

CHAPTER 2

LITERATURE REVIEW

Disposal of wastewater in soil mentle is not new, it has been used for several decades although on a limited basis. In Thailand, septic tank soil adsorption were primary system for treatment of excreta for a long time. During the earth's population continue to increase, more and more people propose to use land as the ultimate sink for man's waste as well as food production. Very little work studies about soil's change and other ecological impact and etc. Some began to studies properties and process in a soil for disposal of waste and used these soil for crop production. In Thailand, most researchers interested in quality of effluent from soil column in vertical direction and movement of pathogenic bacteria (1, 2). In the temperate region, many researchers concentrated about land treatment by using domestic wastewater and soil properties changes. Thus, it could summarize their studies as follows.

2.1 Effects on Physical Properties of Soil by Wastewater

2.1.1 Color

Soil color could be estimated chemical and physical characteristics of soil. However, soil color depend on the quantity of humus, varity of iron and water content in soil texture i.e., red soil color was ferric oxide (Fe_2O_3), yellow soil color was hydrated ferric oxide ($2\text{Fe}_2\text{O}_3 \cdot 3\text{H}_2\text{O}$) and brown soil color was amount of humus. In case of passing wastewater, soil color may be changed as shown by Mc Gauhey and Winneberger (7). They showed

that soil column receiving of sewage water found black layer of soil extending to a depth of 5 to 10 cm. It was assumed that the black color result from ferrous sulfide in particle form or accumulating of organic matter.

2.1.2 Clogging

Physical properties of soil may determine the movement rate of wastewater into and through the soil. Texture and structure may be influenced permeability and hydraulic conductivity of soil. Treating soil with domestic wastewater for a long period of time, may also stimulate clogging of soil pore. Chan et al. (8) and Laak (9) showed that the initial process of soil pore clogging was primarily influenced by physical entrapment of suspended particles in soil pores and followed by a secondary mechanism of microbial growth and biological activity. Whereas chemical clogging seldom occur unless the sodium content of the wastewater is high. Jones and Taylors (10) suggested that clogging in sand is established faster under continuous treating than under intermittent submergence. Particularly, under anaerobic condition clogging would occur 3 to 10 times more faster than under aerobic condition. On the contrary, Mc Gauhey and Winneberger (7) reported that in aerobic system, severe clogging may occur depending on total organic loading in the soil. It means that biomass as degraded product may generate clogging.

Avinimelech and Nevo (11) showed that there were two factors which caused clogging, the C:N ratio of the soil and the decomposition rate of organic matter. Soil with high C:N ratio caused a relatively slower clogging than soil with low C:N ratio, and undecomposed material caused a slightly clogging. Decomposed materials such as polysaccharide, polyuronide and other produced by microorganism could fill up soil pores and reduced permeability of soil (12,13,14,15). In addition, Travis (16) applied feedlot

lagoon water to soil cores and they became clogged. He also proposed that the clogging could be due, in part, to the organic material including cell mass in the lagoon water. Contrary, Allison (12) demonstrated that soil clogging may be resulted from of biochemical processes by organisms within the soil and not a result of filling soil spaces with organisms in added wastewater. A typical nature of biologically clogged soil may appear as a heavy overgrowth of black biological slimes (7). This slime layer is not all bacterial cells but, to a large extent, comprised of ferrous sulfide. However, Mc Calla (13) also suggested that under prolonged submergence the microorganisms in soil may reduce the percolation rate in two ways, one was when they produced gas or highly hydrated organic material such as slimes and another one was that the microorganisms may held in decomposing or changing the structure of stabilizing agents which caused the deterioration of soil structure. Although a wide variety of factors can influence soil clogging development. Siegrist (17) suggested that polysaccharide materials were not responsible for the soil clogging. After 62 months treatment of grey water of septic tank effluent, he found that decreasing concentration of polysaccharide corresponded with increasing effluent loading and clogging severity.

A comparison of bottom, sidewell and top surfaced infiltration rates was made experimentally by McGauhey and Winneberge (7) in Oakley sand and finer Hanford sandy loam. Permeability was greatest into the sidewells because both gravity and infiltration act to deposit solids on the horizontal face of the soil, whereas gravity tends to strip the vertical face of settleable matter or sedimentation in the direction of flow of infiltrating water occurs in the bottom. Interestingly, Viraraghavan and Warnock (18) concluded that the horizontal permeability was found to be about 60 times greater than the vertical permeability. In field study, Bouma et al. (19) showed

that crust formation and clogged layers at the bed-soil interface was confined to top 2 cm of soil filter. However, flow of effluent through this layer was impeded but not stopped.

2.2 Effects on Chemical Properties of Soil by Wastewater

2.2.1 pH

There are many compounds in soil that influence the concentration of H^+ in a soil solution. The situation is complicated by the great diversity of soil materials and their interaction. This property is also affected by passing the aqueous solution through soil for a period of time. For example, sanitary landfill leachate (pH 10, ionic strength = 0.1M) passed through a Yellow earth covering material (pH 4.2) in leaching column over 29 days, to cause the soil pH increase. When the pH of the soil increased above 6 strong sorption of Ca and Mg took place (20).

Change in soil pH is the result of increasing adsorbed H^+ and the content of adsorbed base. When soil colloids are dominated by Ca and Mg on their adsorptive surfaces the soil is neutral or alkaline in reaction, but in soil colloid which is high of H^+ , Al, the soil is acidic (21). Minor fluctuations in soil pH is due to microbial action and distribution of organic residues in the soil (22).

Beek and Hann (23) showed that the pH of the soil treated with domestic wastewater was higher than that of the untreated soil, because soil received bases from the wastewater. Sometimes soil with good permeability could not remove bases from the influent, therefore, they exhibited alkaline reaction (1). Contrary, the reduction of soil pH observed in sludge-amended soil that the initial nitrification of applied NH_4-N and acid produced during the decomposition of organic matter (24).

2.2.2 Plant Nutrient

2.2.2.1 Nitrogen Content

One of the major plant nutrient in domestic wastewater is nitrogen compound, i.e., $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$. They were retained and accumulated in soil. Those processes depended on the soil types and the species of nitrogen compound. Lance (25) and Thompson and Trough (26) showed that $\text{NH}_4\text{-N}$ was adsorbed by clay and organic matter cation exchange sites and /or fixation in soil containing vermiculite mineral. On the other hand, Walker (27) found that the higher capacity of $\text{NH}_4\text{-N}$ adsorption in fine texture soil was higher than coarse texture one; this incidence dued to their largest available surface area per volume. In the same soil texture, ammonium adsorption capacity is limited by the types of clay minerals and amount of humus in soil colloid. The exchange capacity of Cecil clay which dominance of 1:1 type of clays mineral is 4.8 meq/100g dry soil while Susquehanna clay which dominance of 2:1 type of clays mineral is 34.2 meq/100g dry soil. Regarding clay mineral type, fixed $\text{NH}_4\text{-N}$ content follows the order vermiculate > illite > montmorillonite > kaolinite (28). In addition ammonium fixing capacity of the soil depends to some extent on K content and contact time (29). The ammonium adsorption of Zimmerman sand and Hayden silt increased with time of feeding until the complete equilibrium were met. Zimmerman sand needed 2 hours for 90% of equilibrium and 4 hours for complete equilibrium while Hayden silt needed 6 hours for complete equilibrium. Furthermore, ammonium adsorption was decreased when potassium ions present in large amount. The adsorption of Zimmermann sand over 10 times of the feeding sorption has 1050 mg/g of K. Moreover, increasing concentration of $\text{NH}_4\text{-N}$, drying conditions and high in soil pH (such as affect through liming) tends to increase $\text{NH}_4\text{-N}$ fixation (30). Nevertheless, the ability

of the soil to adsorb ammonium is limited; only about 5% of CEC is available for such adsorption (25). For soil with CEC, 15 meq/100g, this results in an ammonium adsorption capacity of 1 g/ft² of soil. Clearly, this capacity could be saturated in a matter of months, even at low ammonium application rates. However, some NH₄-N that retained in soil colloids should be transformed into nitrate ion or nitrification. This process is favored by good aeration of soil, 80-90°F and a neutral to highly alkaline (31). The effect of pH is probably indirect, resulting from an association with the availability of exchangeable bases such as K, Ca, Mg. Frederick (32) also suggested that in similar environment, the lag periods of nitrification in the soil with the low numbers of nitrifiers were long. This information agreed with Preul and Schropffer (29) who indicated that the addition of the nitrifier population, either by inoculation or preincubation, caused a reduction of lag periods but did not greatly influence the magnitude of the maximum water. Lance and Whisler (33) suggested that wetting-drying cycles, e.g., 5 days each, increased aerobic condition and rejuvenate the exchange complex in terms of ammonium retention. During the wetting cycle, NH₄-N was retained in the surface soil by cation exchange reactions and NH₄-N was then nitrified during dry periods and flushed into the groundwater.

Under anaerobic condition, NO₃-N converted into gaseous form, (N₂O or N₂) by chemodenitrification or bidenitrification. Particularly, chemodenitrification restricted to occur in highly acid soil whereas bidenitrification required organic carbon for denitrifying bacteria. Approximately 0.9-1.3 grams of biodegradable organic carbon are needed to denitrified one gram of NO₃-N. (34). For these reasons, denitrification is extremely difficult to occur in soil-absorption system.

2.2.2.2 Phosphorus Content

Orthophosphate in domestic wastewater has been identified as a major element controlling eutrophication of surface water. In general, a large amount of P in wastewater is readily sorbed by soil its movement is restricted to the upper 15 cm layers (35, 36, 37). Removal of P by the soil column over the extent period of time involves both sorption of P on minerals surfaces corresponding to the initial fast reaction and its precipitation in the subsequent slow reaction (38). The slow reaction in these acid soils ($\text{pH} < 5$) may involve precipitation of Al and Fe phosphates while in alkaline soils ($\text{pH} > 6.5$), it was precipitate by Ca (23). In addition, a combination of Al, P and cation in the wastewater was formed a stable compound for instance, Al-P-Ca complex form (39). In the other hand, Ibrahim and Pratt (40) suggested that the other slow reaction are possible that P penetrates into porous solids becoming adsorbed and leaving surface sites to adsorb more P, thus less soluble or it is converse directly into fixed form (41). Hemwell (42) proposed that phosphorus was fixed by aluminium ions which originated from the exchange sites or from lattice dissociation of the clay minerals to form variscite ($\text{AlPO}_4 \cdot 2\text{H}_2\text{O}$), a highly insoluble aluminium phosphate. The order of increasing phosphorus sorption by soils of various mineralogical systems is : quartz, organic materials < 2:1 clays < 1:1 clays < crystalline oxides < desilicated < volcanic ash.

After P treatment in soil for a period of time, P can release as seen by P founding from septic tank drainfield which affected surface water quality (43). Further, Sawhney (38) and Tofflemire and Chan (44) have attempted to describe P movement through soil based on their laboratory determined P sorption characteristics. They observed that (i) all added P was sorbed by the soil initially, (ii) breakthrough of P occurred after certain

amount of P equivalent to that sorbed by the soil after 200 hours of reaction time in an isotherm test was sorbed by the columns (38), and (iii) after breakthrough, leachate P concentration increased steadily with time and reached the same value as that of the effluent (44). On the other hand, Magdoff et.al. (45) showed that wastewater containing about 8×10^{-4} MP was applied at 8 cm/day to a sand fill, the P concentration of the effluent was very slow initially. However, after 70 days, the P concentration of the effluent solution increased to as much as 60% of the influent concentration. In addition, Lance (46) reported that the drainage water from soil columns reached an equilibrium P concentration in response to secondary effluent leaching. Furthermore, Nagpal, (47) concluded that the amount of P sorbed by the soil columns exceeded that predicted by the P isotherms and was depend on effluent percolation rate and leaching periods. Because both precipitation and regeneration of P sorption sites were considered responsible for equilibrium P conditions (47). Moreover, Van Riemsdijk et.al (39) and Stuanes (48) indicated that P was sorbed from wastewater than from pure solution. This can be due to components of the wastewater or different experimental conditions.

Some conditions for regeneration sorption capacity are alternate wetting and drying cycle (49). The exact mechanism of the regenerations is not known. However, other observations in the field laboratory support to the regeneration concept. For example, treatment of undisturbed soil cores with simulated wastewater for 2 years and determination of P sorption by soil after used-up to 15 years. These show that P moved very slowly in the soil from the point of supply and soil surrounding their oldest drainfield were not complete saturated with P retained a sorption of this sorption capacity. Similarly, Kao and Blanchar (35) observed that phosphorus sorption capacity of a soil after phosphorus fertilization for 82 years was the same as the unfertilized soil. Another factor strongly influence on P sorption

behavior are oxidized and reduced condition. Khalid et.al. (50) showed that Crowley silt loam can sorbed more phosphorus under oxidized condition than reduced condition. Likewise, Sikora and Corey (51), and Hill and Sawhney (37) reported that saturated soil caused by heavy passing of domestic wastewater induced the soil to become anaerobic condition. Under this condition, the solubility of fixed P was higher and enhanced the mobility of P.

2.2.3 Cation Content (Na, K, Ca, Mg)

Upon contacting the soil four general cations Ca, Mg, Na and K in domestic wastewater undergo chemical reaction. Fred (52) suggested that adsorption process and/or ion exchange could reduce the mobility of these cations and the soil served as an effective filter of them for a long period of time. The added Na can replaced Ca, despite of its greater intensity of attraction. In fact, the system is in dynamic equilibrium so that the retention of dissolved cations largely depends on their concentration in the solution entering the soil (53). In addition, Amoozegon-Frad (54) indicated that Na and K are released from manure aggregates during the early water applications, replacing Ca and Mg on the soil exchange sites. Na and K moved out of the manure aggregates into solution to be adsorbed by the soil particles or leached to lower layers whereas Ca, Mg was replaced by Na, K on the soil exchange sites. However, soil can retain some soluble salts. This result of replacement Ca and Mg by monovalent cations (Na and K) in swelling with concurrent dispersion and reduction in percolation rate (16). Schuman and Mc Calla (55) showed that high exchangeable K in the feedlot profile was present in high levels and also high in carbon. This zone is very dark and the permeability of the dark layer is low. Therefore, the high exchangeable K in the top few centimeters of the feedlot profile could cause some dispersion and contribute to

the poor physical properties and low infiltration rate of the soil. Another factor affected the movement of these cation is pH. Chan et al. (20) showed that sanitary landfill leachate (pH 10, ionic strength $\sim 0.1M$) passed through a Yellow earth, covering material (pH 4.2) in leaching columns over 29 days. The cations in the leachate were retained by the soil in the order of $Mg > Ca > K > Na$. Enhanced adsorption of Mg and Ca started to occur about pH 6. In addition, De Jong (56) found that adsorption of $Mg > Ca$. It appears that the Fe and Al oxide coatings and particles in the soil could be the site of specific adsorption of Mg. Moreover, Chan et al. (20) concluded that Mg was becoming more strongly bound to the soil as the pH increased and specific adsorption was taking place.

Normally, soil K ion can be fixed as same as NH_4-N whereas other cation such Ca and Na have different ionic radii. Some factors affect the form of K equilibria in soil, for example, the freezing and thawing of moist soil may also be important in the release of fixed K and the fixation of the exchangeable form. And Mg may become fixed in the lattice structure of certain clay minerals under some soil conditions, although Ca does not exhibit such behavior (57). In contrast, Ca may be retained in soil by the formation of insoluble form such as $CaCO_3$, $CaSO_4$ and $CaHPO_4$ and highly insoluble form with phosphate (28). Leaching with the sewage effluent not only increased CEC but also the sum of exchangeable cations ($Na+K+Ca+Mg$) presumably due to displacement of adsorb H and/or Al. However, cation exchange capacity (CEC) of soil would increase by adsorption as shown by Sawhney (38) suggested that in soil at low pH (pH < 5.0) values, the increase in CEC produced by P sorption occurs mainly by replacement of octahedrally coordinate water molecules on mineral surfaces. At higher pH values (pH > 6.5) increases in CEC may be largely due to replacement of hydroxy ions by polyvalent phosphate ions. The above results same as reported

by Jue and Maduakor(58) that applied phosphate fertilized to soil, therefore soil sorb highly phosphate due to increasing CEC of the soil. Because of more negative charges on the surface of the soil particles.

2.2.4 Anion Content (SO₄ and Cl)

The retention of anions by soil was found many years ago. Tisdale and Nelson (57) and Barrow (59) suggested that the capacity for retaining anions increased with a decrease soil pH. Further, anion exchange is much greater in soil that are high in 1:1 clays and containing hydrous oxides of iron and aluminium, than it is in soil with predominately 2:1 clay. In strongly acid soil, where the concentration of hydroxyl ions is extremely low, Cl⁻ and SO₄ are assumed to be competition with the hydroxyl ions found in the hydrous oxide presented in the soil. Therefore, at low pH values and in soil with large amounts of hydrated Fe and Al oxides, the amounts of such anions as SO₄ that are retained may be quite high. And sulfate sorption is relatively more affected by an increasing in pH.

In addition, Johnson and Cole (60) concluded that anion adsorption (primary to amorphous sesquioxide surfaces) is a predominant mechanism for sulfate mobility. Their experiments were set by applications of successive aliquots of 10⁻³, 10⁻², and 10⁻¹N H₂SO₄ followed by distilled water to the second-growth Douglas-Fir ecosystem. The results also showed that total sulfate adsorption depended upon proportional of the sesquioxide content in the soil and capacity of the soil to adsorb sulfate consists of at least two mechanisms which are implied by the shapes of the isotherms. Moreover, Kamprath et al.(61) indicated that sulfate and H₂PO₄⁻ competed for the surface sites in soil also. First, OH⁻ displacement on sesquioxide surfaces resulting in exchangeable sulfate, and the second, -O-displacement within sesquioxide

structure, or anion penetration resulting in strongly bound, less available sulfate. Taylor et.al. (62) composted sewage sludge in Evesboro loamy sand, Christiana silty clay loam, and Fanquier silt and a sand at rates of 0, 2, 4, and 6% of the dry weight, respectively. The SO_4 adsorption capacity of Evesboro was lowest. The adsorption capacity of Fanquier silt loam was similar to the Christiana as a result of its high Fe oxide. In contrast, the amount of phosphate from the Christiana, the extractable sulfate increased after 14 days for the 4 and 6% amendments due to possible competition between sulfate and phosphate for adsorption sites. Contrary, there is a large electrostatic component to the bond between sulfate and the surface, with little sorption occurring in the absence of positive charge. While P can be sorbed on neutral or even negative surfaces (63). Parfitt and Smart (64) also suggested that sulfate sorption is not entirely nonspecific. Even in the presence of high concentrations of Cl (0.025 M CaCl_2 or 0.1 M NaCl), sulfate appeared to be preferentially sorbed on positive sites in soil suggesting that there was a strong selectivity for SO_4 over nonspecifically adsorb anions, such as Cl. Moreover, Marsh et. al. (65) suggested that sulfate is sorbed in calcium media higher than in sodium media because of the larger compression of the double layer and reduced anion exclusion. Closer approach to the surface could be facilitated, not only by the presence of Ca in the medium but also by high ionic strength or high concentration of SO_4 (66).

Hingston et.al.(67) suggested that chloride was adsorbed on sesquioxide surfaces because there was much less inorganic chemical retention in soil than in the case of sulfate. Therefore, Cl is sorbed by relatively weak electrostatic forces. Johnson et.al.(68) applied municipal wastewater in Everest gravelly loamy sand soil (Typic Haslorthod) with vegetative cover. The result showed that concentration of chloride and sulfate in soil solution at depths of 60 cm. was 14.8 ± 3.9 and 8.5 ± 2.1

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meq/hax 10^3 , while municipal wastewater input was 13.8 ± 1.8 and 11.9 ± 1.6 meq/hax 10^3 , respectively.

2.3 Effects on Biological Properties of Soil by Wastewater

2.3.1 Fecal Coliform Content

The most widely accepted bacterial indicator for fecal pollution in natural water is the coliform group (69). E. Coli, or fecal coliform, is used as biological indicators. Generally, these organisms enrich in domestic wastewater. Then passing through the soil, the number of them decrease sharply due to die-off and unsuitably environmental conditions, such as, pH, moisture, antibiotics, toxic substances, competitive organisms, available nutrients, and soil types (70). Another factor which controls the availability of them in the soil adsorption or retention is soil filtration (71). Other factors that increasing retention of E. coli are increasing in clay content, CEC of the soil, soil pH < 7 and specific surface area (72). In addition, intact and disturbed soil affected on retention of bacteria (73). The mean value of E.coli from effluent in intact soil of Crider silt loam, Maury silt loam and Bruno sandy loam was 0.44, 0.22 and 0.79 col/ml, respectively, whereas the ratio of the disturbed soil were 0.07, 0.002, and 0.05 col/ml respectively. In field study, Ziebell et al. (74) reported that soil samples collected 30 cm lateral to 8 cm and 38 cm below the trench produced coliform levels 100 fold, 3000 fold less than the septic tank effluent. In contrast, E.coli can move with infiltrating water or desorption from the soil particles and transport to other place as shown by Bouwer et. al. (75) indicated that fecal coliforms at original concentrations of 10^6 cells/100 ml of sewage effluent were removed in the first 60 cm of soil depth. Avnimelech et al. (11) reported that 99% of the coliforms were removed by movement

of secondary effluent through 3 m of sand. Lance et. al. (76) found that reduction of fecal coliforms by filtration through 250 cm long columns filled with loamy sand as proportional to the concentration of fecal coliform applied at the soil surface. The columns reduced fecal coliform concentration by almost 3 fold during 9-day flooding periods at infiltration rates of 40-50 cm.



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