

Review Article

Microbe-Assisted Biodegradation, Bioremediation and Metabolic Engineering

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We summarize here the recent research and development in the area of microbe assisted biodegradation and remediation processes by different research lab in the country. The various substituted hydrocarbons as pollutants have been studied by different groups. The depth of the study includes the characterization of key bacterial metabolic pathway for target pollutants through both genome and proteome levels. By understanding the pathway, in few cases the metabolic engineering approached have been demonstrated for the efficient utilization of pollutants. The better understanding to key enzyme were explored using protein modeling or by understanding its expression through heterologous host. The review also presents few case studies where pilot scale demonstration of contaminated sites with selected pollutants, have also been carried out effectively.

Keywords: Biodegradation, Bioremediation; Metabolic Engineering

Introduction

A wide range of aromatic compounds including solvents are being used in industries for the production of pesticides, dyes, plasticizer, paints, textiles, etc. The impact of these pollutants is very serious as these are capable of inducing endocrine disruption, genotoxic, mutagenic and carcinogenic effect on human health. The effects are more alarming due to long term persistence of these pollutants in the environment. Therefore, to overcome the effects, it is essential to remediate these compounds. The aromatics in the environment are eliminated by processes like volatilization, photooxidation, chemical oxidation, adsorption to soil matrix or biodegradation by microorganisms (bioremediation). Bioremediation is an environmentally “green” method of using microbes – either naturally-occurring or introduced (in naturally-occurring form or engineered) – to break down pollutants to non-toxic substances. Bioremediation can be *in situ* or *ex situ*. The biological processes for

treating toxic pollutants are gaining interest over chemical and physical methods especially in terms of economic and environmental development; where instead of toxic by-products generated under non-biological methods, biological processes produce useful and relatively less toxic intermediates that enter the biogeochemical cycles. Bioremediation not only serves removal of contaminants but also has other positive aspects including enhancement of the soil quality, waste water treatment, carbon sequestration etc. However, bioremediation also encounters some limitations viz. toxicity of the compound, presence of simpler carbon source, salinity and other stressors. There are number of research groups working in the area of bioremediation and environmental engineering, which will help to clean up the environment. This report gives the recent advances in the field of microbe-assisted biodegradation and bioremediation and metabolic engineering in the country.

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Microbe-Assisted Biodegradation and Bioremediation

Some of the reported bioremediation technologies include natural attenuation, bioaugmentation, bioleaching, bioreactor, biostimulation, electro-kinetic mediated bioremediation and phytoremediation (Paliwal et al., 2012, Pinjari et al., 2013, Kundu et al., 2014, Reddy et al., 2016, Chatterjee et al., 2017, Juwarkar 2012). The starting point for any bioremediation activity is the assessment of the inherent microflora at a contaminated niche. Based on their catabolic capacities, remediation processes can be designed. A large number of bacteria including *Pseudomonas*, *Alcaligenes*, *Sphingomonas*, *Stenotrophomonas*, *Rhodococcus*, *Bacillus* (Qureshi et al., 2007, Tripathi et al., 2011, Verma et al., 2011, Sreenivasulu et al., 2012, Ghosh et al., 2017) etc. have been reported for bioremediation of various contaminated sites.

One strategy of bioremediation is phytoremediation that involves plant-microbe interaction and by which sustainable growth of plants in the contaminated sites could be restored. Microbes-soil-plant interactions for *in-situ* treatment of pollutants in contaminated sites have been reported as Microbe-Assisted-Phytoremediation (MAP) technology (Juwarkar, 2012). MAP technology has been reported to be ecofriendly, economic and effective for restoration of degraded land such as mines, spoil dumps (Juwarkar, 2012). The report has also illustrated the application of MAP technologies at sites where shallow contamination or pollution exist. It has been reported that, at shallow pollution sites, processes such as phytotransformation, rhizosphere bioremediation, phytostabilization, phytoextraction and/or rhizofiltration were best possible to be implemented (Juwarkar, 2012). Industries like textile mills, tanning- leather manufacturing industries, fertilizer factories, mining industries were major sources of contamination where phytoremediation technologies have been promising. To evaluate the potential applications of the plant-microbe interactions remedial strategies, studies have been performed by researchers (Niti et al., 2013, Sarkar et al., 2010).

Biodegradation and Bioremediation of Chloro/Nitro/Amino Substituted Aromatic Compounds

Aromatic compounds namely chlorophenols, p-

nitrophenol (PNP), aminophenols, chloronitrophenols and their derivatives are important environmental pollutants because of their toxicity to many living organisms (Arora and Jain 2012, Arora et al., 2014, Kanekar et al., 2014, Sengupta et al., 2015, Kumari et al., 2017).

Both aerobic and anaerobic biodegradation pathways of aromatic compounds and their derivatives have been deciphered by researchers (Arora et al., 2014, Malik et al., 2014, Tiwari et al., 2017, Ghosh et al., 2017); more extensively the aerobic degradation. In bacteria, major pathways reported for catabolism of aromatic compounds have revealed that initial conversion steps were carried out by different enzymes, but the compounds were transformed into a limited number of metabolites, such as protocatechuate and catechols. These intermediate metabolites were channeled into central metabolic routes via; different ring cleavage pathways enzymes. This generalized scheme of catabolic pathways for aromatic compounds suggests that microorganisms have extended their substrate range by developing “peripheral” enzymes, which were able to transform initial substrates into one of the central intermediates like catechol, resorcinol, benzoquinones etc.

Arora et al. 2014 have reported about bacterial degradation of the toxic compound 4-chloro-3-nitrophenol (4C3NP) and formation of 4-chlororesorcinol intermediate in the degradation pathway. 2-chloro-4-aminophenol degradation by bacteria has also been reported by the same group (Arora et al., 2014, Malik et al., 2014). 4C3NP-mineralizing bacterium, *Pseudomonas* sp. JHN was isolated, by enrichment, from a wastewater sample collected from a chemically-contaminated area in India. Detoxification of 2-chloro-4-nitrophenol (2C4NP) by *Cupriavidus* strain a3 has been reported by Tiwari et al. 2017. These microbes show potential for application in bioremediation. Microbial Biotechnology Research Laboratory (MBRL) reported a bacterium isolated from limestone deposits in Ukhrul, Manipur, India; two strains, (HS4-2 and HS6-1) were isolated as p-nitrophenol (PNP) degraders via hydroquinone and –nitrocatechol pathways (Ningthoujam et al., 2012). In PNP degradative pathway, 4-nitrocatechol and benzenetriol intermediates were mainly formed in Gram negative organism (Chauhan et al., 2010, Qureshi et al., 2002,

Qureshi *et al.*, 2001). Enzyme p-nitrophenol 4-hydroxylase has been reported to be active for parathion biotransformation and further degradation of PNP (Khan *et al.*, 2011, Verma *et al.*, 2011, Wasi *et al.*, 2011, Saha *et al.*, 2014, Goyal and Bansiwala, 2017). Polyphenol oxidase (PPO) enzyme for phenolic contaminants from industrial wastewater has been reported (Mukherjee *et al.*, 2013). Other enzymes such as catechol 1,2-dioxygenase (C12O) and catechol 2,3-dioxygenase (C23O) have been reported to enhance bioremediation process under different conditions (Pandeeti and Siddavattam, 2011, Kumari *et al.*, 2017, Ghosh *et al.*, 2017). Nitroaromatic pollutants trinitrotoluene (TNT) and dinitrotoluene (DNT) degradation has mainly been reported under aerobic conditions (Goel *et al.*, 2012). Also, other pollutants such as 4-nitrobenzoate, 2-nitrobenzoate, 2-nitrotoluene, nitrobenzene, p-nitroaniline and p-nitrophenol were reported to be used as growth substrates by bacteria (Mulla *et al.*, 2011, Srivastava *et al.*, 2013, Qureshi *et al.*, 2001, Qureshi *et al.*, 2002, Mishra and Sardar, 2014, Arora and Sharma, 2015). Under anaerobic condition, nitroaromatic pollutant reduction and biotransformation have been reported by bacteria and their nanobiocomposite as catalyst (Srivastava *et al.*, 2013, Chaudhary *et al.*, 2013).

Biodegradation and Bioremediation of Organochlorines and Organophosphates

Organochlorines viz., DDT- type compounds, cyclodienes like aldrin, dieldrin, endrin, heptachlor, chlordane, endosulphan, lindane, polychlorinated biphenyls etc, organophosphates such as methyl parathion and many chlorinated/nitro/amino-substituted compounds employed as insecticides and pesticides have also been categorized as pollutants. Recently it has been assessed in the Vasai Creek water near Mumbai that organochlorine and organophosphorus pesticide residues were major cause of endocrine disruption leading to ecological risk (Singare, 2016).

Over the past 20 years of organochlorine insecticide lindane production the industry has utilized inappropriate storage and disposal practices, resulting in widespread environmental hexachlorocyclohexane (HCH) contamination and a large number of HCH dumpsites (Jit *et al.*, 2011). One of the HCH dumpsites at Ummari village, Lucknow has been studied for characterization of the microbial diversity

by culture dependent approach (Singh *et al.*, 2009, Dadhwal *et al.*, 2009, Bala *et al.*, 2010, Sharma *et al.*, 2010, Kumar *et al.*, 2013, Negi *et al.*, 2014, Verma *et al.*, 2014). These studies revealed the presence of potent HCH degraders and tolerant bacterial species that arose under the HCH stress. Specifically, many of these bacteria shared components of a unique catabolic system –named the *lin* system which allowed for the degradation of HCH isomers (Kumari *et al.*, 2002, Sharma *et al.*, 2006, Lal *et al.*, 2010). These *lin* genes seem to have played a major role in shaping the degradation of xenobiotic compounds (Dogra *et al.*, 2004) and are diverging to form several catabolic functions (Lal *et al.*, 2006). *linA* (HCH dehydrochlorinase) and *linB* (halohydrolyase) act as the primary enzymes in the pathway. The *linA* GC content is lower than that of the *linXBCD* genes (Suar *et al.*, 2005) *linB* acts on β -HCH, δ -HCH and pentachlorocyclohexanol, yielding a 2,3,4,5,6-pentachlorocyclohexanol (2,3,4,5,6-PCHL), and tetrachlorocyclohexanediol, respectively (Sharma *et al.*, 2006). *LinB* protein has a very broad substrate preference for halogenated compounds whereas the substrate range for *LinA* protein is very restricted to α -, γ - and δ -HCH and their corresponding PCCH products (Kumari *et al.*, 2002). It is due to HCH pressure that *LinA* proteins are evolving to a greater extent at the dumpsite. *LinB*, which is evolutionary more advanced and evolved, is evenly distributed at all the contaminated sites (Kumari *et al.*, 2002, Sharma *et al.*, 2006). It has been found that the upper pathway *lin* genes are constitutively expressed whereas the lower pathway genes are primarily induced in the presence of α - and γ -HCH not β - and δ -HCH (Suar *et al.*, 2004). The high residue level is not the only factor: other biological factors like HGT by IS elements and plasmids may be responsible for the high copy number. Bioremediation of HCH contaminated sites is being attempted by bioaugmentation, biostimulation and enzymatic bioremediation approaches (Lal *et al.*, 2010). Commonly used pesticides namely chlorpyrifos have been reported to be utilized by *Pseudomonas* strain C2A1 as the sole source of carbon and energy (Anwar *et al.*, 2009, Gupta *et al.*, 2016). Degradation of chlorpyrifos have been demonstrated in soil suspension by Das and Adhya 2015. Also reports showed that the capability of microorganism for degradation could be enhanced by characterization and transfer of

pesticide-degrading genes, induction of catabolic pathways and display of cell surface enzymes (Hussain *et al.*, 2009). Recently researchers are trying to explore *Pseudomonas* sp to degrade chlorinated pesticides belonging to different chemical categories (Gupta *et al.*, 2016). Endosulphan degradative pathways and the generated metabolites have also been reported (Bajaj *et al.*, 2010).

Many reports are available on organophosphate (OP) pesticides biodegradation by bacteria (Qureshi *et al.*, 2009; Dubey and Fulekar 2012; Sasikala *et al.*, 2012; Iyer *et al.*, 2013; Pinjari *et al.*, 2013; Kundu *et al.*, 2014; Reddy *et al.*, 2016; Chatterjee *et al.*, 2017). Even, several novel bacterial strains have been reported from agricultural field especially located at Narigram in Burdwan district of West Bengal, India. *Bacillus aryabhattai*, a novel endospore forming strain have been reported to tolerate high concentration (up to 500 $\mu\text{g mL}^{-1}$) of pesticide parathion (Pailan *et al.*, 2015). Study on bioremediation of methyl parathion using *Pseudomonas* sp have also been reported from Visakhapatnam, India (Mulla *et al.*, 2011, Srivastava *et al.*, 2013, Begum and Arundhati, 2016). OP degrading enzymes have been designated as Phosphotriesterases (PTEs) as they are involved in hydrolysis of triester linkage found in OP compounds. The PTEs are classified into three major groups, i.e. organophosphate hydrolases (OPH), methyl parathion hydrolases (MPH) and organophosphate acid anhydases (OPAA). The OPAA is later shown to be dipeptidase and its activity on OP compounds is shown due to structural similarities between OP compounds and dipeptides having proline at the C-terminus. However, the physiological substrates for OPH and MPH are unknown. Considering the catalytic efficiency and structural similarities with the structure of quorum quenching lactonases, the OPH, the product of organophosphate degrading (*opd*) gene, is assumed to have evolved from phosphotriesterase like lactonases (PTLL). The genetic organization of *opd* gene in both *Flavobacterium* sp. and in *Pseudomonas diminuta* have been elucidated. These two strains have been reclassified and renamed as *Sphingobium fuliginis* ATCC 27551 and *Sphingopyxis wildii* (Parthasarathy *et al.*, 2016a). In these two soil isolates the *opd* gene is found in large indigenous plasmids. The 65 kb plasmid pCMS1, isolated from *Sphingopyxis wildii* has been shown to be a self-transmissible plasmid and is involved in

spreading *opd* information among soil microflora (Pandeeti *et al.*, 2011). Similarly, the plasmid pPLD2 is shown to be an Integrated Mobilizable Element (IME) (Pandeeti *et al.*, 2013). In plasmid pPDL2, the *opd* along with an ORF, orf306 is flanked by mobile genetic elements (Pandeeti *et al.*, 2013). The Orf306 is an esterase. When it is expressed in *E. coli*, the cells have started degrading p-nitrophenol, the recalcitrant degradation product generated during OPH mediated *hydrolysis* of OP insecticides. Transcriptome analysis for the *E. coli* strains grown in presence and absence of Orf306. The Orf306, in an unknown way, is shown to down regulated glycolysis and TCA cycle and upregulated both *hca* and *mhp* operons involved in degradation of phenylpropionate and hydroxylphenyl propionate. The enzymes encoded by *hca* and *mhp* operons are shown to degrade p-nitrophenol (Chakka *et al.*, 2015). The IME borne *opd* product (OPH) along with Orf306 is shown to contribute for mineralization of insecticide residues found in soil by stimulating the innate nitrophenol degrading ability of recipient strains. The work done has also shown OPH association with membrane associated transport system involved in phosphate transport. The genetic and biochemical evidences suggest that OPH is involved in phosphate acquisition in soil bacteria (Parthasarathy *et al.*, 2016b).

Biodegradation and Bioremediation of Hydrocarbons and Crude Oil

Crude oil and hydrocarbons comprises of saturates/paraffins, aromatics, resins and asphaltenes. Scientific communities are being urged to develop strategies to minimize the concentrations polyaromatic hydrocarbons (PAHs) from the Gujrat coastlines and restore sites by investigations (Dudhagara *et al.*, 2016). Reports are available consistently on metabolic diversity and versatility of pure cultures and mixed cultures respectively to utilize hydrocarbon pollutants in petroleum crude as carbon source (Varjani and Upasani, 2016). It has been reported that even co-culture(s) of bacteria and fungi enhance degradation rates of diesel oil and PAHs under laboratory conditions (Varjani and Upasani, 2013).

Bioremediation technologies have been reported to be applied to restore petroleum hydrocarbon polluted environments that makes use of natural microbial

biodegradation activity (Varjani, 2017). Both in-situ and on-site treatment process have been reported to breakdown crude petroleum oil components at contaminated sites. In 2007, Das and Mukherjee have studied and compared the efficiency of thermophilic *Bacillus subtilis* DM04 strain and *Pseudomonas aeruginosa* M bacteria isolated from North East India for biodegradation of crude petroleum oil.

Bacillus spp. and *Pseudomonas* have been found suitable for practical field applications for effective PAH and aromatic compounds bioremediation at contaminated sites (Khanna *et al.*, 2011, Sood *et al.*, 2010, Phale *et al.*, 2013, Phale *et al.*, 2007). Even high-altitude bacteria have been reported to be capable of degrading engine oil (Jain *et al.*, 2010). Recently, *B. licheniformis* CFR1 is reported as an excellent candidate for bioremediation and detoxification of aflatoxin (Rao *et al.*, 2017). Novel, thermally stable, halotolerant enzymes like cellulases have been reported to perform in harsh conditions such as high salt, heat and acidic environments (Chandel *et al.*, 2007, Garg *et al.*, 2016).

Bacterial community found in an Indian coal bed has been reported to be capable of in-situ biotransformation of coal into methane (Singh *et al.*, 2012, Singh and Tripathi 2013, Mohanty *et al.*, 2016). Such findings would assist in bioremediation of such contaminated ecosystems.

Recently it has been reported that oleophilic microorganisms could be used for oil spill bioremediation (Varjani, 2017). Scientific communities are being urged to develop strategies to minimize the concentrations of pollutants for examples PAHs from the Gujrat coastlines and restore sites by investigations (Dudhagara *et al.*, 2016). In the similar lines, there is a big break through achieved, in the area of remediation of crude oil contaminated sites, where the technology "Oilzapper" has been successfully demonstrated at commercial scale at various site and received global acceptance (Sharma *et al.*, 2019).

Biodegradation and Bioremediation of Metals and Heavy Metals

Toxicity of metals to ecosystems and their mobility in biological systems needs to be highlighted in Indian scenario. In 2014 Mani and Kumar have reviewed

the bioremediation options especially using microorganisms and plants with eco-friendly and sustainable approach in Indian context. Reports have been available for metal binding proteins or peptides for their potential towards tolerance and accumulation in living systems especially with cadmium which is found significantly in foods (Mejare and Bulow, 2001). Molecular level studies of metal tolerance genes in microorganisms and hyper accumulator plants showed involvement of functional genes for detoxification (Dixit *et al.*, 2015). Also, bioabsorbent such as fungi viz; *Phanerochaete chrysosporium*, *Aspegillus awamori*, *Aspergillus flavus*, *Trichoderma viride* are reported for removal of heavy metals from industrial effluent by Joshi *et al.*, 2011. Fungi such as *Trichoderma* with plant growth promoting (PGP) abilities have been reported as efficient biocatalyst and potent antimicrobial agents (Mishra *et al.*, 2014, Monaharachary *et al.*, 2014, Singh *et al.*, 2014). Removal of metal(loids) using organic amendments such as compost, MSW as nutrients and soil conditioners have been discussed with practical implications to bioavailability by Park *et al.*, 2011. Role of EPS from various sources with special emphasis on bioremediation of heavy metals have been studied by researchers (Pal and Paul 2008, Ahemad 2012). The applicability of chromium resistant fungal species viz., *Micrococcus* sp and *Aspergillus* sp for removal of chromium and nickel up to 10,000 mg/L have been reported (Congeevaram *et al.*, 2007). The potential of bacterial genera (*Pseudomonas*, *Enterobacter*, *Bacillus*, *E.coli*), yeast and fungal species for bioremediation of metal toxicity have been discussed and their impact on environment is also illustrated by Ray and Ray 2009. The mechanism of removal of metal by biological means have been reported where it was found that Pi precipitates the heavy metals as phosphate-metal complexes (Chaudhari *et al.*, 2017). Bioremediation of electroplating industrial wastewater with simultaneous nanoparticles synthesis also have been reported (Sathyavathi *et al.*, 2014, Dasgupta *et al.*, 2015). Several biological methods of heavy metal remediation have been reported by both bacteria and fungi (Kundu and Gupta 2007, Singh and Prasad 2015, Chaudhary *et al.*, 2017, Nagvenkar and Ramaiah 2010).

Treatment of Textiles Dyes

Ichalkaranji and nearby places in Maharashtra are

well known as the textile hub of the state. A large number of large, small and even household dye processors are located in this part. Because of the uncontrolled discharge of the toxic textile dye-stuff in environmental sink, water bodies and soils in the said area have become heavily polluted. The Department of Biochemistry, Shivaji University research group lead by Prof. S. P. Govindwar has been actively involved to find out novel and developed ecofriendly bioremediation strategies for the treatment of these noxious dyes.

In the last five years, a number of bacteria such as *Lysinibacillus* sp. RGS, *Providencia rettgeri* strain HSL1, *Pseudomonas* sp. SUK1, *Bacillus thuringiensis*, *Pseudomonas* sp. LBC1, *Pseudomonas monteilii* ANK, *Pseudomonas putida* PgH, *Brevibacillus laterosporus*, *Bacillus cereus* EBT1 and *Kocuria rosea* MTCC 1532 etc. have been explored for the treatment of textile dyes. Fungal species namely, *Galactomyces geotrichum* MTCC 1360 and *Aspergillus ochraceus* NCIM-1146 were also utilized for detoxification of dyes (Saratale et al., 2011). Novel lichen species called *Permeliaper lata* was also found to treat the textile dyes to significant extents (Kulkarni et al., 2014). Combinatorial systems of bacteria and fungi such as *Galactomyces geotrichum* MTCC 1360-*Brevibacillus laterosporus* MTCC 2298 and *Aspergillus ochraceus* NCIM-1146-*Pseudomonas* sp. SUK1 was observed to be more efficient than the individual cultures in treatment and decolorization of dyes. Of late, plant and bacterial hybrid systems of *Zinnia angustifolia*-*Exiguobacterium aestaurii*, *Portulaca grandiflora*-*Pseudomonas putida* and *Pogonatherum crinitum*-*Bacillus pumilus* were also explored in the constructed wetlands systems for efficient removal of textile dyes from real effluents (Khandare and Govindwar, 2015). A novel bioreactor with *Ipomoea hederifolia* adventitious roots and its endophyte *Cladosporium cladosporioides* was developed for textile dye degradation (Patil et al., 2016). A variety of bioreactors using bacteria were also developed for the treatment of dyes from effluents. The decolorization of real textile industry effluent using immobilized bacterial yeast consortium of *Galactomyces geotrichum* MTCC-*Brevibacillus laterosporus* MTCC in a newly developed triple layered fixed bed bioreactor and the efficiency and performance was tested in continuous operation

(Kurade et al., 2017). Degradation and detoxification of methylene blue dye adsorbed on water hyacinth in semi continuous anaerobic-aerobic bioreactors was also carried out in a novel microbial consortium of *Sacharromyces cerevisiae* and *Bacillus* sp. STIS (Bedekar et al., 2015). Solid state fermentation of the bioadsorbed textile dyes stuffs and distillery industry waste-yeast biomass has been successfully carried out using *Bacillus cereus* EBT1 (Kadam et al., 2013). In independent studies, influence of diazo dyes on soil bacterial population was monitored. Molecular analysis of shift in community after different time intervals was monitored by using PCR-DGGE targeting V3 region of bacterial 16S rRNA gene. UV irradiation was used to introduce random mutations in *Pseudomonas* sp. LBC1. Genetic alterations induced by UV irradiation in selected mutant bacteria were confirmed by random amplification of polymorphic DNA technique. The mutant bacteria named as *Pseudomonas* sp. The dye treatment efficiency of the mutant was found to be improved after mutagenesis (Joshi et al., 2013). Different oxidoreductive microbial enzymes like lignin peroxidase, manganese peroxidase, aryl alcohol oxidase, laccases, tyrosinase, azo reductase, malachite green reductase, DCIP reductase and riboflavin reductase attacks the complex dyes structure and metabolize them through steps like deamination, desulfonation, dehalogenation, hydroxylation, oxidation, reduction, ring opening and cleavage mechanisms. The direct roles of these enzymes have been proved using standard purification protocols (Telke et al., 2015). The metabolites of dye degradation were analyzed by using UV-Visible and FTIR spectroscopy, HPLC, HPTLC, GC and/or LC-MS and NMR techniques. Textile effluents generally have very high COD, BOD, TOC, TDS, TSS, alkalinity, conductivity etc., therefore the microbes which can only survive these stressful conditions could only show dye removal efficacies. The toxicity assessment of dyes, effluents and their metabolites after bioremediation have been carried out using plant, microbes and animal models looking for cytotoxicity and genotoxicity. An intensive research on utilization of bacteria at the large industrial scales is underway and an efficient reactor set up is still awaited.

Effluents generated from the textile mills are one of the major sources of point pollution in rivers. In view of the requirement of color free discharge and detoxified waters, several lines of treatment have

been tested. These include conventional physical as well as advanced biological oxidation processes such as treatment with TiO_2 , UV, Fenton and photo-Fenton oxidations. Many of these do not work effectively on complex azo and pthalocyanine dyes. Laccases and several other oxidoreductases have been identified as promising candidates for decolouration as well as detoxification. The group of Prof. Saroj Mishra uses laccase from *Cyathusbulleri* (Salony *et al.*, 2006) effectively for decolouration of several category of dyes. The reaction centre of this enzyme is made from closely spaced Cu atoms arranged in the form of three reaction centres, namely T1, T2 and T3. Out of these, the geometry at the T1 Cu centre determines the redox potential of the enzyme and its ability to oxidize dyes of high redox potential (Salony *et al.*, 2008). The property of laccase to act on complex dyes as well as real effluent can be further augmented by addition of small mediators in the reaction mixture, protein engineering, supplementation with other powerful oxidoreductase enzymes such as cellobiose dehydrogenases or by producing laccase on complex lignocellulosics such as wheat bran. These approaches have been successfully used in the lab. Amongst a number of mediators, 2,2'-azino-bis (3-ethyl-benzothiazoline-6-sulphonic acid) or ABTS and natural mediator vanillin were found to be very effective (Chhabra *et al.*, 2008) against a wide range of dyes as well as combined effluent collected from textile mills. Based on these findings, a continuous membrane reactor was designed and successfully operated for decolouration of real effluent (Chhabra *et al.*, 2009) using laccase and a synthetic mediator. The treatment with laccase was extended to complex effluents by combining a chemical step (alum treatment) during the early processing stage (Chhabra *et al.*, 2015a). Operationally, the system was equally effective in handling a variety of Reactive dyes (Chhabra *et al.*, 2015b). In the second approach, random mutagenesis was carried out of the C-terminal half of the coding gene (Garg *et al.*, 2012; Kenzom *et al.*, 2014) and mutant library (Kenzom *et al.*, 2015) screened for catalytically superior enzymes. Many of these were effective on pthalocyanine dyes as well. Through biochemical and model building, amino acid residues were identified that affect catalytic activity. (In another approach, combination of laccase with cellobiose dehydrogenase was found to be highly active on pthalocyanine dyes (Bashir *et al.*, 2015;

Gangwar *et al.*, 2016) with extensive degradation noticed of this category of dyes. The treatment on real effluent indicated this combination of enzymes (wild type and the mutant ones) to be effective for both decolouration and reduction in phytotoxicity. Several other mutants available in the author's laboratory are being analysed to identify sites in the enzyme that determine the redox potential as well as the electron transfer property of laccase. The efficacy of laccases produced on complex lignocellulosics was recently demonstrated on complex effluent from denim dyeing industry (Vats and Mishra, 2017) indicating these to be promising in treatment of textile wastewaters.

Bioremediation of Soil Salinity

Papers published in Nature in 1950s claimed saline soil reclamation using native populations of cyanobacteria, based on which ICAR advocated use of these microbes for such purpose. Dr. Apte's laboratory demonstrated that actually sodium exclusion and its entrapment in extracellular polysaccharides underlie cyanobacterial tolerance to salinity (Apte and Thomas, 1986). While this provides temporary relief to a growing crop by transient sequestration of sodium/chloride ions and by supplementing fixed nitrogen, the technology offers no permanent solution to the problem of soil salinity which is restored subsequent to death and decay of cyanobacteria (Apte and Thomas, 1997). The work also revealed entirely different mechanisms to be involved in salinity and desiccation tolerance (Apte, 2001). Agronomic practices which curtailed sodium influx enhanced salt tolerance (Apte *et al.*, 1987). Desiccation tolerance, in contrast, depended on de novo synthesis of osmotic stress proteins (Fernandes *et al.*, 2000).

Integrating Conventional Bioremediation Technology with Genomics

Over the years it has been learnt that for successful implementation of biological remediation technologies, multidisciplinary approach from experts of various areas are required. Moreover, for effective bioremediation, complete understandings of environment (i.e. physical, chemical and biological properties) to be treated are essential. With the advancement in the technology, our lab has integrated the conventional approach with genomics-driven strategies to make bioremediation technologies more

potent. We perused the hypothesis that microbes are the best answer for counteracting the devastating consequences of our activities on environment. And to demonstrate the viability of our proposition, we have developed and enriched more than 100 bacterial consortia (mixed cultures) and isolated bacterial pure cultures from polluted soil, sediments, water, industrial and CETP effluents exhibiting significant degradation potential for various dyes and dye intermediates, aromatics, polyaromatic hydrocarbons (PAH), distillery spent wash and detoxification of heavy metals under different environmental conditions (anaerobic, microaerophilic, aerobic, sequential microaerophilic-aerobic and anaerobic-microaerophilic) (Patel *et al.*, 2012a and b; Jain *et al.*, 2012; Shah and Madamwar, 2013a; Patel and Madamwar, 2013; Balapure *et al.*, 2014; Chattaraj *et al.*, 2016a; Shah *et al.*, 2016a). The flask scale results were reevaluated during scaling up process in various bioreactors (anaerobic biphasic fixed film, sequential anoxic-oxic batch process, sequential anaerobic-microaerophilic process, periodic discontinuous batch operation) and processes were optimized for simulated and real industrial effluents (Acharya *et al.*, 2011; Shah *et al.*, 2016b; Chattaraj *et al.*, 2016b; Venkata *et al.*, 2012; Balapure *et al.*, 2016). It is imperative to know the degradation mechanism of the xenobiotic compounds to make technology more robust. Our studies show that due to consorted metabolic activities of the consortia complex aromatic compounds such as dyes, PAH are gradually degraded into simpler intermediates through generating different intermediates and finally led to the complete mineralization via TCA cycle. With two different consortia the degradation pathways for degradation of reactive azo dye Remazol Brilliant Violet 5R (i.e. Reactive Violet 5R) was proposed and observed that the dye was metabolized through formation of α -naphthol and benzene sulphoate (Jain *et al.*, 2012; Shah *et al.*, 2016a). In PAH degradation (phenanthrene), the study revealed that the compound was degraded through phthalic acid-protocatechuate acid pathway (Patel and Madamwar, 2013). During azo dye degradation the initial step is the reduction of azo bonds ($-N=N-$), catalyzed by azoreductase. One of such azoreductases was cloned from consortia V9 of 537 bp containing an ORF of 178 amino acids showing optimum activity in the presence of NADPH. The host harboring azoreductase clone showed 90% azoreduction within 7 min in cell free extracts (Shah

and Madamwar, 2013a). On the course of developing feasible bioremediation strategy, whole genome of three bacterial cultures, *Paenibacillus* sp. strain DMB20, *Achromobacter* sp. strain DMS1 and *Paenibacillus* sp. strain DMB5 (showing <97% similarity of phylogenetic marker with available database) was sequenced. Annotation of these genomes revealed the presence of various genes required for aromatic degradation, antibiotic and heavy metal resistance, towards stress response in all three strains, which signifies their active role in xenobiosis (Shah *et al.*, 2015; Amin *et al.*, 2015; Johnson *et al.*, 2016). The complete degradation pathway for benzonitrile, benzamide, benzoate, and catechol degradation was mapped in *Paenibacillus* sp. strain DMB5 (Johnson *et al.*, 2016). Besides whole genome, we have sequenced more than 10 metagenomes from different polluted sites across the 'Golden Industrial Corridor' of Gujarat State, corresponding to about 8 Gb base pairs. Nearly 100,000 reads corresponding to >300 enzymes and enzymatic pathways potentially involved in biodegradation of xenobiotic compounds were mapped. Complete and partial degradation pathways for several xenobiotic compounds were mapped (Shah *et al.*, 2013b). The diversity community shift due to persistent pollution at certain polluted sites was also evidently observed (Patel *et al.*, 2014, 2015). The work describes the potential of bacterial system to utilize the recalcitrant xenobiotics and community organization at polluted environment along with their functional capabilities by knowing their genetic composition. Nonetheless, it is a preliminary study for complete understanding of entire ecology/ecosystem of polluted environment to improve the natural bioremediation.

Consortium Development in Bioremediation

Remediation of contaminated resources is one of the major thrust areas of the National Environmental Engineering Research Institute (NEERI). Since, bioprocesses play key role in remediation, especially when the contamination is of organic nature, major thrust was taken in the area of optimization of such processes (Purohit *et al.*, 2003a, Purohit *et al.*, 2003b, Moharikar *et al.*, 2005, Kapley and Purohit, 2009). Bioremediation processes are employed either for bio-stimulation or for bio-augmentation (Khardenavis *et al.*, 2008, Khardenavis *et al.*, 2010, Paliwal *et al.*, 2012, Purohit *et al.*, 2016). The strategy involved in

this program is to enrich and select bacteria, which have the potential to utilize target pollutants as carbon source for their growth. The selected pollutants were hydrocarbon, -nitro and -chloro substituted pesticides and intermediates used for synthesis of pesticides and pharma products. The studies over a period of 10 years a bank of isolates was established from different environment with wide catabolic potential (Selvakumaran *et al.*, 2008; Thangraj *et al.*, 2008, Verma *et al.*, 2011). Since, most of these contaminated sites show a multi-substrate challenge, the defined mixtures of bacteria with selected genotype were used to design consortia with flexible degradative capacities (Selvakumaran *et al.*, 2011).

The pollutant targeted in the program included the hydrocarbons from crude oil (aromatics and long chain aliphatic), nitro and -chloro substituted aromatics or heterocyclics. Enrichment techniques were designed to isolate for these difficult to degrade molecules. One of the major achievements was the development of a consortium of four bacteria, which could emulsify and degrade the crude oil efficiently in 12 h. These cultures were used for development of wastewater treatment process (Moharikar *et al.*, 2005, Kapley *et al.*, 2007, Domde *et al.*, 2007). Another area of application was the remediation of contaminated soil. Initially controlled microcosm experiments were planned using designed bacterial consortia, which were further evaluated using pilot scale field trials; and demonstrated to user agencies (Qureshi *et al.*, 2009). Different bioremediation processes were designed either by using bio-stimulation (balancing of nutrients or induction of degradative capacities using inducer molecules) or by bio-augmentation (applying the required bacteria to bring out the degradation). These studies were supported by characterization of the associated genotypes that were predicted by annotation of genome sequence data and revealed the basic mechanism of degradation pathways for hydrocarbons, p-nitrophenol and atrazine (Tikharia *et al.*, 2016).

The biostimulation to enhance the expression of degradative capacities was carried out using the intermediate of the metabolism such as catechol, salicylate and even phenol and analyzed for different aromatic ring cleavage enzymes (Selvakaumaran *et al.*, 2011). Similarly, the metal ions were also used

while the sludge was recycled for enhancing the required catabolic potential to microbial community. The other issues is for difficult to degrade wastewaters such as dye containing waste for that consortiums has been prepared (Dafale *et al.*, 2010) and also to solve the issue of generated increase in ammonia level has been dealt with specialized organisms (Khardenavis *et al.*, 2007; Pal *et al.*, 2015).

Engineering Microbes Towards Bioremediation

Biodegradative potential of degraders can overlap with bacteria from unrelated genera, or alternatively, genes with completely different sequences can catalyze the same reaction. Understanding these complex reactions requires a multi-dimensional approach of whole genome sequencing and bioinformatics analysis of sequence data (Sagarkar *et al.*, 2016). Next generation sequencing technologies allows us to explore the complete catabolic profile of the isolate and this tool has been used to identify many multi-substrate degrading bacteria used in soil microcosm/mesocosm studies. Taking this further into in situ bioremediation of contaminated soil is more complex. Environmental scenarios very often involve more than one contaminant and the microbial community containing different metabolic pathways is governed by diverse soil types and characteristics, climate conditions and anthropogenic activities. These shortcomings are now being addressed via the systems biology approach that analyzes interactions between microbes and their environment and helps predict their function and survival (Chakraborty *et al.*, 2012).

Mapping the microbial populations of a contaminated niche allows for a systematic development of a bioremediation strategy. The metagenomics tool provides insight into the phylogenetic composition of the microbial community as well as its functional genes (Jadega *et al.*, 2014; Yadav *et al.*, 2014). Shifts in microbial community of a wastewater treatment plant have been demonstrated to correlate with performance efficiency (Kapley *et al.*, 2015). Besides analyzing the community function, the new tools generate information on the structure and function of catabolic enzymes, the actual unit carrying out the biodegradation process. Knowledge of their detailed structure offers options for improvement by site directed mutagenesis or cloning

that can improve the efficiency of the reaction or provide the degradative capacity where it did not exist (Bassalo *et al.*, 2016). More recently, researchers have engineered metabolic pathways in microorganisms to generate engineered biocatalysts. Despite the evolution of tools and increased knowledge database, the field application of bioremediation progresses slowly and more studies need to be directed towards the survival and performance efficiency of target microbes in field conditions.

Engineering Cyanobacteria for Enhanced Nitrogen Biofertilizer Potential in Stressful Agricultural Environments

Discoveries in Dr. Apte's laboratory that stress-responsive gene expression in cyanobacteria has a generic stress response and a stress-specific response (Apte and Bhagwat, 1989; Bhagwat and Apte, 1989; Alahari *et al.*, 2001; Alahari and Apte, 2004), led to discoveries of genes important for survival under major agricultural stresses, such as salinity and heat-shock (Rajaram and Apte, 2003 and 2010; Rajaram *et al.*, 2014), both of which diminish nitrogen fixation. Several of these genes were cloned using the versatile approach of differential RNA hybridization developed by him (Apte and Haselkorn, 1990) or by cloning of stress-specific genes (Rajaram *et al.*, 2001). Mutagenesis and/or overexpression of these genes established their distinct role in general/specific stress tolerance (Chaurasia and Apte, 2009; Rajaram and Apte, 2008). The genes were engineered in to nitrogen-fixing *Anabaena* strains and expressed from strong light-inducible *psbA1* promoter to further enhance the inherent stress tolerance of these strains. An eco-friendly, integrative expression vector was developed for *Anabaena* for the first time and used to construct recombinant strains with enhanced nitrogen biofertilizer potential in stressful environments (Chaurasia and Apte, 2011).

Engineering Cyanobacteria For Oxidative Stress Tolerance

Realization from Dr. Apte's work that oxidative stress is central to all the abiotic stresses, led to the identification of oxidative stress responsive genes and proteins using genomic, transcriptomic and proteomic approaches, in three *Anabaena* strains typical of Indian paddy fields (Panda *et al.*, 2014 and 2015).

Biology of several oxidative stress alleviators, such as MnSOD, FeSOD, Mn-Catalases, and 4 peroxiredoxins unique to cyanobacteria was elucidated from his laboratory (Raghavan *et al.*, 2011, 2013 and 2015; Bannerjee *et al.*, 2012a and 2012b). All the corresponding genes and their variants were also over-expressed in *Anabaena* to demonstrate enhanced oxidative stress tolerance as also tolerance to other environmental stresses (Bannerjee *et al.*, 2013).

Engineering *Deinococcus radiodurans* for Enhanced Oxidative Stress Tolerance and Uranium Bioremediation

Dr. Apte's laboratory also took the lead in unravelling the molecular basis of extreme radioresistance in *Deinococcus radiodurans* and contribution of its DNA repair and oxidative stress tolerance abilities to this phenomenon in particular (Misra *et al.*, 2004 and 2006). A novel molecule pyrroloquinoline quinone (PQQ) was characterized from this microbe and shown to play very important role in radioprotection *in vitro* and *in vivo* in *D. radiodurans* (Misra *et al.*, 2004). Transgenesis of PQQ synthase gene was also shown to enhance oxidative stress tolerance of *E. coli* significantly (Khairnar *et al.*, 2003). PQQ was also shown to play a role in radiation-induced signalling and induction of DNA repair through a periplasmic protein kinase (Khairnar *et al.*, 2007). *D. radiodurans* was found to survive very high oxidative stress triggered by exposure to tellurite. Using genomic and proteomic approaches its tolerance to radiation was shown to be superior to tellurite induced oxidative stress, though the two stress responses shared a lot of common genes and proteins (Narasimha *et al.*, 2015).

On account of its radioresistance, *D. radiodurans* is an automatic choice for bioremediation of nuclear waste. Acid and alkaline phosphatase genes (*phoN* and *phoK*, respectively) were engineered into this microbe to construct strains that could bioprecipitate uranium from low concentrations of 0.5 mM up to 10mM over a pH range of 5-9, resulting in impressive uranium loading of up to 7-10 mg/g dry biomass (Appukuttan *et al.*, 2006; Nilgiriwala *et al.*, 2008 and 2009; Kulkarni *et al.*, 2013). The phosphatase activity and uranium bioprecipitation capabilities were retained up to 2 years in lyophilized cells (Seetharam *et al.*, 2009;

Appukuttan *et al.*, 2011). At acidic pH precipitated U was cell bound and easy to recover by settling and decantation, while at alkaline pH the extracellular precipitate could be immobilized in to columns and beads for online Uranium remediation in a flow through system and facilitated complete and easy recovery of U (Seetharam-Misra *et al.*, 2012; Kulkarni *et al.*, 2013 and 2016). Methods were also developed for surface display of bioremediation-active proteins in *D. radiodurans* by tagging them to surface layer proteins (Seetharam-Misra *et al.*, 2014). Radiation responsive promoters have been characterized from this microbe (Ujaoney *et al.*, 2010; Narasimha *et al.*, 2016 and 2017) and used to express phosphatases to further enhance U bioremediation in high radiation environments (Seetharam-Misra *et al.*, 2014). The bacterium has also been engineered to express plant *NiCoT* genes for purpose of removal of cobalt from waste solutions (Gogada *et al.*, 2015).

Developing Cyanobacteria for Uranium Bioremediation

Uranium resources on land are getting depleted with time and a novel idea is to recover uranium from extremely low concentrations of 3ppb from the seawater which holds nearly 60% of Earth's uranium. Marine cyanobacteria (*Synechococcus elongatus* and *Anabaena torulosa*) were developed to sequester U from such low concentration and at an alkaline pH of 7.8, typical of seawater (Acharya *et al.*, 2009 and 2012a). The sequestered U was shown to be immobilized in extracellular polysaccharides or in unique surface associated polyphosphate bodies in marine cyanobacteria (Acharya and Apte, 2013a and 2013b). Proteomic analyses revealed that the ability to prevent availability of U in the cytosol underlies uranium tolerance of certain *Anabaena* species (Panda *et al.*, 2017). Such strains have also been used to remove U from RO water generated from desalination plants (Acharya *et al.*, 2012b).

Developing Microbes for Biodegradation of Organic Solvents and Pesticides

Dr. Apte's laboratory was the first in India to demonstrate the molecular basis of lindane (α -HCH or α -hexachlorocyclohexane) biodegradation from *Sphingomonas paucimobilis* (Adhya *et al.*, 1996) that subsequently led to cloning of several of the corresponding *linA* and *linB* genes and their variants

from several *Sphingomonas* spp. in India. His laboratory also overexpressed the *linA* gene in (a) *E. coli* to construct a very sensitive and robust biosensor for lindane (Anu-Pratap *et al.*, 2012), and (b) *Anabaena* to develop a lindane biodegrading strain for rice field (Chaurasia *et al.*, 2013). They also isolated and characterized *Sphingobium* sp. strain RSMS, capable of degrading tributyl phosphate (TBP) and utilizing it as carbon and phosphorus source resulting in complete mineralization of this recalcitrant organic solvent of U and Pu in nuclear industry (Rangu *et al.*, 2014). The biochemical pathway of TBP degradation was elucidated and techniques developed to simultaneously degrade TBP and use the phosphate released for precipitation of traces of uranium still present in TBP (Rangu *et al.*, 2016).

Pseudomonas spp: Degradation of Many Aromatics

Pseudomonas putida CSV86: An evader of CCR. *P. putida* CSV86, a soil isolate from Dr. P Phale lab, utilizes an array of aromatics as the sole source of carbon and energy. This strain has a unique ability to utilize aromatics preferentially over glucose and co-metabolizes aromatics and organic acids (Basu *et al.*, 2006). Aromatics and organic acid mediated repression of glucose metabolism as well as transport was found to be responsible for this ability. In the strain CSV86, glucose enters only through high affinity ATP dependent ABC transporter. This transport system comprises of a periplasmic glucose binding protein, GBP; outer membrane porin, OprB; two inner membrane ABC transporter proteins, GlcF and GlcG and a membrane bound cytoplasmic ATPase, GlcK (Figure 6C) and present together as a probable transcription unit (Paliwal *et al.*, 2014). The periplasmic glucose binding protein confers specificity to this system and has been studied at the structural level to understand the basis of the specificity (Figure 6D, Modak *et al.*, 2014, Pandey *et al.*, 2016).

Pseudomonas aeruginosa strain PP4. *Phthalate degradation*: *P. aeruginosa* strain PP4 is a soil isolate, capable of degrading phthalate isomers (phthalate, isophthalate and terephthalate) as sole sources of carbon and energy. Studies revealed that strain PP4 utilize phthalates efficiently by overcoming the metabolic inhibition adapting different strategies like modulating different forms of GDH or expressing higher concentration of the same enzyme (Vamsee-

Krishna and Phale, 2010), suggesting it to be a better candidate for the on-site bioremediation.

Pseudomonas sp. strain C5pp: Evolution in process: Carbaryl (1-naphthyl *N*-methylcarbamate) is widely used as a broad-spectrum insecticide. *Pseudomonas sp. strain C5pp* isolated from soil metabolizes carbaryl as the sole source of carbon and energy. Carbaryl is metabolized via 1-naphthol, 1,2-dihydroxynaphthalene, salicylaldehyde, salicylate and gentisate to TCA cycle intermediates. The carbon-source dependent enzyme induction studies suggested that carbaryl degradation pathway is segmented into 'upper', 'middle' and 'lower' (Singh et al., 2013) pathway. The combined approach of genomic library analysis and genome sequencing was used to identify

the genes involved in carbaryl metabolic pathway. The sequence analysis suggested genes involved in the pathway have been acquired as a result of horizontal gene transfer.

Strain C5pp appears to be in early stages of evolution and can serve as an interesting model for studying evolution of metabolic pathway for recently introduced pesticide. As strain CSV86 has ability to preferentially utilize aromatics (globally) over simple carbon source like glucose it increases the potential for its use in biodegradation and biorefinery. Moreover, this strain also serves as efficient genetic tool for studying and metabolic engineering, so as to construct a robust strain for efficient removal of aromatic pollutants from the contaminated environment.

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