2.1 Hazards of Insecticides

Insecticides are used both in agriculture and as vector control agent in public health programmes. Significant amounts are used in forestry and livestock protection. More than 2600 different formulations of insecticides have been used in agricultural activities (Singh, 2009). Environmental contamination by means of insecticides is widely studied because of their extensive use in gardens, agricultural crops and homes. Due to their chemical characteristics, insecticides represent a type of toxicant that shows variable persistence for photochemical and biochemical degradation (Bandala et al., 2007).

Balwinder et al. (2006) studied the persistence of insecticides in soil under field conditions and reported that approximately 90-95% of its residues persist in the environment. Insecticide disturbs the delicate balance of a functioning ecosystem (Khan and Francis, 2005). Insecticide also affects the soil microflora which in turn results into infertility of soil (Rouchaud et al., 1994). Addition of insecticides affects the microbial components of an ecological niche and therefore effect is observed on biotransformation reaction occurring in soil (Bharadwaj and Garg, 2012).

The growth of symbiotic nitrogen fixing bacteria is also hampered due to the excessive use of insecticide in the agriculture field (Lakshmikantha, 2000). Insecticides, agrichemicals, and environmental contaminants inhibit the growth of rhizobium on plant roots which results into lower rates of nitrogenase activity, formation of fewer root nodules, and a reduction in overall plant growth and yield (Fox et al., 2007). Root nodule forming nitrogen fixing bacteria, free living phosphate solubilizers, antibiotic producing bacteria, mycorrhizae, growth hormone producing bacteria also get affected due to the insecticide application (Chen et al., 2001).

Insecticides along with its additives show a marked non target effect on rhizosphere. Rhizosphere population may either increase or decrease depending upon the insecticide employed. The diversity of resident microbes on phyllosphere and in rhizosphere also destabilized as the insecticides reach these areas. Only specific microorganisms which are tolerant to insecticides may remain and further multiply (Zhang and Bennett, 2005).
Certain bacteria possess the ability to survive under insecticide contaminated soil (Majumdar et al., 2012). Due to the extreme use of these compounds, a new problem concerned with harmful effect on non target economical insects like silkworm, honeybee, and on earthworms has been reported (Liu et al., 2007).

When croplands are treated, some impacts of insecticides were studied by Surendra Nath, (2000) and reported that insecticide residues shows non-target effect on terrestrial as well as on aquatic ecosystems.

High concentration of insecticides in soil requires remediation. If the insecticides are not degraded, leaching of them from soil to underground water occurs (Omar, 1998).

Insecticides are frequently found in aquatic environments in a particular concentration at which ecotoxicological effects can be expected (Kylin et al., 1998).

Insecticides normally leach through sandy soils more easily and rapidly than fine textured soil. Similar to leaching process the most serious source is soil erosion where insecticides directly mixed into streams with no time for decomposition (Bakshi et al., 1999).

Insecticides are commonly found in lakes and streams due to natural rain, irrigation or outfall from pesticide manufacturing factories as a result they contaminate the ground water and through leaching process mixed into river water where they inhibit the growth of macrophytes and fishes (Ayoola, 2008).

Due to accidental spillage of insecticides deaths of endangered and threatened species gives a great concern today. An insecticide’s capacity to harm aquatic animals is depends on its characteristics like exposure time, toxicity, applied concentration and persistence in the environment (Franklin et al., 2010).

Fishes are particularly sensitive to the water contamination (Begum, 2004). Hence, insecticides may significantly damage biochemical and physiological processes of the fishes (John, 2007). Various insecticides are responsible to show serious problems associated with health of fishes (Banaee et al., 2009).

Repeated exposure to certain insecticides can result in to lower resistance to disease, decreased body weight, reduced fish egg production and hormonal changes. The
insecticide exposure to aquatic animals reduces population abundantly and lower down the adult survival (Banaee et al., 2011).

Due to insecticide exposure several cases of neurologic and mental illnesses have been reported (Gupta, 2004). Insecticide mainly inhibits the receptors of central nervous system and destroys the integrity of ionic channels of the nerve cells.

Exposure of insecticide residues during pregnancy could results into mental retardation, reproductive organ anomalies, developmental disorders, behavioral disorders and various forms of endocrine disrupting effects on the offspring has been reported (Abhilash and Singh, 2009).

### 2.2 Bioaccumulation

Bioaccumulation refers to the accumulation of a substance in a biological tissue. From the total application of insecticide into the farm only 0.1% of insecticide reaches the target organism whereas rest ultimately enter soil or aquatic ecosystem, where if not metabolized or detoxified it may persist for longer periods and results in to bioaccumulation by various organisms (Konda and Pasztor, 2001).

Certain insecticides are readily absorbed by plant roots and results into bioaccumulation (Aziz et al., 1997).

Tilak et al. (2004) reported the factors responsible for accumulation of insecticides in the tissues of an organism as:

1. The concentration of pollution in the water.
2. The water temperature - if the metabolism of the organism increases.
3. The age and sex of the organism.

Most of the insecticides are more soluble in oil than in water. Thus, when ingested by non-target organisms they will bioaccumlate in the organism. They are more commonly deposited in the lipid globules (Surendra nath, 2000).
Landrum et al. (1996) reported about bioaccumulation of insecticide from sediment especially at lower levels of aquatic horizon, where invertebrates at benthic region plays a crucial role in remobilization of contaminants into aquatic food webs by means of sediment ingestion.

Aquatic animals are generally exposed to insecticides by three primary ways (Monteiro et al., 2006) as,

1. Dermal, direct absorption through the skin by swimming in insecticide contaminated water

2. Breathing, by direct uptake of insecticides through the gills during respiration, and

3. Orally, by drinking insecticide-contaminated water.

Due to the organs lipophilicity, insecticide can easily pass into biological membranes and responsible to increase the sensitiveness of aquatic animals. After passing into biological membranes they are rapidly metabolized and finally concentrated in various organs of fish (Sharma et al., 2009).

Bioaccumulation of insecticides in fish depends on amount of fat reservation in different tissues, on the species life stages and on physical and chemical properties of insecticides (Siang et al., 2007).

The solid Organophosphorus insecticide is known to accumulate in human adipose tissue and they are also known to affect nervous system function (US EPA, 2008).

When an organism containing a pollutant is ingested, the pollutants are easily accumulated in the tissues of predator (McKay and Fraser, 2000). Consumption of many preys by organism may results in to increase in concentrations of the pollutant in its tissues. In food chain this process may continue, leaving the top predator with high and lethal concentrations of the pollutant (Reigart et al., 1999).
2.3 Phytotoxicity

Chouychai and Lee (2012) reported that some insecticide shows a marked effect on cell division in root meristem of *Bidens laevis* when grown in presence of 0.01-5µg\(^{-1}\) of concentration of insecticides hydroponically.

Safi *et al.* (2002) analysed residues of chlorpyrifos, carbofuran and other pesticides on tomatoes, cucumber and strawberries by GC-MS technique and concluded that tomatoes showed the least number and level of insecticide residues while strawberries showed the greater number and level of insecticide residues.

Lutzinsky *et al.* (1996) reported about toxicity of Imidacloprid on field-grown plants, such as tomato and cucumber. It appears that excess imidacloprid in the plant disrupts metabolism sufficiently to cause senescence of older leaves and abnormal development of new leaves that culminates in marginal necrosis (Omar, 1998).

2.4 Toxicity of Organophosphate insecticide on environment

As far as the mode of action of organophosphorus insecticides is concerned, it is known to inhibit acetylcholinesterase (AChE) activity which causes the disruption of nerve function and eventual death. Inactivation of AChE results into accumulation of the neurotransmitter acetylcholine which results into disruption of signal transmission and synaptic blockages (Ferrari *et al*., 2004).

Organophosphorus insecticides are responsible to inhibit acetyl choline esterase and cause physiological damage (Rao *et al*., 2003).

In case of aquatic environments they inhibit the AChE and induces alteration in the shaking palsy, spasms, swimming behavior of fishes and other undesirable effects (Sharbide *et al*., 2011).

Organophosphate insecticides like quinalphos, dimethoate, diazinon, and chlorpyriphos had a range of effects on soil environments particularly on soil microorganisms (Pandey and Singh, 2002).

Effect of organophosphorus insecticide on soil enzymes was also reported (Menon *et al*., 2004; Singh and Singh, 1998).
As far as the aquatic toxicity of insecticide is concerned, dichlorovos is responsible for dysfunction or reproductive failure in aquatic animals (Ragnarsdottir, 2000).

In case of organophosphorus insecticides reproductive toxicity indicates changes in fertilization rate, hatchability of larvae, fecundity and the pattern of breeding response. Fish behavior is greatly affected by physiological change caused by accumulation of the acetylcholine due to use of various insecticides (Cong et al., 2008). These changes disturb the overall survivability of the animals in their natural environment.

Thus, due to exposure of chlorpyrifos, propoxur and diazinon AChE inhibition results and are considered as a specific biomarker of exposure organophosphorus insecticides (Uner et al., 2006). Organophosphorus insecticides are responsible for death in more than 70% cases and showed high toxicity in certain developing countries (Gupta et al., 2001).

As far as the effect of organophosphorus insecticide on soil environment is concerned, they take several months to degrade in the soil. When a large concentration of methyl parathion and malathion reaches the soil as an accidental spill, the degradation occurs after many years (Esteve-Nunez et al., 2001).

Madhuri and Rangaswamy (2002) observed that soil samples receiving 2.5 kg ha\(^{-1}\) of the insecticides dichlorvos, phorate and methomyl for more than 20 days are responsible for the decrease in phosphatase activity of soil. The nitrifying bacteria are most susceptible and inhibition of nitrification has occasionally been observed after an application of insecticides. Nodulation and nitrogen fixation processes are reduced and sometime abolished by recurring insecticide use (Dar, 2010).

Adiroubane et al. (2003) conducted an experiment to study the impact of monocrotophos 36 SL and phosphamidon 85 EC on the total heterotrophic bacterial and nitrogen fixing bacterial population in the rhizosphere soil and phyllosphere of irrigated rice ecosystem. The observation indicated that there was a sudden reduction in bacterial population immediately after spraying, but subsequently recovery was occurred.

Pandey and Singh (2002) observed short term inhibitory effects on the total bacterial population after chlorpyriphos and quinalphos application which were recovered within 60 and 45 days of treatment in soil. The population of fungi was significantly enhanced after chlorpyriphos treatment.
Singh et al. (2004) studied the effects of fenamiphos chlorothalonil and chlorpyriphos individually and in combination on soil microbial activity and reported that the soil microbial parameters like total microbial biomass and enzyme activities were stable in the insecticide free control soils for about 90 days of inoculation period but all of them were adversely affected in the presence of added pesticides.

Aggarwal and Gupta (1996) worked on the effect of organophosphorus insecticides amidithion and phosphamidon on nitrification and ammonification in soil and concluded that these insecticides did not inhibit nitrification and ammonification processes but slightly stimulated the processes at lower concentrations (25 and 125 ppm) while they were depressed at higher concentration of 1250 ppm.

The soil enzyme like dehydrogenase, phosphatase, β-glucosidase, and urease are reduced due to the application of phorate, malathion, parathion in soil was reported (Rowland et al., 1991).

Garg and Tandon (2000) studied the effect of organophosphorus insecticide on bacteria and actinomycetes of salt affected alkaline soil and concluded that these insecticide persist more in such environment.

The bacteria which can degrade the insecticide are isolated by many workers. It includes an organophosphorus insecticide degrading *Rhodococcus* sp., a atrazine degrading *Pseudomonas* sp. and chlorpyriphos degrading *Pseudomonas* sp. (Parekh et al., 1994; Ralebits et al., 2002; Mallick et al., 1999).

Other important bacterial genera capable of degrading organophosphorus insecticide include *Burkholderia* and *Hyphomicrobium* (Wang and Chen, 2006).

Degradation of organophosphorus insecticide at molecular level was reported by Somara et al. (2002) and concluded that it is concern with the *opd* genes that have been described in *Flavobacterium* and *Pseudomonas* species and are plasmid borne.

A similar gene, *opdA*, is present in *Agrobacterium radiobacter*'s chromosome responsible for organophosphorus insecticide degradation has been reported (Horne et al., 2002). Cheng et al. (1993); Zhang et al. (2006) and Cheng et al. (1999) were also reported about *Alteromonas* species bearing *opaA* genes.

Zhongli et al. (2001) reported about gene *mpd*, encoding for organophosphates hydrolase in *Plesiomonas* sp. It has also been found in other genera like
*Pseudaminobacter*, *Ochrobactrum Brucella* and *Achromobacter* and is located in the chromosome (Zhang et al., 2005).

### 2.5 Toxicity of Neonicotinoid insecticide on environment

As far as the toxicity of neonicotinoid insecticides are responsible to cause toxic effects on non target organisms. They are more effective in case of sucking and chewing pests. They act as competitive inhibitors in central nervous system with respect to nicotinic acetylcholine receptors (nAChR) (Elbert et al., 2008).

Neonicotinoids interact with nicotinic acetylcholine receptors at the peripheral and central nervous system which results into paralysis, excitation and death (Liu and Casida, 1993).

Neonicotinoids are the most widely studied group at the molecular levels and most widely used against many sap-sucking insects (Dai et al., 2007).

The other insecticides are nitenpyram, thiamethoxam, thiacloprid and acetamiprid all of them possess the property to act on the nicotinic acetylcholine receptor (Reddy and Rao, 2008).

Neonicotinoids also shows toxicity to soil organisms like earthworms (Ishaaya and Degheele, 1998).

An and Lee (2008) reported that after application of imidacloprid around trees it shows adverse effect on earthworm population. Insecticide like imidacloprid is responsible for the mortality of earthworms.

Imidacloprid was also found to be more toxic than chlorpyriphos, fipronil and carbaryl in case of earthworm *Aporrectodea trapezoide* (Mostert et al., 2002).

The reproduction and growth rate of earthworms are also found to be affected due neonicotinoid insecticides and are considered as major environmental threat (van Gestel et al., 1992; Wu et al., 2011).

Toxicity of imidacloprid to the non-target terrestrial arthropod *Porcellio scaber* was also reported (Drobne et al., 2008).

The imidacloprid was also found to affect honeybees severely and the honey collected by them is found to contain about 3 µg kg⁻¹ of imidacloprid (Chauzat et al., 2006).
Insecticides such as triazophos and dimethoate also show very toxic and hazardous effects on honey bees (Mineau, 2002).

The nymphicidal and ovicidal activities of acetamiprid application on cotton seedlings are more effective than imidacloprid; however in soil imidacloprid was found to be more effective than acetamiprid (Horowitz et al., 1998).

Neonicotinoid insecticide also found to affect aquatic organisms severely. Imidacloprid is found to affect the fresh water organism *Daphnia magna* and *Cypridopsis vidua* whereas thiacloprid toxicity is observed with *Sympetrum* and *Gammarus*.

Both of these insecticides show 50% mortality of several species of exposed arthropods (Sanchez-Bayo, 2011).

The toxic effect of diazinon on acetyl choline esterase is temporary on the contrary imidacloprid effects could last much longer and the acetyl choline esterase is permanently blocked.

Punitha et al. (2012) has been studied the impact of acetamiprid on selected soil enzymes urease and phosphatase and reported that the acetamiprid insecticide shows toxicity in soil and inhibitory effect was reduced after degradation of applied insecticide.

Yao et al. (2006) studied soil toxicity of acetamiprid with reference to soil enzyme activity and reported that it does not show any threat to soil enzyme but responsible to show a toxic effect on soil respiration.

As far as the biodegradation of neonicotinoid insecticide is concerned, *Pseudomonas* sp. 1G was found to degrade and therefore can be used for the biotransformation of neonicotinic compound imidacloprid and thianethoxam (Pandey et al., 2009).

Chen et al. (2008) also reported about acetamiprid degradation by *Stenotrophomonas maltophilia* CGMCC 1.1788.

Kumar et al. (1996) studied microbial degradation of pesticides with special emphasis on the role of catabolic genes and reported that by using recombinant DNA technology in the development of an organism, the degradation of several xenobiotic compounds is possible.
2.6 Ecotoxicity of insecticide additives

As far as the ecotoxicity of insecticide additives is concerned, little is known about the environmental effect of additives after application on agricultural land (Bunemann et al., 2006).

The lethal effect of insecticides along with its additives is not usually confined to target pest alone but in the course non target organisms are also affected (Hayo et al., 1996).

Insecticide additives show a marked effect on the growth of soil microorganisms. They not only reduce their number but also are responsible for germination inhibition of seeds (Racke et al., 1997).

Certain insecticide additives like petroleum distillates used in the formulation of malathion are more toxic than the insecticide itself (Olgun and Misra, 2006).

2.6.1 Tallowamine ethoxylate

Tallowamine ethoxylate is a surfactant generally used in the formulation of organophosphorus insecticides. Surfactants are a class of xenobiotics which, due to their chemical nature, accumulate at interfaces including the solid/liquid interface of stones and sediment particles in rivers (Stewart, 2007). Tallow amine ethoxylate is responsible to show considerable ecotoxicity.

Surfactant such as alkylamine ethoxylates and alcohol ethoxylates are the most commonly used additives in insecticide formulations (Krogh et al., 2003). Aliphatic amine ethoxylates and alcohol ethoxylate are responsible for decreasing the microbial flora of soil by means of clay adsorption (Krogh et al., 2003).

In case of glyphosate insecticide formulation use of polyoxyethylene amine is common today. It was found to be more toxic to microorganisms than IPA salt of glyphosate or glyphosate acid (Tsui and Chu, 2003).

Effect of ethylamine from a glyphosate formulation was studied by Santos et al. (2005) and reported that it shows a more toxic effect on Bradyrhizobium.

Due to physical properties of tallowamine ethoxylate like low temperature behavior, miscibility in cold water, ecotoxic problem such as severe flocculation in soil was observed today (Merlet et al., 2011).
The tallowamine ethoxylate is more toxic than the active ingredient of insecticide. It is more toxic for aquatic organisms particularly for the amphibians. The LC$_{50}$ value reported is about 1.1mg/L for the single amphibian species (Howe et al., 2004).

The tallowamine ethoxylate persist for longer time in soil and its high concentration hampers the growth of other soil microorganisms (Rokade and Mali, 2012). Cano and Dorn (1996) reported that surfactant absorption is always correlated with the clay percentage and organic carbon content of soil. The adsorbed tallowamine like additives interfere with the soil energy transfer processes by showing adverse effect on the growth of bacteria which thus leads to soil infertility problems (Rokade and Mali, 2012).

2.6.2 Benzothiazole and its derivatives

Benzothiazoles derivatives interferes with the soil microbial activity and mainly interferes with the nitrogen cycle thus benzothiazole are considered as nitrification inhibitors and hampers the germination of plants (De Wever and Verachtert, 1997).

Benzothiazole derivatives are commonly found in the both surface water as well as in the industrial effluent (De Wever and Verachtert, 1997). An ecotoxic effect of benzothiazole has been reported (Hendriks et al., 1994), and (Reemtsma et al., 1995).

2-Mercapobenzothizole like compound is interfered with the bacterial cell membrane and affects respiratory mechanism of the bacterial cell (De Wever et al., 1997; De Wever et al., 1998).

Benzothiazole derivatives are found to have activity against the common housefly *Musca domestica* and other arthropod insects like *Tetranychus urticae*, *Culex spp.*, *Myzus persicae*, *Tribolium castaneum* and *Pluetella xylostella* (Blade et al., 1992).

As far as the toxicity of benzothiazole concerned it is also known to induce tumor and found to be toxic in the concentration of 600mmol L$^{-1}$ for the aquatic life (Yoshioka and Ose, 1993).

Benzothiazoles are potential tumourogenic agent and responsible to show adeno-carcinomas or pituitary adenomas to rodents upon oral exposures (McMahon, 2006).
The common symptoms of benzothiazole toxicity are necrosis and severe stomach lesions including hyperplasia and inflammation in humans has been reported (US EPA, 2006).

Ghosh and Rokade (2012) studied the toxicity of benzothiazole derivative 2-Mercaptobenzothiazolyl-(Z)-(2-aminothiazol-4-yl)-2-(tert-butoxycarbonyl) isopropyriminoacetate and reported that benzothiazole derivative show severe toxicity on soil environment and such toxicity can be lowered by means of genetically modified bacterial strains.

In aquatic environments, toxicity of 2-(thiocyanomethylthio) has been reported and found to be responsible to reduce the swimming ability in juvenile *Salmonids* (Nikl and Farrell, 1993).

Benzothiazole derivative like 2-(Thiocyanomethylthio) benzothiazole can persist in river water for 4 months whereas its persistence period increases in sea water. It was previously get studied that 2-(Thiocyanomethylthio) benzothiazole is responsible to show a severe toxicity to fishes like *Oncorhynchus kisutch* and *coho salmon*, where decrease in blood lactate level of them was reported (MacKinnon, 1992).

These additives act as endocrine disrupter and blockers of sex hormones. They found to be interfering with the abnormal sex ratios, unusual mating behavior and abnormal sexual development (Nawrocki, 2005).

In case of benzothiazoles determination from environmental samples, identification of 2-(4-Morpholinyl) benzothiazole by means of gas chromatography equipped with a flame photometric detector (GC-FPD) has been reported (Kumata *et al.*, 1996).

Table 3 shows the identification of benzothiazoles by means of analytical instruments by some authors.
Table 3: Summary of benzothiazoles identification (Krouani-Harani, 2003)

<table>
<thead>
<tr>
<th>Benzothiazoles</th>
<th>Medium</th>
<th>Method of identification and/or quantification</th>
<th>Authors</th>
<th>Year</th>
<th>Observations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mercaptobenzothiazole and derivatives</td>
<td>Petrotatum and in buffer solutions</td>
<td>HPLC</td>
<td>Hansson and Agrup</td>
<td>1993</td>
<td>Good separation</td>
</tr>
<tr>
<td>Benzothiazole</td>
<td>River water</td>
<td>Solid-phase extraction-GC-MS</td>
<td>Louter et al.</td>
<td>1994</td>
<td>Well suitable for the trace-level of surface water determination</td>
</tr>
<tr>
<td>2-substituted benzothiazoles</td>
<td>wastewater</td>
<td>HPLC/Electrospray tandem mass spectroscopy (HPLC/ESI-MS/MS)</td>
<td>Reemtsma</td>
<td>2000</td>
<td>Suitable method for the direct analysis of wastewaters</td>
</tr>
</tbody>
</table>

2.6.3 Benzyl benzoate

It has been known from many years that house dust is responsible to cause asthma, rhinitis and eczema. The mite *Dermatophygoids pteronyssinus* has been identified as a major source of house dust allergen. Chronic exposure of mites can cause bronchial hyper reactivity and chronic asthma. For reducing the proliferation of dust mites the acaricidal solution must contain the active ingredient benzyl benzoate (Suh *et al.*, 1999).

Benzyl benzoate shows toxic effects on rats which includes hematological changes (Kockaya and Kilic, 2011). In man, it is a skin sensitizer and causes gastro-intestinal effects in an infant and can affect the nervous system especially when used on children. Symptoms like jerking movements of the legs or arms and loss of consciousness have been reported (Schachner and Ronald, 2011).

As far as the biodegradation of insecticide additive is concerned, the chief ways of biodegradation of insecticide additives from the environment includes biosorption, bioaccumulation, reduction, solubilization, precipitation (Alexander, 1999).
The success of biodegradation technique is directly related to the metabolic capability of involved microorganism and can be affected by the surrounding environment. Siddavattam et al. (2004) studied biodegradation of such hazardous pollutants and concluded that it can be possible by means of microbes having genetic diversity and metabolic versatility. For the degradation of insecticide along with its additive bacteria carry mechanism like intracellular accumulation, oxidation or reduction was reported (Ahmad et al., 2010). Conversion of these pollutants was studied by Liu and Sulfita (1993) and concluded that it results into less harmful end products which often integrated into natural biogeochemical cycles. The extreme toxicity of pesticide or insecticide additives is often lost in the first transformation step. De Wever et al. (1998) isolated a pure culture and identified as a *Rhodococcus erythropolis* which was capable of degrading benzothiazole and its derivatives. Biodegradation of benzothiazole by means of *Rhodococcus* strain PA has been reported (Gaja and Knapp, 1997). In aerobic degradation, thiocyanomethyl thiobenzothiazole was transformed to mercaptobenzothiazole and benzothiazole by unknown pathways (Reemtsma et al., 1995). Basu et al. (2003) studied benzyl benzoate degradation where conversion of benzyl alcohol to benzaldehyde and finally to benzoic acid by means of *Pseudomonas putida CSV86* has been reported. The biotransformation of benzyl alcohols to benzaldehyde in *Escherichia coli* JM101 shows that the enzyme XylB contributes to the benzaldehyde formation by means of benzyl alcohols dehydrogenation (Buler, 2000). In case of anaerobic degradation of aromatic compounds, benzoate molecule plays a central role where a large input of energy is required (Egland et al., 1997). In case of brown-rot basidiomycetes like *Gloeophyllum trabeum* and *Tyromyces palustris* the benzaldehyde degradation study shows simultaneous oxidation and reductions where benzoic acid and benzyl alcohol are the major products (Kamada et al., 2002).
2.7 Biodegradation

Degradative reaction involves oxidation; hydrolysis, cleavage of important bonds etc. are the central conduit of insecticide degradation (Martinkova, 2008).

Ghosh et al. (2010) reported that biodegradation of insecticide compounds not only requires the knowledge of microorganism that can degrade a particular compound but also an understanding of the pathways involved in the degradation both at physiological and molecular level.

Consortium of microorganism has a powerful potential of biodegradation. Due to exchange of genetic information in between the organism the degradation of complex compounds occurred more rapidly (Mohapatra, 2008).

Bacteria possess the ring fission metabolism during degradation of insecticide compounds (Singh, 2008).

Microbes can degrade these compounds either totally or partially depending on the number of aromatic units and especially on the type of substituent’s (Jothimani and Bhaskaran, 2003).

Bhalerao and Puranik (2007) reported that by means of aerobic degradation and using bacteria like Klebsiella oxytoca, Bacillus spp., Micrococcus sp. insecticides like alpha-endosulfan, beta-endosulfan can be degraded.

In case of aerobic metabolism of insecticides several bacteria like Pseudomonas aeruginosa, Clavibacter michiganense, Arthrobacter atrocyaneus, Bacillus megaterium Pseudomonas mendocina, Agrobacterium radiobacter and other Pseudomonas species has been reported (Bhalerao and Puranic, 2009).

Similar to aerobic process, anaerobic transformation also plays an important role in the degradation of insecticides particularly in deep soil, or ground water region (Andre, 2007).

In this region molecular oxygen is not available for the degradation, so the anaerobic bacteria play an important role in biodegradation of insecticide contaminated ground water region (Allard et al., 1997).
Figure 8 shows the mechanism of aerobic degradation of aromatic compounds.

![Aerobic degradation of aromatic compound](http://blog.nus.edu.sg/yiuyan/2009/11/21/hazardous-waste/)

Figure 9: Aerobic degradation of aromatic compound (Discoveries, 2009)

Anaerobic bacteria like methanogens, *Desulfovibrio* species, Fe (III) reducing bacteria clostridia and sulfate reducers have the ability to reduce the nitroaromatic compounds (Iwamoto and Nasu, 2001).

After the initial breakdown of insecticide additives in soil region, growth of strict anaerobic methanogenic bacteria like *Methanobacterium*, *Methanobacillus*, *Methanosarcina*, and *Methanococcus* also occurs (Subba Rao, 2000).
Thus, biodegradation of organic chemicals of solid waste or biomass or waste water treatment is occurred in two ways i.e.either in the presence of oxygen by respiration or in absence of oxygen by means of methanogenesis and denitrification (Welander et al., 1999).

2.8 Phytoremediation

Phytoremediation is a newly developed and emerging technology which uses plants to remove pollutants from water and soil. The term “phytoremediation” is relatively new coined in 1991(U.S. EPA.1999).

*Phytoaccumulation* is the process where plants absorb contaminants into the roots and above ground shoots or leaves (US EPA, 1999).

Phytoremediation is the new technology where the organic contaminants are removed from the environment either by means of plant microbe interaction or by means of plant mechanis (Gerhardt et al., 2009). It is helpful technology as it saves remediation cost by bioaccumulating contaminants from a large area.

Phytoremediation results in to the uptake of pollutant from soil or water and subsequently cause their transformation to more stable and less toxic form (Raskin and Ensley, 2000). Microorganisms that degrade organic pollutants in culture sometimes may fail to function when inoculated into natural environments because the pollution levels in nature may be too low to support their growth (Baudouin et al., 2002).

The disadvantage of this technique involve that there are concerns of bioaccumulation by crop plants by means of root uptake. It leads to active or passive process of translocation where it also transfers to other part of plant body (Chiou et al., 2001).

This process also produces a plant mass and the contaminants can be transfer for recycling or disposal (Correia et al., 1994).

The phytoremediation technology not gives assurity that it will always make an insecticide free environment (Li et al., 1994).