2. REVIEW OF LITERATURE

Intensive examination of our environment associated with microorganisms, largely implicated in soil fertility, and their interactions with different chemicals that assume the status of pollutants provide insights into some of the ways that the pollutants alter growth and development of organisms, the response of living organisms to different pollutants themselves, and their adverse impacts on the environment itself (Ramakrishnan et al., 2010; 2011). Investigations to assess the effects of environmental pollutants include studies on nontarget influence of toxicants toward ecologically beneficial microorganisms, and those that deal with the impact of soil microorganism on the persistence of the toxic chemicals. Following is the literature that pertains, in particular, to the effect of insecticides on soil enzyme activities, an important parameter which helps in maintaining the soil health and fertility.

2.1. Nontarget effects of insecticides on soil enzyme activities

Several enzymes are known to be present in the soil which catalyze organic matter turnover. These enzymes are produced by various organisms, and are mainly of bacterial and fungal origin, and act intra- or extra-cellularly. Only a small fraction is derived from plants and/or animals. Soil enzymes play a role in the degradation of litter and "foreign" substances. The role of soil enzymes, in terms of the ecosystem, is increasingly important and is defined by the relationships between soil enzymes and the environmental factors affecting their activities (Paul and McLaren, 1975; Burns, 1982). More importantly, the enzymes most often present in soil are dehydrogenase, catalase, phosphatases, amylase, cellulases, xylanase, pectinase, saccharase, proteases and urease. Although much soil enzyme research has evolved without thought to the ecological implication, soil enzymes are useful in describing and making predictions about an ecosystem's function, quality and the interactions among subsystems. Perhaps the most valuable single use of soil enzymes is to assess the effects of various inputs on the relative "health" of the soil. Numerous studies have been conducted to
determine changes in a soil's enzyme activities caused by acid rain, heavy metals, pesticides, and other industrial and agricultural chemicals. Nonetheless, soil enzyme activities commonly correlate with microbial parameters (Frankenberger and Dick, 1983) and have been shown to be sensitive indices of long-term pesticide effects. In this direction, various reports have indicated that under field and/or laboratory conditions, insecticides applied at commercially recommended rates, exerted an adverse effect on microbiological properties of soil as manifested by the observed altered enzymatic activities.

According to a more recent study, the treatment of soil with 50 and 100 μg g⁻¹ of methyl parathion showed a clear initial reduction in bacterial population count, but there was a gradual increase in bacterial count on the 35th day of incubation in all the three concentrations (Bindhya et al., 2009). Singh et al. (2002) reported that three measured soil microbial parameters (enzyme activities and total microbial biomass) were stable in the pesticide free control soils throughout the 90-d incubation period, but they were all adversely affected by the presence of chlorothalonil in soil; whilst effects from fenamiphos or chlorpyrifos on the soil microbial characteristics were either very small or insignificant. Michael and Turgeona (1978) suggested that the rates of glucose utilization, nitrification of ammonium, amylase synthesis were significantly lower in soil underlying treated turf than in control soil. Surprisingly, although short-lived inhibitory effects on activities of microbes and enzymes were caused by the insecticides, the soil indigenous microbes can tolerate the chemicals used for control of soil pests (Tu, 1995). Cernakova et al. (1992) found that both bacterial growth and the activities under study were negatively influenced by high concentrations of actellic whereas lower concentrations stimulated the overall metabolic soil activity. Similarly, the electron transport system/dehydrogenase activity
displayed a negative correlation with triazophos, bensulfuron-methyl, chlobenthiazone concentration of pesticide increased (Xie et al., 2004).

An intriguing recent discovery is that the impact of long-term DDT pollution in soil by using different criteria (Megharaj et al., 2000). The criteria used included chemical analysis of DDT residues, microbial biomass, and dehydrogenase activity, viable counts of bacteria and fungi, and density and diversity of algae. The experimental results indicated that the viable counts of microalgae and bacteria decreased with increasing DDT contamination, while fungi, microbial biomass, and dehydrogenase activity increased in the medium-level contaminated soil (27 mg DDT residues kg$^{-1}$ soil). All of the tested parameters were greatly inhibited in the high-level contaminated soil (34 mg DDT residues kg$^{-1}$ soil). More recently, the effect of fenamiphos, a widely used OP pesticide, on important soil microbial activities such as dehydrogenase, urease and potential nitrification in four soils from Australia and Ecuador were studied by Caceres et al. (2009). The results showed that fenamiphos in general was not toxic to dehydrogenase and urease up to 100 mg kg$^{-1}$ soil. However, potential nitrification was found to be highly sensitive to fenamiphos with a significant inhibition recorded even at 10 mg kg$^{-1}$. In general, nitrification activity in soils was decreased with an increase in fenamiphos concentration. In contrast, dimethoate and malathion added to soil at 10 and 100 µg g$^{-1}$ caused an initial stimulation of CO$_2$ production, and there was an increase in total counts of bacterial propagules. Application of all insecticides increased bacteria producing phosphatases from the first week until week 4 after the application; bacteria then returned to the original levels (Congregado et al., 1979). There is an evidence that the repeated applications of some herbicides (e.g., atrazine, 2,4-D, paraquat, and trifluralin) over many years may compound a negative impact, change microbial community structure, or build-up
biodegradation capacity (Pankhurst, 2006). As such, there is no information available on the interaction between the selected insecticides, viz., buprofezin and acephate, and microorganisms that are implicated in the transformation of carbon, nitrogen and phosphorous in soil.

2.1.1. Cellulases

Cellulose is the most abundant organic compound in the biosphere, comprising almost 50% of the biomass synthesized by photosynthetic fixation of CO$_2$ (Eriksson et al., 1990). Growth and survival of microorganisms important in most agricultural soils depends on the carbon source contained in the cellulose occurring in the soils (Deng and Tabatabai, 1994). However, for carbon to be released as an energy source for use by the microorganisms, cellulose in plant debris has to be degraded to glucose, cellobiose and high molecular weight oligosaccharides by cellulase enzymes (White, 1982). It is apparent that cellulases are a group of enzymes that catalyze the degradation of cellulose, a polysaccharide built of $\beta$-1,4 linked glucose units (Deng and Tabatabai, 1994). This group consists of endo-$\beta$-1,4-glucanase, exo-$\beta$-1,4-glucanase, and $\beta$-$\alpha$-glucosidase. It has been reported that cellulases in soils are derived mainly from plant debris incorporated into the soil, and that a limited amount may also originate from fungi and bacteria in soils (Richmond, 1991). The activity of cellulase was indicated by the degradation of substrate like cellulose polymer of cellophane (Markus, 1955), cellulose powder (Rawald et al., 1968), carboxymethyl cellulose (Kong and Domergues, 1972), and its activity was measured by Pancholy and Rice (1973) through appearance of reducing sugars measured spectrophotometrically. Nevertheless, the cellulase activity was potentially correlated with fungal and bacterial population in soil (Joshi et al., 1993). Furthermore, studies have shown that activities of cellulases in agricultural soils are affected by several factors. These include
temperature, soil pH, water and oxygen contents, the chemical structure of organic matter and its location in the soil horizon (Rubidge, 1977), quality of organic matter/plant debris and soil mineral elements (Burns, 1978), trace elements from fungicides (Atlas et al., 1978). However, introduction of anthropogenic chemicals (pesticides) into soil may have lasting effects on soil cellulase activities and thus soil health.

The organophosphorus insecticide Selecron at 10 and 50 ppm significantly decreased respiration, mycelial protein, extracellular protein and mycelial dry weight of Aspergillus fumigatus, A. terreus and Myceliphthora thermophila when grown at 45 °C (Omar et al., 1993). C_x and C_1 cellulases of tested fungi were significantly decreased. However, C_1 cellulase of A. fumigatus was slightly increased. Ross and Speir (1985) suggested that concentrations of 930 mg kg\(^{-1}\) fenamiphos had a deleterious effect on cellulase activity, which was reduced by 24% after 62 days treatment under laboratory conditions. Lodhi et al. (2000) reported that cellulase activity was not much affected, although an increase of 18.5% was observed at 1.6 µg g\(^{-1}\) soil of baythroid. At the highest level of baythroid, however, cellulase activity was reduced by 25.9%. In contrast, monocrotophos, quinalphos, and cypermethrin (Gundi et al., 2007), fungicides such as tridemorph, captan (Srinivasulu and Rangaswamy, 2006) significantly enhanced the soil cellulase activity at lower doses, whereas, higher rates of the pesticides were either innocuous or toxic to the enzyme activities. The insecticide, cartap hydrochloride, did not affect soil cellulase activity (Endo et al., 1982).

2.1.2. Amylase

Amylase is a starch hydrolyzing enzyme (Ross, 1976). It is known to be constituted by α-amylase and β-amylase (Pazur, 1965). Studies have shown that α-amylases are
synthesized by plants, animals and microorganisms, whereas, β-amylase is mainly synthesized by plants (Thoma et al., 1971). It has long been known that amylase is widely distributed in plants and soils, and plays a significant role in the breakdown of starch. Research evidence suggests that several other enzymes are involved in the hydrolysis of starch, but the major ones include α-amylase which converts starch like substrates to glucose and/or oligosaccharides and β-amylase that converts starch to maltose (Thoma et al., 1971). Studies have, however, indicated that the roles and activities of α-amylase and β-amylase enzymes may be influenced by different factors ranging from cultural practices, type of vegetation, environment, and soil types (Ross and Roberts, 1968).

The changes of soil amylase activity in response to simultaneous and sequential applications of several pesticides were studied in laboratory experiments. Lodhi et al. (2000) observed an increase in amylase activity by a maximum of 91.5% at 1.6 μg g⁻¹ of baythroid. At higher concentration of 6.4 μg g⁻¹ soil of baythroid, however, the activity was reduced by 47.9%. For instance, metsulfuron-methyl at 5 μg g⁻¹ caused a reduction in amylase activity (Ismail et al., 1998). Tu (1995) found inhibitory effect of imidacloprid on amylase activity after 1 wk while significant recovery was observed after 3 wks. Similarly, amylase activity was reduced when soil treated with dimethoate (Mandic et al., 1997) or chlorothalonil (Singh et al., 2002). In contrast, individual application of monocrotophos, quinalphos and cypermethrin greatly enhanced the activities of amylase in soil (Gundi et al., 2007).
2.1.3. Invertase

Invertase catalyzes the hydrolysis of sucrose to glucose and fructose due to β-fructofuranosides, and is predominantly available in microorganisms, animals and plants (Kiss and Peterfi, 1959). This enzyme brings out all hydrolysis of sucrose under either acid or alkaline conditions (Spalding, 1979). Very little information is available on invertase activity in soil polluted with pesticides. A study was conducted with chlorothalonil (Yu et al., 2006) to evaluate its effects on invertase activity in soil after repeated applications. After the first addition, activity was significantly reduced, but marked inhibition was observed after second application. The most important impact of these findings was the transient negative effects that became weaker following the third and fourth treatments. Similarly, soil treated with monosulfuron (Yong-hong and Yu-bao, 2005), carbaryl and atrazine (Gianfreda and Sannino, 1993) resulted in an obvious inhibition of invertase activity in soil. Surprisingly, soil invertase activity was not sensitive to omethoate (Xiang et al., 2009), other insecticides such as pyrethirns and Neemix-4E (Antonions, 2003). In contrast, Lodhi et al. (2000) reported that invertase activity increased by 110.9% at 1.6 μg baythroid g⁻¹ soil followed by a decrease of 40.3% at the highest level tested (6.4 μg g⁻¹ soil). Similarly, glyphosate and paraquat (Gianfreda and Sannino, 1993; Sannino and Gianfreda, 2001) increased soil invertase activity.

2.1.4. Proteases

Proteases are widely distributed among soils and show a wide range of activities (Ladd and Butler, 1972). These enzymes are involved in the initial hydrolysis of protein components of organic nitrogen to simple amino acids. Proteases in soils hydrolyze not only added protein, but also native soil proteins (Kiss et al., 1975). More importantly, hydrolytic degradation of proteins is an important step in the nitrogen
cycle. However, activity of proteases in soil is known to be affected by several biotic and abiotic factors. In agricultural soils, applications of various pesticides cause significant changes in the activities of soil proteases. Two organophosphorus insecticides, monocrotophos, quinalphos and two synthetic pyrethroids, cypermethrin and fenvalerate (Rangaswamy et al., 1994) influenced soil protease activity in a dose-dependent fashion. Activities were increased with increasing concentrations of the insecticides up to 2.5 kg h$^{-1}$ and later declined. Similarly, soil treated with metsulfuron-methyl (Ismail et al., 1998) has shown short-lived inhibitory effect on protease activity in soil. Conversely, decreased protease activity was observed in soils treated with herbicides (Pahwa and Bajaj, 1999), insecticides (Omar and Abd-Alla, 2000) and chlorothalonil (Singh et al., 2002).

2.1.5. Urease

Urease enzyme is responsible for the hydrolysis of urea fertilizer applied to the soil into NH$_3$ and CO$_2$ with the concomitant rise in soil pH (Andrews et al., 1989). This, in turn, results in a rapid N loss to the atmosphere through NH$_3$ volatilization (Fillery et al., 1984). Due to this role, urease activities in soils have received lot of attention. Since it was first reported by Rotini (1935), a process considered vital in the regulation of N supply to plants after urea fertilization. As one might expect, soil urease originates mainly from plants (Polacco, 1977) and microorganisms, and released as both intra- and extra-cellular enzymes (Mulvaney and Bremner, 1981).

It is well documented that urease activity in soils is influenced by many factors which include cropping history, organic amendments, heavy metals, and environmental factors such as temperature (Tabatabai, 1977). Nevertheless, the agricultural chemicals especially pesticides are perhaps the largest groups of poisonous substances being disseminated throughout the environment. Available
reports indicate that some pesticides increased enzyme activities in soil and others decreased them. Lethbridge and Burns (1976) observed urease activity in sand clay loam soil treated with three organophosphorus insecticides, viz., malathion, accothon and thimet. Inhibition of urea hydrolysis, 60 days after application of 1000 μg g\(^{-1}\) insecticide to a sandy clay loam, approached to 40% for accothon and 50% in the case of malathion and thimet. Similar inhibitory effects were recorded using a silt loam soil with which 200 μg g\(^{-1}\) applications also produced inhibition ranging from 14% (accothon) to 23% (thimet) after 10 days. With lower concentrations of insecticide (50 μg g\(^{-1}\)) the inhibition, though again significant, was of a more ephemeral nature. All three insecticides, at a concentration of 1000 μg g\(^{-1}\) prevented almost any hydrolysis of urea by jack bean urease. Effects of metsulfuron-methyl on the activity of urease in loamy sand and clay loam soil was evaluated for up to 28 days by Ismail et al. (1988). Metsulfuron-methyl at 5 μg g\(^{-1}\) caused a reduction in urease activity for the entire period of study. Kalam et al. (2004) suggested that soil urease activity was affected markedly in presence of profenofos and was 62% at 1000 mg kg\(^{-1}\) level after 80 days. Similarly, soils treated with diazinon (Ingram et al., 2005), acetamiprid (Singh and Kumar, 2008), and omethoate (Xiang et al., 2009) could significantly inhibit soil urease activity. Soil treated with chlorothalonil (Yu et al., 2006), amitraz, tebufirinphos and aztec (Tu, 1995) and dimethomorph (Wu et al., 2010) showed short period inhibitory effects. By contrast, in soils treated with baythroid (Lodhi et al., 2000), glyphosate and paraquat (Sannino and Gianfreda, 2001), chlorimuron-ethyl and furandand (Yang et al., 2006) the rates of urease activities were significantly higher.

2.1.6. Acid phosphatase

The term phosphatase in soil is used to describe a group of enzymes that are responsible for the hydrolytic cleavage of a variety of ester-phosphate bonds of
organic phosphates and anhydrides of orthophosphoric acid ($H_3PO_4$) into organic phosphate. Phosphatases are thus a broad group of enzymes that are capable of catalyzing hydrolysis of esters and anhydrides of phosphoric acid (Schmidt and Laskowski, 1961). In soil ecosystems, these enzymes are believed to play a critical role in P cycles (Speir and Ross, 1975) as evidence shows that they are correlated to P stress and plant growth. Apart from being good indicators of soil fertility, phosphatase enzymes play key roles in soil system (Dick and Tabatabai, 1992). With their predominant occurrence in bacteria to mammals, phosphatases indicate their importance in fundamental biochemical process (Posen, 1967). Acid and alkaline phosphatases are exoenzymes and may be protected from degradation by adsorption to clays or to humic substances (Skujins, 1976). Because of their significance in soil fertility, the changes of phosphatase activity in response to simultaneous and sequential applications of several pesticides were studied in laboratory experiments. One study traced a general inhibitory effect (from 5 to 98%) for phosphatase in the presence of glyphosate (Sannino and Gianfreda, 2001). Similarly, propiconazole, profenofos and pretilachlor (Kalam et al., 2004), acetamiprid (Yao et al., 2006) had a strong negative influence on phosphatase activity in soil. Soils treated with chlorothalonil (Yu et al., 2006), insecticides such as cyfluthrin, imidaclorpid, tebupirimphos, aztec and amitraz (Tu, 1995) have shown short-term inhibitory effects.

Pozo et al. (1995) suggested that activities of acid phosphatase significantly decreased initially at concentrations of 2.0 to 10.0 kg ha$^{-1}$ chloropyrifos, but recovered after 14 d to levels similar to those in control soil without chloropyrifos. Madhuri and Rangaswamy (2002) found that soil samples receiving 2.5 kg ha$^{-1}$ of the insecticides dichlorovos, phorate and methomyl and those soil samples receiving 5.0 kg ha$^{-1}$ of chloropyrifos and methyl parathion, the activity of phosphatase was significantly more
at 20 days period of incubation and decreased progressively with incubation. In contrast, Sikora et al. (1990) reported that over 40% of the insecticide-treated soils had higher acid phosphatase activity than the fence row soils which had no previous exposure to the insecticide. Over 2/3rd of the soils treated with fonofos had higher acid phosphatase and phosphotriesterase activity than the fence row soils. Similarly, brominal (herbicide) and insecticide selecron (Omar and Abdel-Sater, 2001) significantly increased the phosphatase activity in soil.

2.1.7. Alkaline phosphatase

Both acid and alkaline phosphatases particularly hydrolyze the ester bonds binding to P to C (C-O-P ester bonds) in organic matter. During the process, inorganic P is released from organically-bound P such as leaf litter, dead root systems, and other organic debris without crucial role in the phosphorous acquisition of plants and microorganisms, and thus in the cycling of it with in the soil (Schneider et al., 2001). However, as described earlier for other enzymes, these enzymes are also highly sensitive to soil treatments during agricultural management practices such as pesticide addition.

Many studies have reported changes in alkaline phosphatase activities in soil upon pesticide treatments. A short-term inhibitory effect of alkaline phosphatase activity was noticed in soil treated with chlorothalonil (Yu et al., 2006). Similar observation was reported by Tu (1995) in soil treated with 5 different insecticides. On the other hand, phosphatase activities were adversely affected when soils were treated with glyphosate (Sannino and Gianfreda, 2001), propiconazole (Kalam et al., 2004) and acetamiprid (Yao et al., 2006). Surprisingly, alkaline soils treated with brominal and selecron exhibited increased alkaline phosphatase activity even at higher applications rates (Omar and Abdel-Sater, 2001).
2.2. Utilization of organophosphorus insecticides by bacteria isolated from soil

Organophosphorus compounds are among the most common chemical classes used in crop and livestock protection and account for an estimated 34% of world-wide insecticide scales. OP compounds possess very high mammalian toxicity and therefore early detection and subsequent decontamination and detoxification of the polluted environment is essential. The wide use of OP pesticides such as isofenphos, chlorpyrifos, diazinon, phorate, ethoprophos, terbufos, phosalone, pirimphos methyl has created numerous problems, including pollution of the environment. OP pesticides, in general, are regarded as non-persistent. Chemical and physical methods of decontamination are not only expensive and time-consuming, but also in most cases they do not provide a complete solution. These approaches convert toxic compounds into less toxic states, which in some cases can accumulate in the environment and still to be toxic to a range of organisms. We now know that bioremediation provides a suitable way to remove contaminants from the environment as, in most of the cases, OP compounds are totally mineralized by the microorganisms. Fortunately, most OP compounds are degraded by microorganisms in the environment as a source of phosphorus or carbon or both. In this direction, several soil bacteria have been isolated and characterized, which can degrade OP compounds in laboratory cultures and in the field. Likewise, the biochemical and genetic basis of microbial degradation has received considerable attention. Available literature on the microbial degradation of xenobiotics indicates that most studies have considered three aspects: i) the fundamental basis of biodegradation, ii) evolution and transfer of such activities among microorganisms, and iii) bioremediation techniques to detoxify contaminated environment (Singh et al., 1999). However, the use of microorganisms for bioremediation requires an understanding of all physiological, microbiological,
ecological, biochemical and molecular aspects involved in pollutant transformation (Iranzo et al., 2001). The net result of interaction between xenobiotics and soil microflora is notoriously difficult to predict. Because, microbial communities that can degrade or can develop tolerance to, or are inhibited by, chemical mixtures greatly contribute to resilience and resistance in soil environments (Ramakrishnan et al., 2011).

In recent years, the role of soil microorganisms in affecting the persistence of agricultural pesticides has been the subject of two areas of study. The first is the capacity for rapid elimination of highly persistent or toxic chemicals. The reduced pesticide efficacy is attributed to enhanced biodegradation particularly of chemicals applied under a continuous cropping program. In one study, a streptomycete was isolated from a field soil sample previously treated with the insecticide isofenphos and found to be capable of growing on several commercial carbamates and OP insecticides (Gauger et al., 1986). In another laboratory study, degradation of widely used OP insecticide, monocrotophos in two Indian agricultural soils at two concentrations, 10 and 100 µg g⁻¹ soil, under aerobic conditions (60% water-holding capacity) at 28 ± 4 °C was studied Gundi and Reddy (2006). The degradation of monocrotophos at both concentrations in black vertisol and red alfinsol was rapid accounting for 96-98% disappearance of the applied chemical and followed the first-order kinetics. The rate constants (k) for vertisol and alfinsol were 0.0753 and 0.0606 day⁻¹, and half lives were 9.2 and 11.4 days, respectively.

Catabolism and detoxification occur when a soil microorganisms uses the pesticide as a carbon and energy source. The later process is facilitated by resistant microorganisms (Matsumura, 1988). The reduced persistence of OP insecticides was attributed to the activity of soil microorganisms (Chapman and Harris, 1982; Gorder et
The degradation of xenobiotic compounds by members of soil microflora is an important means by which these compounds are removed from the environment, thus preventing them from becoming pollution problems. Much work has been directed towards understanding the complexity of pesticide-microflora interactions in soil. Many studies have employed pure cultures of soil isolates or agar plate counts of soil populations (Visalakshi et al., 1980; Digrak and Ozcelik, 1998). Rache and Coats (1988) reported that a bacterial strain (Pseudomonas) was isolated from an isofenphos-treated culture medium, and it proved capable of using isofenphos as a carbon source. Several Pseudomonas spp. that metabolize OP and carbamate insecticides have been isolated from soil (Siddaramappa et al., 1973; Chaudhry and Wheeler, 1988). Some OP insecticides, such as diazinon, chlorpyrifos, ethion, parathion, fonofos, malathion and gusathion, are susceptible to microbial hydrolysis and many serve as carbon sources for the growth of pure and mixed cultures Flavobacterium sp., Pseudomonas sp., and Arthrobacter sp. (Ghisalba et al., 1987; Digrak et al., 1995). Gauger et al. (1986) reported that Streptomyces pilosus was capable of growing on several insecticides (carbofuran, cloethocarb, trimethacarb isofenphos, fonofos and phorate) although growth on terbufos was found to be nonexistent. Digrak and Kazanici (2001) reported that the total viable bacterial count in the isofenphos-treated soil sample was found to be higher than that of the untreated control soil samples during incubation. Moreover, it was observed that the treatment had no inhibitory effect on the development of other groups of microorganisms. Also, isofenphos-degrading Arthrobacter sp. was able to rapidly metabolize this compound. A granular formulation (5%) of monocrotophos, applied at a rate of 1.5 g a.i. ha\(^{-1}\) to an Indian clay soil, was dissipated rapidly with a half-life of 10 days (Agnihotri et al., 1981). Enhanced biodegradation responsible for rapid loss of
another OP insecticide, chlorpyrifos from Australian cane fields was attributed to fallen efficacy against cane grub (Robertson et al., 1998). Repeated treatments with an OP nematicide, fenamiphos resulted in enhanced biodegradation of the compound in soils of the United Kingdom with high pH (7.7) but not in soils with acidic pH (Singh et al., 2003). Adebayo et al. (2007) reported that the bacterial population in soil was significantly increased upon soil treatment with karate and thiodan. Among the pesticides, few significant effects of herbicides on soil organisms have been documented by Bunemann et al. (2006). Such reports are important because they disclose the behavior and potential harm caused by these chemicals in the environment, and these reports are useful in identifying data gaps for remediation by future research.

One of the important OP insecticides is acephate, a foliar spray insecticide, is used for control of a wide range of biting and sucking insects. Acephate dissipates rapidly with half-lives of < 3 and 14 days in aerobic and anaerobic soils, respectively. Laboratory degradation studies have been demonstrated that acephate can degrade through microbial degradation and aqueous hydrolysis. Since the rate of hydrolysis increases with increasing pH, degradation may occur more rapidly in alkaline soil than in acidic soil. A review of available literature indicated that the average half-life for acephate is 3-4 days under aerobic (flooded) conditions (Chevron, 1972a). Furthermore, acephate is rapidly degraded in soil by microorganisms under both aerobic and anaerobic conditions. The soil types in this experiment included loamy sand, sandy clay, silty clay loam, loam, and clay (Chevron, 1972a; 1972b). The same degradation products are formed in both aerobic and anaerobic soils. The metabolites formed are methamidophos and O-methyl N-acetylphosphoramidate (Chevron 1972c; 1972d). According to US EPA (1987), acephate dissipates rapidly with half-lives of less than 3 and 6 days in aerobic and anaerobic soils, respectively. The major
metabolite was found to be carbon dioxide in both types of soils. Most recently, Zhi et al. (2008) provided a thorough review of the literature that presented acephate-degrading bacteria isolated from soils in which acephate was used for a long period. The bacterial strain that belongs to *Chrysobacterium* sp. XP-3 has a strong ability of growth and reproduction in medium containing 1500 mg L\(^{-1}\) acephate.

Major advances in the degradation of buprofezin, an important insecticide, in flooded and upland soils under laboratory conditions have been made by Funayama et al. (1986). Buprofezin was gradually decomposed in soils under flooded and upland conditions, with half-lives of 104 and 80 days, respectively. After 150 days, five degradation products were identified by thin-layer cochromatography which include: 2-tert-butyliminio-5-(4-hydroxyphenyl)-3-isopropyl-perhydro-1,3,5-thiadiadin-4-one, 3-isopropyl-5-phenyl-perhydro-1,3,5-thiazin-2, 4-dione, 1-tert-butyl-3-isopropyl-5-phenyl-biuret, 1-isopropyl-3-phenylurea and phenylurea. As minor products, 2-tert-butylimino-5-phenyl-perhydro-1,3,5-thiazin-4-one or buprofezin sulfoxide were found in the flooded or the upland soil. \(^{14}\text{C}\)Carbon dioxide and bound \(^{14}\text{C}\) residue accounted for 23-24% and 13-21% of the applied radioactivity, respectively. Degradation of buprofezin remarkably delayed in sterile soils. Since neither formation of \(^{14}\text{CO}_2\) nor ring hydroxylatibn was observed in the sterile soils, buprofezin seems to have undergone complete mineralization in the soils under both flooded and upland conditions through biological transformation by soil microorganisms. Though biodegradation of OP insecticides and other pesticides by microorganisms in soil has been widely reported (Racke and Coats, 1988; Sharmila *et al.*, 1989; Digrak, 1994; Digrak *et al.*, 1995), the impact of soil bacteria on acephate and buprofezin has received less attention.