The field of aquatic toxicology has been defined as the qualitative and quantitative study showing adverse or toxic effects of chemicals on aquatic organisms (Rand and Potrocelli, 1985). The subject of enquiry deals with the study of transport, distribution, transformation, and the ultimate effect caused by the chemicals in the aquatic environment. Today, as a result of the ever booming industrialization, urbanization and the unchecked expansion of modern agricultural practices, excessive load of chemicals, and xenobiotics like pesticides reach the aquatic environment by land runoff and leaching. Insecticides/herbicides are among the most widely used toxic chemicals for various purposes in industry, forestry, horticulture, agriculture and household. Water often serves as a sink for these chemicals after their application in these different fields. These pollutants are often not entirely specific for their target organisms. Non-target species, such as aquatic biota, can be affected because of the universal presence of insecticides in the environment. It is reported that there exists a large amount of circumstantial evidence to indicate a decline in the population level (Houlahan et al., 2000), impairment in growth, and to show an increase in disturbed behaviour (Alvarez and Fuiman, 2005), deformities (Klump et al., 2002) and various diseases in aquatic organisms (Zelikoff et al., 2002). These effects suggest that pesticides at sublethal concentrations are present in the environment (Tanguy et al., 2005; Amando et al., 2006).

Pesticides are the only toxic chemicals deliberately released into the environment in large amounts. Their potential to cause adverse effects to human and wild population has been a subject of intense study and that has led to the development of stringent and encompassing regulations for the risk assessment of novel formulations in order to control the use of compounds currently in use. The organophosphorus insecticides (OPs)
were introduced as replacements for persistent organochlorine insecticides after the tendency of DDT and its metabolites to bioaccumulate in the ecosystem and to cause adverse health effects, particularly in top predators (Woodwell et al., 1967; Peakall et al., 1975; Murphy, 1986). In some countries the latter was legally banned in 1970s and in others only restrictive uses were allowed. The increased use of OPs, originally seen as harmless to the environment due to their rapid breakdown and low persistence, has led to a different range of ecotoxicological problems associated with their high acute toxicity. Evidence gathered from the last twenty years of experiments show that OPs can interfere with immune system and exert immunotoxic effects in laboratory animals through both anticholinergic and non-cholinergic pathways (Wong et al., 1992; Barnett and Rodgers, 1994; Vial et al., 1996).

Way back in 2000, November 23, globally threatened Sarus cranes *Grus antigon* who were resident at Keoladeo National Park World Heritage site and the surrounding area near Bharatpur, Western Rajasthan, India, were joined by wintering common cranes *Grus grus*. Fifteen Sarus cranes and three common cranes were found dead in a field adjacent to the park, where wheat seed had been sown the previous day. Chemical analyses of seed samples from the field and the cranes’ alimentary tract contents identified residues of the organophosphate insecticide monocrotophos. Monocrotophos concentrations of 0.8 and 1.8 ppm were found in wheat samples, and 0.2–0.74 ppm in the alimentary tract contents of five of the seven cranes examined. No other organophosphate or organochlorine pesticides were detected. Scientists concluded that the cranes died from Monocrotophos poisoning after eating treated seed.
Monocrotophos (MCP), commonly known as Azodrin, is one of the most extensively used OPs, in agriculture and in animal husbandry. It is a broad spectrum systemic organophosphate insecticide used on a range of crops, primarily cotton, soybeans and rice, and other crops like wheat, potatoes, alfalfa maize and sugarcane; also on vegetables (Guth., 1994) and in fruit orchards (Donzel., 1994). Monocrotophos has been used for pest control since the 1960s, and its use has resulted in poisoning of non-target species in large number. It works outs its way systemically. On contact it will be absorbed through skin. It works its way through inhalation and ingestion too. It has been categorized as extremely hazardous (Bhadbha, 2002). Hydrolysis rates of monocrotophos in soil and aqueous environment are pH dependent and half-lives of monocrotophos in pH 3 and 9 at 25 °C are 13 days and 26 days respectively and it persists in soil in the dark for 30 days at neutral pH (Lee et al., 1990). It is hydrolysed in alkaline conditions. Solubility of monocrotophos in water is 100% (Tomlin., 1995). Monocrotophos sprayed on crops, can remain as soil residue and enter water sources such as rainwater and ground water by leaching through soils.

Monocrotophos was one of the most stable OPs so far studied. In a study on the persistence of pesticides in river water, monocrotophos was found not to degrade (100% recovery as parent) after 8 weeks (Eichelberger and Lichtenberg, 1971). Water used as sample in this study was taken from a small stream into which domestic and industrial waste together with farm runoff had already run in. The sample showed 7.3 as pH value, and it increased to 8.0 after 8 weeks. Dosed fish were kept at laboratory temperature in both natural and artificial (room) light. Analysis was done by proper extraction and Gas Chromatography. However, a study conducted in paddy field showed rapid degradation in the aqueous phase (Osgerby and Page, 1969).
After deliberately killing of 62,000 birds with monocrotophos on a farm in Argentina in 1997, and poisoning thousands of Swainson’s hawks with secondary infliction (Goldstein et al., 1999), Novartis, one of the manufacturers of that time, announced they were phasing out all the manufacture and sales of monocrotophos globally (Winegrad, 1998). A number of studies were conducted on the toxicity of monocrotophos on different aquatic organisms, and finally it was adjudged to be a potent neurotoxicant (Qadri et al., 1994; Rao et al., 1991; Rao et al., 1992a; Venkateswara Rao et al., 2001). Several studies confirmed the occurrence of monocrotophos in aquatic systems in concentrations that can be deleterious to various life forms.

Reviewing Overseas Regulatory Actions, it is of importance to observe that, monocrotophos has been voluntarily withdrawn from sale in the US in 1989 following concern on its toxicity to non-target species. It is also banned in Indonesia, Sri Lanka and Philippines; severely restricted in Kuwait (for use on plants in flowering stage only); Malaysia (for use on coconut tree and on oil palm by trunk injection) and Germany (not to be handled by adolescents, pregnant and nursing women). monocrotophos is not used in the United Kingdom. In 1996, the companies selling monocrotophos in Argentina have voluntarily agreed to withdraw the product from the market, and bought back all existing supplies following concerns over bird deaths from its use in grasshopper control (Pesticide Action Network North America Updates Services, 4 November 1996).

As chemical analyses alone may not suffice to explain the adverse effects of the complex mixtures of chemicals present at contaminated sites, determining the extent and severity of water contamination by pollutants is often difficult. In the case of aquatic environment, because of its
peculiarity, biomonitoring by means of biomarker parameters assessed in
different native species is a useful tool. It has the advantage of providing a
quantitative response as well as valuable information of ecological relevance
on the chronic adverse effects caused by water pollution (van der Oost et al.,
2003).

Indiscriminate use of pesticides in agricultural operation adversely
affects the aquatic environment to a very great extent potentially affecting
fish. The responses of fish to such environmental challenges are initially
reversible, but prolonged exposure to environmental pollutants brings about
permanent (pathological) changes in fish physiology. Sometimes, death of
the aquatic living resources such as fish may not occur but they suffer
physiological perturbations like altered enzyme activity mainly due to high
load of these environmental anthropogenic agents (Racicoot et al. 1975).
With the increasing awareness of the impact of these pollutants, especially
in developing economies, it is essential that sensitive biochemical and/or
physiological parameters which could act as early warning signals of
pollution at sublethal levels be studied. Hence, changes in the activity of
enzymes or other biomarkers (Moore and Simpson, 1992) have been used
as possible tools for aquatic toxicological research.

Many circumstances, biotic and abiotic, promote the antioxidant
defense response in fish. Factors inherent to the fish itself, such as age,
phylogenetic position and feeding behavior, and environmental factors such
as types of diet available, temperature, dissolved oxygen, toxins present in
the water, pathogens, and parasites can either enhance or decrease the
antioxidant defenses. Research in fish has demonstrated that mammalian
and piscine systems exhibit similar toxicological and adaptive response to
oxidative stress. This suggests that piscine models in addition to traditional
mammalian models may be useful for further understanding the mechanisms underlying oxidative stress response (Kelly et al., 1998). Studies on the oxidative stress in fish open a number of research lines aimed at providing greater knowledge of fish physiology and toxicology. Over and above, such studies would give more accurate information concerning the response of antioxidant defense in different species under different circumstances as well as the regulatory mechanism of the responses. Such future studies will, of course, do good to aspects allied to fish farming and aquaculture production.

Poisoning by pesticides from agricultural fields is a serious water pollution problem and its environmental long-term effect may result in the incidence of poisoning of fish and other aquatic life forms (Jyothy and Narayan, 1999). Fishes like *Heteropneustes fossilis* and *Clarius batrachus* are especially prone to serious pesticide pollution as their habitat is mostly the agriculture area. Though only few studies are conducted in this area, it can be assessed from the local information that, population of such fish is on the verge of vulnerability due to extensive use of pesticides. The knowledge of sublethal effects of xenobiotic compounds on hematological parameters, enzyme activities and metabolite concentrations is very important to delineate the fish health status and provide a future understanding of ecological impacts. These pesticides act by causing inhibition of cholinesterase enzymes (ChE) by formation of enzyme inhibitor complex (O’Brien, 1976) and damaging the nervous system. These effects may result in metabolic disorders. Associated to cholinesterase activities, a study of other enzymes such as phosphatases and aminotransferases close to intermediary metabolite determination provides a wider view of metabolism. Interest in toxicological aspects has grown in recent years and research is now increasingly focused on mechanistic aspects of oxidative...
damage and cellular responses in biological system. The term 'biomarker' is generally used in a broad sense to include almost any measurement reflecting an interaction between a biological system and a potential hazard, which may be chemical, physical or biological (WHO, 1993). As biomarker stands for immediate responses, they are used as early warning signals of biological effects caused by environmental pollutants.

The present work attempts to assess the toxicity of organophosphorus insecticide monocrotophos on the experimental organism selected for this study namely stinging catfish (*Heteropneustes fossilis* (Bloch)), and to probe into the stress responses of the organism.