DAS until 119 DAS, whereas during the second season, the water samples were analyzed only up to 77 DAS due to dry conditions. The pH of water ranged from 3.43 to 5.96 (first season) and from 3.46 to 7.13 (second season). The pH values of water for soil treated with GML and HL were higher compared to the control and soil treated with LL throughout the first season. During the second season, the pH of water increased proportionally with rice growth from 28 DAS until 64 DAS for all treatments. Aluminium (Al) content in water for the first season was higher than during the second season, with the value ranging from 0.06 to 39.85 mg L^{-1} and from 0.15 to 4.12 mg L^{-1}, respectively. During the first season, the Al content was < 2 mg L^{-1} from 49 DAS onwards until the harvest. Water samples from soil treated with LL contained higher amount of Al compared to soil treated with GML, HL and untreated soil. The iron content values were > 1 mg L^{-1} throughout rice cultivation for both season, except during the first season at 21 DAS with the highest value of 87.53 mg L^{-1} (soil treated with LL).



*GML: ground magnesium limestone
*HL: hydrated lime
*LL: liquid lime
*2S: apply on both seasons
*no F: No fertilizer applied

Fig. 5 Effect of treatments on soil pH. Means marked with the same letter for each season were not significantly different at $p < 0.05$ (Tukey's test).

Table 2 Selected liming materials used for the rice cultivation experiment.

Symbol	Treatments
T1	Control (no liming material)
T2	[a]Ground magnesium limestone (GML) (4 t ha^{-1})
T3	[a]Hydrated lime (2 t ha^{-1})
T4	[b]Liquid lime (applied only for first season) (20 L ha^{-1})
T5	[b] Liquid lime (applied only for first and second season) (20 L ha^{-1})
T6	[a]Hydrated lime (no fertilizer applied) (2 t ha^{-1})

[a] Applied one month prior rice seeding
[b] Applied one day prior rice seeding

Table 3 Elemental composition of the selected liming materials used for the rice cultivation.

Elemental composition	Ground magnesium limestone (GML)	Hydrated lime (HL)	Liquid lime (LL)
pH	8.93	9-10	9.1
CaO (%)	31-38	65	99% as $CaCO_3$
MgO (%)	15-18	n.a	0.2
SiO_2 (%)	< 0.2	n.a	n.a
Fe_2O_3 (%)	< 0.2	n.a	n.a
Al_2O_3 (%)	n.a	n.a	n.a
Particle size	100% passing thru a 20 mesh screen, 70% passing thru a 100 mesh screen and > 40% passing thru 200 mesh screen.	200 mesh size	1.5-5 µm
Chemical formula	$CaMg(CO_3)_2$	$Ca(OH)_2$	$CaCO_3$
Calcium carbonate equivalent	109	135	100

Table 4 Effect of treatments on selected soil chemical properties after the harvests of the first and second seasons.

Season	Treatments	Exch. Al ($cmol_c\ kg^{-1}$)	Exch. Ca ($cmol_c\ kg^{-1}$)	Exch. Mg ($cmol_c\ kg^{-1}$)	CEC ($cmol_c\ kg^{-1}$)	Fe ($mg\ kg^{-1}$)	P ($mg\ kg^{-1}$)	C (%)	N (%)
1st season	Control	7.27^{a}	0.51^{abc}	2.81^{b}	15.57^{ab}	333.32^{a}	11.90^{a}	2.85^{a}	0.28^{a}
	4 t ha^{-1} GML	8.35^{a}	0.70^{ab}	3.39^{a}	19.07^{a}	309.52^{a}	12.48^{a}	2.92^{a}	0.35^{a}
	2 t ha^{-1} HL	7.28^{a}	0.77^{a}	2.94^{b}	14.03^{b}	281.97^{a}	13.32^{a}	3.05^{a}	0.33^{a}
	20 L ha^{-1} LL	8.67^{a}	0.37^{bc}	2.96^{b}	14.41^{b}	264.45^{a}	12.17^{a}	2.42^{a}	0.30^{a}
	20 L ha^{-1} LL (2S)	8.72^{a}	0.33^{c}	3.07^{b}	15.29^{ab}	198.52^{a}	12.76^{a}	2.67^{a}	0.35^{a}
	2 t ha^{-1} HL (no F)	7.22^{a}	n.a	n.a	n.a	295.57^{a}	12.57^{a}	3.52^{a}	0.37^{a}
2nd season	Control	6.73^{a}	0.60^{b}	3.01^{b}	13.9^{ab}	358.36^{a}	11.47^{a}	2.74^{a}	0.32^{a}
	4 t ha^{-1} GML	6.43^{a}	0.98^{a}	3.99^{a}	15.31^{a}	371.96^{a}	10.72^{a}	2.95^{a}	0.32^{a}
	2 t ha^{-1} HL	6.14^{a}	0.95^{a}	3.27^{b}	13.30^{b}	365.93^{a}	11.79^{a}	2.74^{a}	0.27^{a}
	20 L ha^{-1} LL	6.87^{a}	0.49^{b}	3.15^{b}	13.66^{ab}	335.18^{a}	11.19^{a}	2.39^{a}	0.24^{a}
	20 L ha^{-1} LL (2S)	6.84^{a}	0.45^{b}	3.07^{b}	14.29^{ab}	316.50^{a}	13.02^{a}	2.20^{a}	0.24^{a}
	2 t ha^{-1} HL (no F)	7.50a	n.a	n.a	n.a	421.66^{a}	12.72^{a}	3.20^{a}	0.27^{a}

Means marked with the same letter for each season were not significantly different at $p < 0.05$ (Tukey's test)
*n.a: data not available

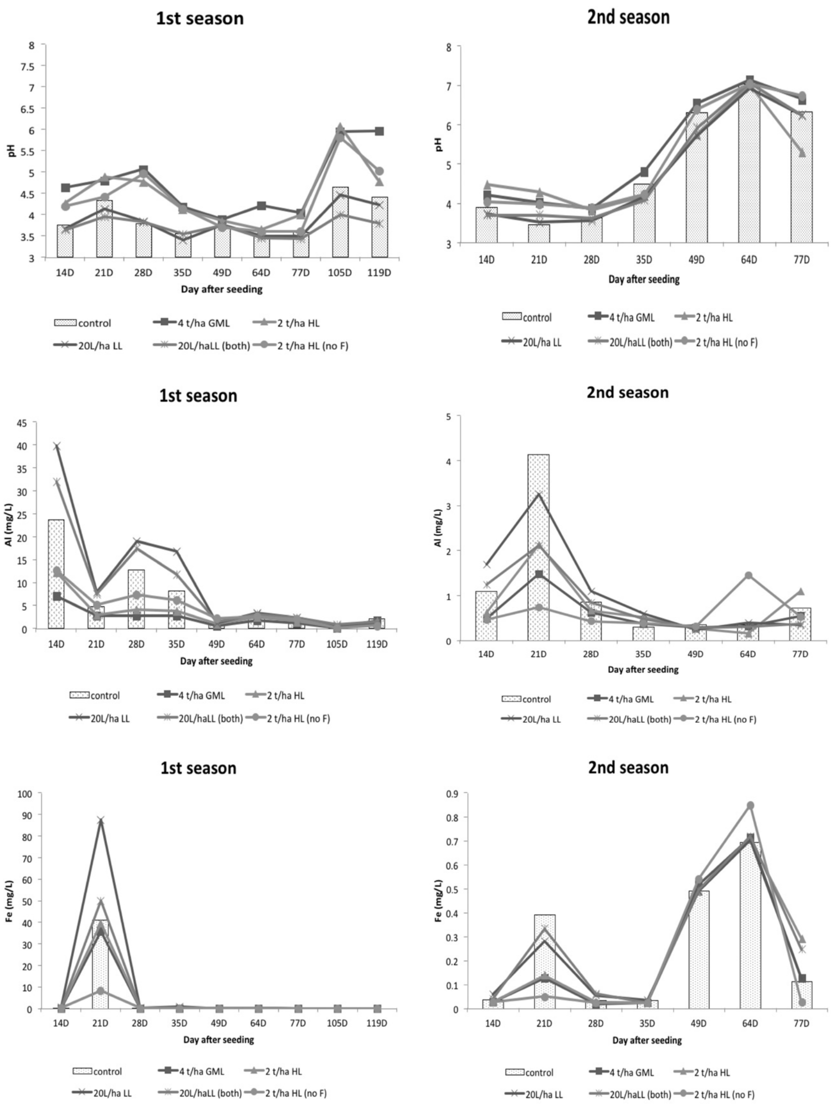


Fig. 6 Effect of treatments on pH, Al, and Fe (in water samples) during the first and second seasons

Rice Yield Components Analyses

Table 5 shows the rice yield components for two consecutive rice-cultivation seasons. During the first rice-season, the yield ranged from 1.79 t ha^{-1} to 3.71 t ha^{-1}, whereas for the second rice-season, the yield ranged from 3.21 t ha^{-1} to 4.34 t ha^{-1}. A significant difference was observed in rice yield among treatments during the first season: 1) the rice yield for soil treated with 4 t ha^{-1} GML was significantly higher than soil treated with LL; 2) soil treated with HL in-combination with GML showed significant effects compared to untreated soil and LL treatments on 1,000 grain weight. A significant effect was observed between the two seasons of rice cultivation for tillers number (1 m^2), 1,000 grain weight (g), spikelets per panicle, and percentage filled spikelets. The excess of water in the experimental plots (during seeding) may have promoted rice plant growth. This effect was observed in significantly lower tillers number during the second season. During the second season, no significant effects were found for rice yield and rice yield components among treatments. The rice yield increased by 21.21% to 49.15% during the second season.

Calcium and Aluminum Content in Rice Plants

Calcium and aluminum contents in the aboveground parts and root at 75 DAS for both seasons are shown in Figure 7. The calcium content in the root was lower than in the aboveground parts of rice plants. A significant effect was observed for Ca content between seasons in the aboveground parts and roots. No significant effects of Ca among the treatments during the second season in aboveground parts and roots were observed. Soil treated with 4 t ha^{-1} GML showed a similar effect with 2 t ha^{-1} HLwithout fertilizer, and these treatments were significantly higher than in soil treated with LL from first season. Calcium content in roots ranged from i) first season: 0.0003% to 0.0019% and, ii) second season: 0.0020% to 0.0040%, with increment of Ca in root rice root system up to 90%.

Aluminum content in the root was higher than in the aboveground parts. No significant effect on Al content among treatments was observed in aboveground parts during the second season. Soil treated with 4 t ha^{-1} GML was significantly higher in Al content compared to untreated soil during the first season. In addition, no significant effect was observed on Al content in roots among treatments for both seasons.

Table 5 Rice yield components for two rice-cultivation seasons.

Season	Treatments	Tillers number (m^{-2})	# Spikelet per panicle	% filled grains	1000 grain weight (g)	grain yield ($t\ ha^{-1}$)
1st season						
S1	T1	198^{a}	119^{a}	47.79^{ab}	23.00^{b}	2.71^{ab}
S1	T2	228^{a}	131^{a}	48.43^{ab}	25.31^{a}	3.71^{a}
S1	T3	216^{a}	118^{a}	53.26^{a}	24.70^{a}	3.38^{ab}
S1	T4	190^{a}	100^{a}	40.62^{b}	22.80^{b}	1.79^{b}
S1	T5	207^{a}	103^{a}	41.60^{ab}	22.36^{b}	1.97^{b}
S1	T6	226^{a}	111^{a}	51.78^{ab}	24.99^{a}	3.26^{ab}
2nd season						
S2	T1	152^{a}	144^{a}	71.46^{a}	24.89^{a}	3.95^{a}
S2	T2	169^{a}	153^{a}	71.57^{a}	24.24^{a}	4.34^{a}
S2	T3	168^{a}	149^{a}	68.52^{a}	24.89^{a}	4.29^{a}
S2	T4	151^{a}	134^{a}	70.58^{a}	25.12^{a}	3.52^{a}
S2	T5	164^{a}	131^{a}	68.62^{a}	24.90^{a}	3.68^{a}
S2	T6	161^{a}	126^{a}	63.12^{a}	24.90^{a}	3.21^{a}

Means marked with the same letter for each season were not significantly different at $p < 0.05$ (Tukey's test)

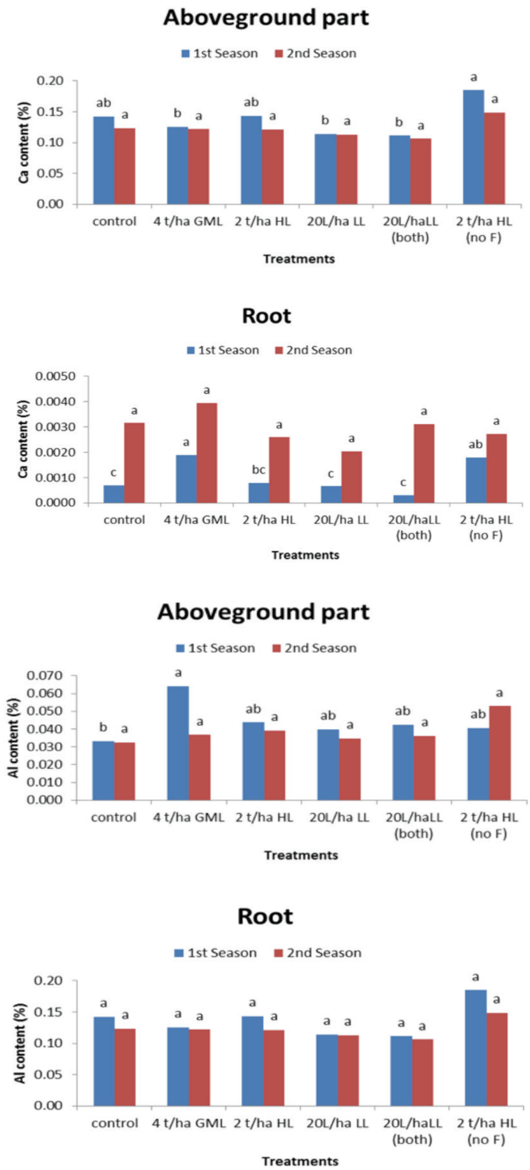


Fig. 7 Calcium and aluminum content in the aboveground parts and roots at 75 days after sowing (DAS). Means marked with the same letter for each season were not significantly different at $p < 0.05$ (Tukey's test).

Ameliorative Effect of Selected Liming Materials

The soil investigated was low in pH and high in exchangeable Al (Table 1). Soil pH throughout the soil profile was < 3.5. Exchangeable Al in the soil was very high throughout the soil depth. The topsoil (0–15 cm depth) comprises the zone where development of rice roots occurs. The pH values and exchangeable Al of the topsoil were 2.63 and 6.70 $cmol_c$ kg^{-1}, respectively. The pH values were lower than the critical level for rice production, whereas exchangeable Al exceeded the critical level for rice production of 1–2 mg kg^{-1} as suggested by Dobermann and Fairhurst. (2000). The pH and concentration of Al in the water at the soil pit was 3.70 and 878 µM, respectively. The concentration of Al far exceeded the critical toxic level of 74 µM for rice growth (Dent, 1986).

The favorable pH for optimal rice (MR 219) root growth is 6 (Elisa *et al.*, 2011; Alia *et al.*, 2011). However, to raise the pH up to this level is costly, and many ordinary farmers may not be able to afford the measures required. Aluminium toxicity can occur in soil when pH < 3.5 (Van Breemen and Pons, 1978). A study conducted in Japan showed that the growth of an Al-tolerant rice variety began to be inhibited when the Al^{3+} ion concentration exceeded 900 µM (Cate and Sukhai, 1964). This value is close to the aluminium concentration observed in the current study at 878 µM; thus, rice growth in the current study area can be inhibited by Al. However, the rice plants grew well and provided a reasonable yield (Shamshuddin *et al.*, 2013) stated that Al^{3+} is attracted to the negatively-charged cell walls of rice roots, thereby triggering the rice roots to secrete citric, oxalic, and malics acids. These acids in turn chelate the Al^{3+}, rendering it inactive. This defense mechanism allows rice plants to reduce the effects of Al^{3+} toxicity.

A total of three liming materials were studied, namely ground magnesium lime (GML), hydrated lime (HL), and liquid lime (LL). The rate of liming materials studied were based on the farming practice in Malaysia, as shown in Table 2. All the liming materials used are available locally. Normally, farmers at the study area apply 2 t ha^{-1} HL during the dry season (approximately March to April), because it is easier for machinery to enter the plot for lime application during this period. The farmers select HL because it provides rapid effects on their rice field upon application. From this experiment, the soil pH for soil treated with 2 t ha^{-1} HL (T3) showed the highest soil pH value of 3.36 compared to other treatments during the first season (Fig. 5). However, the soil pH decreased by 6.84% to a value of 3.13 during the second season. This indicated that the effect of HL on soil pH was fast and significant, however only short-term.

Soil treated with 4 t ha^{-1} GML provided the highest rice yield for both seasons with 3.71 t ha^{-1} and 4.34 t ha^{-1} (Table 5), respectively. Besides increment of rice yield, 4 t ha^{-1} GML also significantly increased 1,000 grain weight compared to the control and soil treated with LL (Table 5). No significant effect of treatments on spikelet $panicle^{-1}$ and tillers number for soil treated with 4 t ha^{-1} GML, however they produced the highest spikelet and panicle and tiller numbers of 132 and 228 m^{-2}, respectively. This finding is further supported by Ting *et al* (1993), noted that application of 2 t ha^{-1} GML annually increased the rice yield to 4.5 t ha^{-1}, which is higher than the national average rice yield of 3.8 t ha^{-1}. The rice yields achieved with the selected liming treatments are all higher compared to normal farming practices (± 2 t ha^{-1}).

GML application showed positive effect as it significantly increased exchangeable Mg compared to other treatments, with values achieved during the first and second seasons of 3.39 and 3.99 $cmol_c$ kg^{-1}. This value was above the critical value of 1 $cmol_c$ kg^{-1} (Dobermann and Fairhurst, 2000). In addition, GML significantly increased exchangeable Ca for both seasons. However, exchangeable Ca was below the require level of 2 $cmol_c$ kg^{-1} (Palhares *et al.*, 2001). A higher amount of Ca in the soil results in rice plants that are able to assimilate higher amounts of Ca as shown in Figure 7. The positive effects shown by GML occur according to the following reactions:

$$(Ca, Mg)(CO_3)_2 \rightarrow Ca^{2+} + Mg^{2+} + CO_3^{2-} \qquad \text{(Equation 2)}$$

$$CO_3^{2-} + H_2O \rightarrow HCO_3^- + OH^- \qquad \text{(Equation 3)}$$

$$Al^{3+} + 3OH^- \rightarrow Al(OH)_3 \qquad \text{(Equation 4)}$$

GML dissolves readily on application to the acidic soil, releasing Ca and Mg (Equation 2), and these macronutrients could be assimilated by the growing rice plants. Subsequently, hydrolysis of CO_3^{2-} (Equation 3) would produce hydroxyls that neutralized Al by forming inert Al-hydroxides (Equation 4). In addition, soil to which GML was applied showed significantly increased cation exchange capacity (CEC) compared to soil which received LL (Table 4).

The ameliorative effects of liming materials were improved during the second season (Fig. 8 and 9). The relationships between soil pH and relative rice yield during the first and second seasons are shown in Figure 8, as this indicates that the relative rice yield positively correlated with soil pH. The regression line for the second season shifted to a higher level, showing that the relative rice yield had increased after the first season, even though the soil pH ranged between 2.9 to 3.4 (high acidity) for both seasons. Despite this fact, there was only a 10% drop in relative rice yield corresponding to a change in soil pH of 3.38 and 3.17 for first season and second seasons, respectively.

The relationship between exchangeable Al and relative rice yield are shown in Figure 9. After the second season, the regression line shifted to the left. This indicated that the Al toxicity had been reduced (ameliorative effect) after the second season. A 10% drop in relative rice yield corresponding to exchangeable Al of 6.13 and 6.67 $cmol_c\ kg^{-1}$ for first season and second seasons, respectively was evident. However, no significant difference was observed among treatments for exchangeable Al (Table 4).

Soil treated with LL produced a rice yield of < 2 t ha^{-1} during the first season, and consistently acidic soil pH of 3 was noted. Liquid lime (LL) applied in liquid form, and lime in solution form is suspected to be quickly leached through the soil column. In addition, the particle size of LL (1.5–5 μM) is far smaller than GML (20–200 mesh size) and HL (± 200 mesh size), which are in a powdered form. Smaller size particles are often subjected to faster losses through leaching than larger size particles.

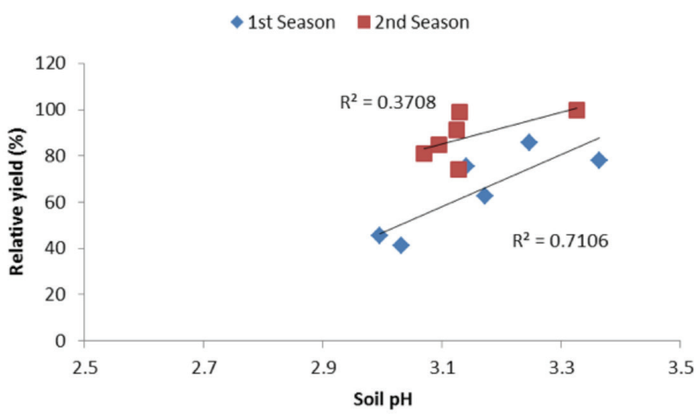


Fig. 8 Relationships between soil pH and relative yield during first and second seasons.

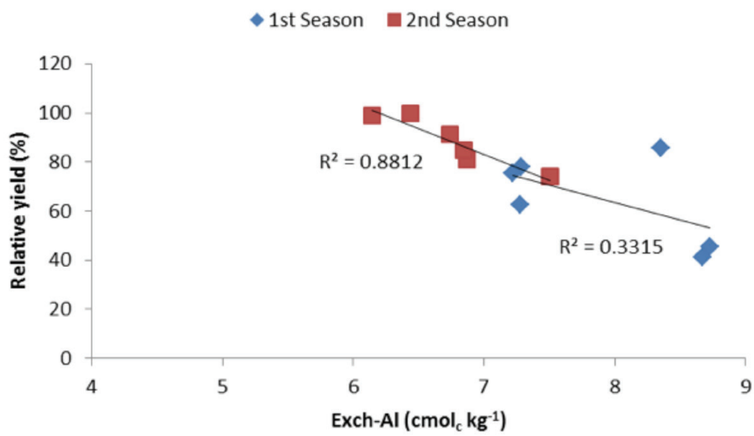


Fig. 9 Relationships between exchangeable Al and relative yield during the first and second seasons.

Growing rice in an area with low pH and high Al concentration would inhibit the elongation of plant roots (Elisa *et al.*, 2011; Alia *et al.*, 2015; Horst *et al.*, 2009). The disruption of the root cap forming processes and a decline in cell division and deposition of lignin would occur (Susan *et al.*, 2007). As a result, nutrient uptake is curtailed, and multiple nutrient deficiencies occur (Godbold *et al.*, 1988; Ridolfi and Garrec, 2000). These assertions have been confirmed which showed that the concentration of Ca in roots was significantly higher with 4 t ha^{-1} GML application compared to control, 2 t ha^{-1} HL, and 20 L ha^{-1} LL application (Fig. 7). As an improved ameliorative effect was shown during the second season, Ca content in the rice root increased by 33.33% to 90.20% during the second season for all treatments. However, no significant difference was observed among the treatments.

Besides soil acidity and Al toxicity, farmers in the area face additional challenge of drought (Fig. 3a and 3b). Lowland rice is extremely sensitive to water shortages and drought problems when soil water contents drop below saturation (Bouman and Tuoang, 2001) and this scenario will reduce leaf area expansion and result in closure of stomata, leaf rolling, deeper root growth, enhanced leaf senescence, reduced plant height, delayed flowering, and reduced number of tillers, panicle, spikelet, and grain weight (Bouman and Tuoang, 2001).

The paddy field was dry when the seeds were sown during the first season. There was no proper water irrigation and drainage system in place, resulting in dependence solely on rain water, which falls erratically throughout the growing season; hence, crop watering was insufficient. As a result, the sown seeds did not germinate well, and the seedlings suffered because their roots were unable to tap the underground water. Therefore, water was pumped in from the nearest canal to germinate the seeds (Fig. 3b). This had affected the subsequent growth of rice seedlings and hence, the eventual rice yield. However, treatments with GML and hydrated lime raised the pH of water throughout rice planting, with pH ranging from 4 to 6, as shown in Figure 6.

An ameliorative effect was shown during the subsequent season (second season) as water pH was increased drastically between 28 to 64 days after seeding for all treatments. In addition, Al and Fe of water were also alleviated from 4.12 to 0.3 mg L^{-1} and from 0.84 to 0.02 mg L^{-1}, respectively.

Cost-Benefit Analysis of Liming Materials

Based on the farmer common practice, rice yields are often ± 2 t ha^{-1} in the Merbok acid sulfate areas, and this is far below the national average of 3.8 t ha^{-1}. Soil amendment with liming materials (GML, hydrated lime, and liquid lime) are noted to increase rice yield at an average of 30%. Thus, to increase the income of farmers and reduce production costs, a cost-benefit indicator is presented in Table 6. Fertilizer and pesticides used by farmers are subsidized by the Malaysian government to promote agricultural sustainability in the rural areas. Therefore, the cost for liming materials and labor cost to apply the selected liming materials were taken into account.

Table 6 Cost-benefit indicator of selected liming materials. Means marked with the same letter for each season were not significantly different at $p < 0.05$ (Tukey's test).

		Control	4 t ha^{-1} ground magnesium limestone	2 t ha^{-1} hydrated lime	20 L ha^{-1} liquid lime (applied only 1st season)	20 L ha^{-1} liquid lime (applied 1st and 2nd seasons)	2 t ha^{-1} hydrated lime (without fertilizer)
Rice yield (t ha^{-1})	Season 1 (a)	2.71^{ab}	3.71^{a}	3.38^{ab}	1.79^{b}	1.97^{b}	3.26^{ab}
	Season 2 (b)	3.95^{a}	4.34^{a}	4.29^{a}	3.52^{a}	3.68^{a}	3.21^{a}
	Yield increment (%)	+31.39	+14.52	+21.21	+49.15	+46.46	-1.55
	Total yield (a+b)	6.66	8.05	7.67	5.31	5.65	6.47
Cost for liming	Price	-	USD 50 t^{-1} = USD 200	USD 140 t^{-1} = USD 280	USD 97 20L^{-1} = USD 97	USD 97 20L^{-1} = USD 194	USD 140 t^{-1} = USD 280
	Labour	-	USD 45 t^{-1} = USD 180	USD 45 t^{-1} =USD 90	USD 16 ha^{-1} = USD 16	USD 16 ha^{-1} =USD 32	USD 45 t^{-1} =USD 90
	[d]Total		USD 380	USD 370	USD 113	USD 226	USD 370
Price of rice (USD 315/ t)	[c]Total price for yield	USD 2098	USD 2536	USD 2416	USD 1673	USD 1780	USD 2038
Profit	c-d	USD 2098	USD 2156	USD 2046	USD 1560	USD 1554	USD 1668

The net profit was calculated based on rice yield for a year per hectare. Rice yield had increased by 14% to 50% during the second season due to treatments applied along with fertilizers. The rice yield for soil treated with 2 t ha^{-1} HL without fertilizer recorded a minimal decrease of 1.55%. Application with 4 t ha^{-1} GML recorded the highest rice yield for both seasons, with yields of 3.71 t ha-^{1} $season^{-1}$ and 4.34 t ha^{-1} $season^{-1}$ for the first and second seasons, respectively. GML application produced significantly higher yield, however, the total cost was higher compared with other liming materials application. The cost to treat the soil with 4 t ha^{-1} GML was USD 380, providing the highest rice yield with 8.05 t ha^{-1} $year^{-1}$ compared to other treatments. The selling price of rice is USD 315 t^{-1}, and for 8.05 t ha^{-1} $year^{-1}$, this roughly provides a profit of USD 2,156 per hectare per year.

Treatment Without Fertilizer

No significant effects were observed on rice yield and rice yield components between plots treated with 2 t ha^{-1} HL with fertilizer (T3), and without fertilizer application (T6) for both seasons. However, plots treated with 2 t ha^{-1} HL with fertilizer (T3) showed a yield increment during the following season (second season) of 21.21%, whereas plots treated with 2 t ha^{-1} HL without fertilizer showed a small decrease in rice yield of 1.55%. The total yields obtained for T3 and T6 for a year were 7.67 t ha^{-1} $year^{-1}$ and 6.47 t ha^{-1} $year^{-1}$, respectively. The differences between the two season was 1.2 t ha^{-1} of rice yield. From the result, it is evident that the application of fertilizer is not necessary for at least one season (T6) at the respective area. Therefore, farmers could reduce costs by reducing fertilizer application.

Conclusion

Application of ground magnesium limestone (GML) and hydrated lime (HL) increase rice yields for both seasons. In contrast, liquid lime (LL) application showed no signifiant increment of rice yield on acidic sulfate soil as compared to the control for both seasons. Therefore, ground magnesium limestone (GML) is a suitable liming material under acidic sulfate soil conditions of Merbok, Kedah (MALAYSIA). Application of 4 t ha^{-1} $year^{-1}$ increased rice yield by 31.39%, with the highest recorded rice yield was 8.05 t ha^{-1} $year^{-1}$ and, with a calculated profit of USD 2,156 compared to other treatments.

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References

Alia F., Shamshuddin J., Fauziah C., Ahmad Husni M., and Panhwar Q. (2015). Effects of aluminum, iron and/or low pH on rice seedlings grown in solution culture. *Int. J. Agric. Biol.*, Vol. 17, pp. 702–710.

Bray R. and Kurtz L. (1945). Determination of total, organic and available forms of phosphorus in soils. *Soil Sci.*, Vol. 59, pp. 39–45.

Bouman B. and Tuoang T. (2001). Field water management to save water and increase its productivity in irrigated rice. *Agric. Water Management.* Vol. 49, pp. 11–30.

Crooks R. and Prentica P. (2012). "*The importance of silicon weathered soils,*" presented at the 8th International Symposium on Plant Soil Interactions at Low pH, Bengalaru, India, pp. 412–413.

Cate R. and Sukhai A. (1964). A study of aluminum in rice soils. *Soil Sci.*, Vol. 98, pp. 85–93.

Dobermann A. and Fairhurst T. (2000). *Rice: Nutrient Disorders and Nutrient Management. Handbook Series.* Potash and Phosphate Institute (PPI), Phosphate Institute of Canada (PPIC) and International Rice Research Institute (IRRI). Los Banos, Philippines.

Dent D. (1986). *Acid Sulfate Soil: A Baseline for Research and Development*. Wageningen, The Netherlands: ILRI Publication 39.

Elisa A., Shamshuddin J., Che Fauziah I., and Roslan I. (2014). Increasing rice production using different lime source on an acid sulphate soil in Merbok, Malaysia. *Pertanika J. Trop. Agric. Sci.*, Vol. 37, No. 2, pp. 223–247.

Elisa A. A., Shamshuddin J., and Fauziah C. (2011). Root elongation, root surface area and organic acid by rice seedling under Al^{3+} and/or H^{+} stress. *Am. J. Agric. Biol. Sci.*, Vol. 6, No. 3, pp. 324–331.

Godbold D., Fritz E., and Huttermann A. (1988). Aluminum toxicity and forest decline. *Proc Nat Acad Sci*, Vol. 85, pp. 3888–3892.

Hiradate S., Ma J., and Matsumoto H. (2007). Strategies of plants to adapt to mineral stresses in problem soils. *Adv. Agron.*, Vol. 96, pp. 66–132.

Horst W., Rangel A., Eticha D., Ischitani M., and Rao I. (2009). Aluminum toxicity and resistance in Phaseolus vulgaris L.*Physiology Drives Molecular Biology*, pp. 53–54.

Najim M., Lee T., Haque M., and Esham M. (2007). Sustainability of Rice Production: A Malaysian Perspective,*Journal of Agricultural Science, Sabaragamuwa University of Sri Lanka.* Vol. 3(1): pp. 1-12.

Palhares de Melo L., Bertioli D., Cajueiro E., and Bastos R. (2001). Recommendation for fertilizer application for soils via qualitative reasoning. *Agric. Syst.*, Vol. 67, pp. 21–30.

Ridolfi M. and Garrec J. (2000). Consequences of the excess Al and a deficiency in Ca and Mg for stomatal functioning and net carbon assimilation of beech leaves. *Ann. For. Sci*, Vol. 57, No. 3, pp. 209–218.

Shazana M., Shamshuddin J., Fauziah C. I., Panhwar Q., and Naher U. (2014). Effects of applying ground basalt with or without organic fertilizer on the fertility of an acid sulfate soil and growth of rice. *Malays. Soc. Soil Sci.*, Vol. 18, pp. 87–102.

Shamshuddin J. (2006). Acid Sulfate Soils in Malaysia. UPM Press, Serdang, Selangor, Malaysia.

Shamshuddin J., Sharifuddin H., Che Fauziah I., Adwards D., and Bell L. (2010)., Temporal changes in chemical properties of acid soil profiles treated with magnesium limestone and gypsum. *Pertanika J. Trop. Agric. Sci.*, Vol. 33, pp. 277–295.

Shamshuddin J., Sharifuddin H., and Bell L. (1998). Longevity of ground magnesium limestone applied to an Ultisol. *Commun. Soil Sci. Plant Anal.*, Vol. 29, No. 9 &10, pp. 1299–1313.

Shamshuddin J., Elisa A., Shazana M., and I. Che Fauziah (2013). Rice defense mechanism against the presence of excess amount of Al^{3+} and Fe^{2+} in the water. *Aust. J. Crop Sci.*, Vol. 7, No. 3, pp. 314–320.

Susan C., Hue N., and Michael A. (2007). *Handbook of Plant Nutrition*. London, England.

Ting C., Rohani S., Diemont W., and Aminuddin B. (1993). "*The development of an acid sulfate area in former mangroves in Merbok, Kedah, Malaysia*," presented at the Ho Chi Minh City Symposium on Acid Sulfate Soils. pp. 95-101. Int. Inst. for Land Reclamation and Improvement, Wageningen. The Netherlands.

Van Breemen N. and Pons L. (1978). *Acid sulfate soil and rice*, In IRRI (Eds.), Soils and Rice. International Rice Research Institute, Los Banos, Philippines.

Yoshida S. (1981). *Fundamental of rice crop science*. International Rice Research Institute. Los Banos, Philippines.

[Online]. Available: http://www.humibox.com/products/soil-acidity-correctors/liquid-lime. [Accessed: 21-Apr-2016].