Aerobic and Anaerobic Biodegradation of PCBs: A Review
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ABSTRACT

This review summarizes recent research results on the biodegradation of polychlorinated biphenyls (PCBs). These compounds, commonly believed to be indestructible, have repeatedly been shown to biodegrade under a variety of conditions. Two distinct classes of bacteria have now been identified that biodegrade PCBs by different mechanisms. The focus of this manuscript is current research involving the aerobic biodegradation of PCBs (natural systems, recombinant organisms, and soil applications) and the dramatic new results demonstrating microbial reductive dechlorination of even highly chlorinated PCBs under anaerobic conditions.

These two PCB-degradative systems include aerobic bacteria which live in oxygenated environments and anaerobic bacteria which live in oxygen-free environments such as aquatic sediments. The aerobes attack PCBs oxidatively, breaking open the carbon ring and destroying the compounds. Anaerobes, on the other hand, leave the biphenyl rings intact while removing the chlorines. This anaerobic dechlorination degrades highly chlorinated PCBs to less chlorinated derivatives, and the two naturally occurring processes are complementary, and a two-step treatment may permit the biological destruction of nearly all of the PCB mixtures commonly used.

I. INTRODUCTION

A. Definition

Polychlorinated biphenyls (PCBs) are a family of compounds produced commercially by the direct chlorination of biphenyl using ferric chloride and/or iodine as the catalyst.27 The biphenyl molecule is made up of two connected rings of six carbon atoms each (see Figure 1), and a PCB is any molecule having multiple chlorine attached to the biphenyl nucleus. Chlorines can be placed at any or all of the ten available sites, with 209 different PCB compounds theoretically possible, varying in the number and position of the attached chlorines. The individual isomers and homologs are generally referred to as congeners. Of the 209 possible congeners, only about half are actually produced in the synthesis due to steric hindrance. The position of the chlorines is indicated by the numbering scheme shown in Figure 1. The reaction shown in Figure 1 would produce a large number of different PCB structures; only the 2,3,4,5,6-pentachlorobiphenyl (2,3,4,5,6-PCB) is drawn as an illustration.

PCBs were manufactured and sold as complex mixtures differing in their average chlorination level. The crude mixtures...
mability, electrical, and stability properties, PCBs found application in a wide variety of industrial uses including heat transfer fluids, hydraulic fluids, fire suppressants, plasticizers, flame retardants, organic dyes, and dielectric fluids. 8

In a 50-year period approximately 1.4 billion pounds of PCBs were produced. Such extensive application of these chemically and thermally stable compounds has resulted in widespread contamination. 9,10 It is estimated that several hundred million pounds have been released to the environment. 11 The lipophilic nature of PCBs contributes to their tendency to accumulate in fatty deposits and results in a magnification in the food chain. 12

C. Health Risk

This accumulation of PCBs in organisms and the past exposure of some industrial workers was initially a cause for concern. 13-15 But the toxicity associated with PCBs has recently become re-evaluated, as newer evidence has been presented that "...the only observed acute effects have generally been minor. So far, no significant chronic health effects have been causally associated with exposure to PCBs or PBBs." 16

Another health risk commonly associated with PCBs involves their role as suspected human carcinogens. This premise stems from early reports that high levels of Aroclor 1260 caused liver cancer in rats. 17 But a study by the National Cancer Institute (1978) concluded that Aroclor 1254, a mixture of PCBs having a slightly lower level of chlorination than Aroclor 1260, was not carcinogenic. 18 In addition, a recent thorough review of the epidemiological literature stated that "No conclusive evidence thus far reported shows that occupational exposure to PCBs causes an increased incidence of cancer." 19

Most reviews concerning the biological and toxic effects of PCBs note that the relative potency generally correlates with the degree of chlorination. 20,21 These results suggest that the toxicities of the mixtures are variable, and it is therefore reasonable to say that the activities of individual congeners may also differ considerably. Valuable data involving structure/activity relationships for individual congeners is now available. 22,23 Safe has concluded from animal studies carried out in his laboratory that the most toxic PCB congeners contain two para and at least two meta chlorines, and the addition of ortho chlorines reduces this effect significantly. 24

II. AEROBIC BIODEGRADATION OF PCBs

A. Enrichments

Most environmental contamination by PCBs is in the form of complex commercial mixtures (e.g., Aroclor 1242) containing >60 different congeners with varying degrees of chlorination. Biodegradation of this large number of distinct substrates therefore requires broad enzymatic specificity. In addition, chlorinated organic materials frequently resist microbial degradation. 25 Although these complex chlorinated mixtures can be difficult to biodegrade, the aerobic bacterial biodegradation of PCBs is known and has been well studied. 26-30 Previous reviews on the aerobic biodegradation of these materials have been published. 31,32 This review concentrates on research reported after their publication.

Using a rapid screening procedure, Bedard et al. isolated natural aerobic bacteria capable of degrading PCBs in nearly every contaminated soil they tested. Soil and sediment samples were collected from PCB-contaminated sites and cultures were enriched on biphenyl as the sole carbon and energy source available to the bacteria. The bacterial enrichments obtained were assayed for their ability to degrade defined mixtures of PCBs. Using this approach, a diverse group of 25 strains of PCB degrading bacteria were isolated and characterized. 33 This method allowed the rapid determination of PCB competence for a large number of isolates. In addition, the use of defined mixtures in place of complex environmental investigations into the nature of the enzymatic specificity observed. The results of this screening technique are shown in Figure 2. 34 Note that all of the organisms isolated are capable of degrading the lightly chlorinated PCBs. Characterization (genus and species for some of the PCB degrading organisms isolated by several different workers is shown in Table 1. These results indicate that naturally occurring organisms can degrade PCBs, are quite common in the environment, and that the organisms consist of many different microbiological types. It is interesting to note that nearly two-thirds of the organisms represented in this survey are members of the genus Pseudomonas.

This pathway is similar to the degradation pathways for other aromatic substrates deduced for biphenyl and toluene. 35,36 The first two steps in the metabolism of biphenyl involve dioxygenase attack at the 2,3-position with subsequent dehydrogenation to the catechol. 37 The next step involves fission of the ring to the meta-cleavage product. 38 These authors also proposed that this ring fission product was further converted to benzoic acid, as this metabolite was identified from crude cell-free mixtures incubated with 2,3-dihydroxybiphenyl. This cleavage to benzoic acid was later confirmed. 39

In the earliest reported isolation of PCB-degrading strains, Ahmed and Focht identified both the meta-cleavage product and p-chlorobenzoic acid as metabolites of the degradation pathway. These authors postulated that the PCB degradation pathway is the same as that determined earlier for biphenyl and other aromatic hydrocarbons. This hypothesis was confirmed by Fukutaka et al. 40 with the identification of the meta-cleavage product and chlorobenzoic acids as metabolites of PCB.

In general, most PCB degrading aerobic bacteria are able to degrade only the lower chlorinated PCB congeners (e.g., mono- to tetra-substituted). 41-43 It is possible that higher chlorination levels result in steric hindrance of 2,3-dioxygenation by chlorine substitution at either of these two positions. 44 But several aerobic bacterial strains have demonstrated the exceptional ability to degrade an even larger range of congeners, up to and including penta-, hexa-, and even several heptachlorophenyls (Pseudomonas strain LB400). 44,45 Alcaligenes eutrophus HB850, 46,47 Corynebacterium MB1, 48,49 and Acinetobacter strain P6. 50,51 One of these organisms has demonstrated the capacity to degrade more than 90% of the PCBs present in the mixture Aroclor 1242 (LB400). 52

Although these organisms use the 2,3-dioxygenase degrada-
tion pathway described above, they may also metabolize through other routes. It is known that congeners containing a 2,5-chlorophenyl ring are preferentially degraded by strains HB850 53 and LB400. 54 In addition, the reduction of different metabolites led to the proposal that a significant mechanism for PCB metabolism in these organisms involves a novel 3,4-dioxygenase attack. 55 This proposed 3,4-dioxygenase attack has been named by Gibson in both HB850 and LB400 by identification of the expected chlorodien intermediate from 2,5,2'-5'. 56 This additional dioxygenase pathway may partially explain the exceptional range of PCB-degrading activity demonstrated by A. eutrophus HB850 and Pseudomonas strain LB400.

It is not currently known if the 2,3- and 3,4-dioxygenase activities originate from the same enzyme. It is clear, however, that the congener specificity indicates two distinct classes of dioxygenases. The dioxygenase type present in Acinetobacter P6 and Corynebacterium MB1 is particularly active against congeners containing dodeca-para-substituted PCBs, while the enzyme from Alcaligenes HB850 and Pseudomonas LB400 prefers 2,5-substitution patterns. In general, these specificity are complementary and treatment with an organism from each class results in even greater PCB degradation. 57

C. Optimization

It has been demonstrated that growth on biphenyl as the sole carbon source enhances the degradative activity (LB400). 58 This is a disadvantage in soil applications where other carbon sources are available. The degradation of PCBs bound to soil has been investigated. 59,60 Although PCBs are degraded in these systems, the rates decrease significantly (more than 50-fold) compared to the biphenyl assays. One possible explanation is that biphenyl is required as the sole carbon source for growth. In this case, PCB degrading enzymes. The importance of biphenyl in the soil degradation of PCBs has been investigated by Focht 61,62 and enhanced degradation of
PCBs on soil were observed upon the addition of biphenyl as a carbon source. In addition, the PCB-degrading activity of growing cells was significantly greater for Actinobacter sp. P6 and Arthrobacter sp. B18 than the activity observed with resting-cell suspensions. Biphenyl was utilized as the carbon source, and it is reasonable to require for maximal PCB degradative competence as an inducer of this dioxygenase pathway.

The PCB degradation pathways described earlier produce chlorobenzenes that are not further metabolized by these strains, although other organisms are known to mineralize these compounds. Although many organisms can grow on mono- chlorobiphenyls (LB400, HB50, KTF75, KTF77, KTF78, Q1, M5, BM-2, BM1), microorganisms which could use complex PCB mixtures as a carbon source may perform better in soil applications. Strains which can degrade monochlorobenzenes and further metabolize the chlorobenzoate have been reported. A Pseudomonas strain JBI was isolated that can grow on mono- chlorobiphenyls, degrade mono-chlorobenzenes, and can metabolize other congeners. In addition, Focht and Huang have developed a new strain which is also capable of degrading monochlorobiphenyls and metabolizing the chlorobenzoate intermediates, resulting in growth on 3-ClB as the sole carbon source. This strain was generated via a method that facilitates the rapid exchange of genetic material between two parent strains. The development of new strains that could grow on the more highly chlorinated PCBs would represent a major advance in the aerobic biodegradation of PCBs.

Other methods to enhance the aerobic bacterial biodegra- dation of PCBs have also been reported. The addition of the aminopolysaccharide polymer chitin has been observed to in- crease the rate of PCB degradation. The effects of polymer addition are shown in Table 2. Note that this method generally resulted in a twofold increase in the degradation rate by the indigenous soil microorganisms. The chitin appears to act as a solid substrate for growth as well as an efficient sorbing component for the PCBs, and therefore increases the bio availabilty of these hydrophobic compounds. The addition of adapted PCB degrading bacteria resulted in even greater soil degradation rates (Table 2, microbe addition).

D. Genetic Engineering

The genes encoding microbial degradation of PCBs have been isolated and utilized to construct recombinant organisms capa- ble of degrading PCBs. These studies utilized soil microorganisms from the genus Pseudomonas that degraded PCBs via the 2,3-dioxynaphthalene pathway (see Figure 3). Furuhashi and Miyazaki obtained the genes from Pseudononas putida strain K707, which is known to degrade mono- to tri-ClB, including 4-ClB, 2,3-ClB, 1,2-ClB, 2,4-ClB, and 2,4,4'-ClB. The genes were then cloned into a broad-host-range plasmid and a transformant was isolated that was capable of degrading PCBs. The researchers discovered that the genes encoding three of the four enzymes involved in PCB degradation (bphA through bphC) were localized on a small DNA fragment (7.9 kb). In addition, the 2,3-dihydroxy- biphenyl dioxygenase (bphA) was isolated and sequenced from different bacteria. Several other methods have been developed to express enzyme activity in transformed bacterial strains. In one method, the entire gene for a Pseudomonas strain LB400 was cloned into the plasmid pBluescript II, followed by the cloning of an additional copy of the gene into the plasmid. The resulting plasmid was then transformed into the Escherichia coli strain DH5α. The recovered colonies were then screened for the ability to grow on biphenyl as the sole carbon source. The recombinant strain LB4500 was found to grow on biphenyl. The recombinant strain LB4500 was then transformed with the plasmid containing the bphB and bphC genes, resulting in the recombinant strain LB4500B. The resulting strain was found to grow on biphenyl as the sole carbon source.

The correspondence between the PCB-degrading organisms and their resistance to PCBs was determined. The results indicated that the recombinant strain LB4500B was capable of degrading PCBs at a rate comparable to the wild-type strain. The high correlation among all of these distinct organisms implies that the PCB-degrading genes did not evolve independently and the genes must have been acquired through some form of DNA transfer. This result may have important implications as it demonstrates that natural organisms can transfer and have transferred the PCB-degrading genes in the environment. These mobile genes enable the organisms to attack a broad range of PCB congeners.

Although the recombinant strain LB4500B can degrade PCBs better than the wild-type strain LB400, the recombinant strain shows a significant advantage in soil remediation applications. In addition to faster growth rates to higher cell densities, the recombinant demonstrates superior viability and temperature resistance. It is hoped that the recombinant E. coli strains will be developed to survive 27 d on soils, significantly lower than the 27 d survival of the wild-type strain. More important is the fact that the recombinant LB4500B does not require growth on biphenyl as the sole carbon source for optimal PCB-degradative competence. Therefore, on soils where other organic material is readily available, the recombinant should display superior PCB-degradative competence.

E. Fungi

Microorganisms other than the bacteria, notably fungi, have also been reported to aerobically degrade PCBs. The filament- ous fungus Aspergillus niger, used as a model of mammalian aromatic hydroxylase, has been shown to degrade the lower chlorinated PCBs in the commercial mixture Chlophen A 30. The wood-decay white-rot fungus Phanerochaete chrysospor- ium has also been utilized in the degradation of PCBs at very low concentrations. The results indicate that P. chrysosporium is capable of the complete degradation of highly chlorinated PCBs, and activity has only been observed at very low concentrations (0.008 to 0.009 for PCBs 209 and 210, respectively). The results indicate that P. chrysosporium is capable of degrading chlorinated PCBs at very low concentrations, and activity has only been observed at very low concentrations (0.008 to 0.009 for PCBs 209 and 210, respectively). The results indicate that P. chrysosporium is capable of degrading chlorinated PCBs at very low concentrations, and activity has only been observed at very low concentrations.
as many as seven chlorines. (4) For the known cases, the 2,3,7,8-tetra- chlorodibenzo-p-dioxin pathway is common and quite similar in otherwise unrelated organisms. (5) Similarities in the genes encoding PCB degradations imply that these genes are being transferred between bacteria in the environment. (6) In general, the effect of aerobic bacterial PCB biodegradation is to remove the less chlorinated congeners. (7) No aerobic microorganisms have been reported that degrade the more highly chlorinated commercial mixtures Acorcl 1260 or Clophen A 60.

III. ANAEROBIC BIODEGRADATION OF PCBs

A. Environmental Evidence

Despite the extensive research on the aerobic biodegradation of PCBs, little was known about their fate in anaerobic environments such as river or lake sediments until very recently. Early studies indicated that anaerobic fermentations did not alter PCB concentrations with organisms from sludge or marine sediments. But more recently, alterations of the PCBs present in anaerobic culture have been observed. These alterations involve the selective removal of highly chlorinated PCB congeners with corresponding increases in congeners containing only a few chlorines (mono- and dichlorobenzylidene).

Several different patterns or alterations were observed for Hudson River sediments originally contaminated with Acorcl 1242 (see Figure 5). All three patterns showed markedly lower levels of most tri-, tetra-, and pentachlorobenzylidene and increased levels of mono- and dichlorobenzenes. Note that the detector response displayed in the chromatogram is non-linear and particularly poor for the congeners containing very few chlorines with short elution times. Quantitation of the individual capillary chromatographic peaks in all sediments the 2.6, 2.6', 2.6', and all dichlorobenzenes increased 2- to 10-fold, and the level of the monochlorobenzylidene PCB increased 3- to 40-fold. The observed transformations are congener specific, demonstrating selective removal of meta and para chlorines and increases in the expected partially dechlorinated PCB congeners. No transformed transformation processes such as evaporation or aerobic degradation could account for the striking changes observed, and it was therefore proposed that anaerobic microorganisms in Hudson River sediments are reducing dechlorinating the PCBs. In addition, transfers of chlorines from the highly chlorinated Acorcl 1260 had been observed in the environment.

Anaerobic dechlorination of chlorinated aromatic compounds is not unprecedented. Tidege and coworkers identified an anaerobic sulfidogenic bacterium strain DCB-1 capable of reductively dechlorinating dichlorobenzones. This organism represents the first and only anaerobe in pure culture capable of aromatic reductive dechlorination. It was isolated from an anoxic anaerobic consortium capable of mineralizing chlorobenzenes. A review of the anaerobic degradation of pesticides in recently developed methods to determine dechlorination of a number of aromatic substrates, including chlorobenzenes, chlorophenols, chloroaranes, and herbicides. Reductive dechlorination of aromatics has also been reported with aerobic bacteria (chlorinated phenols and chlorinated quinones).

B. Laboratory Confirmation

The proposed microbial dechlorination in anaerobic river sediments was confirmed in the laboratory. The result of anaerobic dechlorination of Acorcl 1242 by microorganisms in Hudson River sediments is shown in Figure 6. Note the dramatic loss of the highly chlorinated congeners with corresponding increases in the less chlorinated products. These microorganisms dechlorinate the PCB mixture so extensively that it is converted from 85% tri- and tetra-chlorinated PCBs to 85% mono- and dichlorinated products. The end result of this natural process is the conversion of the more highly chlorinated PCBs into congeners of low toxicity that are degraded by a large number of aerobic bacteria.

The dechlorination was found to be selective removal of meta and para chlorines as well, confirming that this natural process observed in the lab is the same as the dechlorination found in the environment. Therefore an additional benefit of this anaerobic dechlorination is the removal of the meta and para chlorines known to contribute to PCB toxicity. The similarity between environmental and laboratory changes can be seen by comparing Figures 5 and 6.

Methods to accelerate this desirable natural process have also been identified. The addition of a simple minimal medium first described by Shelton and Tiedje containing nutrients and trace minerals results in a significantly more rapid activity compared to that of unsupplemented cultures. Other factors which increase the rate of dechlorination include the addition of a complex carbon source (fluid thioglycolate medium with beef extract) or a detergent (Triton X-705). These effects are additive and a combination of variables results in even greater enhancement. Importantly, the dechlorination of the highly chlorinated Acorcl 1260 has also been observed in the laboratory. The concentration of the most highly chlorinated congeners (hexa-, hepta-, and octachlorobenzenes) has been decreased by more than one-third the original level in Acorcl 1260. Anaerobic dechlorination of Acorcl 1260 has recently been observed by others as well.

C. Single Congener

Methods for the synthesis of individual PCB congeners and commercially available material have provided single compounds for a more detailed study of this anaerobic dechlorination process. The interconversion of the dechlorination observed with Hudson River sediments on the congener 2,3,4,3',4'-CB is shown in Figure 7. The dechlorination activity observed demonstrates a sequential pathway from the penta- (2,3,4,3',4'-CB) to tetra- (2,3,4,3'-CB), tri- (2,3,4,3'-CB), di- (2,3,4-CB), and mono- (2,3-CB) chlorinated biphenyls (major products shown in parentheses). The result of this process is the conversion of one of the more toxic congeners (2,3,4,3',4'-CB) into a mono- chlorobiphenyl (2-CB) which has low toxicity and is easily metabolized by aerobic bacteria and higher organisms.

This result again confirms that chlorines are removed from only the meta and para positions as was observed for the river sediment itself. The selectivity of each individual step is surprising; for example 2,3,4,3'-CB is the only detectable tetrachlorinated product observed from 2,3,4,3',4'-CB, as it is produced in stoichiometric amounts.

This microbial selectivity for only meta and para chlorines is different than that observed from the direct electrochemical reduction of PCBs. Furul et al. determined that although the PCB reduction pathways were complex, dechlorination via volatilization was observed from the ortho, meta, and para positions. In some cases the ortho dechlorinated species was the major product, for example 2,3-CB yields 89% 3-CB and only 11% 2-CB. This can be contrasted to the microbial dechlorination of 2,3'-CB, where 2-CB is the only observable product.

The dechlorination of single congeners with higher toxicity has also been demonstrated by Tiedje et al. These investigators found that 2,3,4,3',4'-CB and 3,4,3',4'-CB were dechlorinated at rates comparable to other penta- and tetra-chlorobiphenyls, even in the presence of the complex PCB mixture Acorcl 1242.

From such studies utilizing single PCB congeners, one can prove that microbial reductive dechlorination is occurring in the sediments. The stoichiometric production of PCBs containing fewer chlorines demonstrates the substitution of hydrogen in place of chlorine. It is believed that the anaerobic microorganisms are utilizing the chlorine as the terminal electron acceptor, involving the addition of the electron to the carbon-carbon bond, followed by chloride loss and subsequent hydrogen abstraction (see Figure 8). The compound from which the hydrogen is ultimately abstracted is unknown, and potential primary electron donors include water, hydrogen, or an organic compound. The availability of hydrogen in anaerobic microbial systems may make it the most likely primary source of reducing equivalents.

Recently it has been shown that Hogenkamp and coworkers that vitamin B12 can catalyze the reductive dechlorination of carbon tetrachloride and other chlorinated methanes. Vitamin B12 is a known hydride transfer agent and this result suggests an alternative dechlorination mechanism involving a single step.
Ar-Cl + e⁻ + R-H ⇄ Ar-H + Cl⁻ + R (H₂O) (HO) (H) (Cl⁻)

FIGURE 8. Possible mechanism for reductive dechlorination catalyzed by anaerobic microorganisms. In the proposed scheme, the organism utilizes PCBs as an electron acceptor, with addition of the electron to the carbon-chlorine bond, chlorine loss, and hydrogen abstraction from an unknown species.

concerted process catalyzed by this cobalamin cofactor or other corinoids present in these anaerobic microorganisms.

D. Anaerobic Degradation

The dechlorination process described earlier does degrade highly chlorinated PCBs, but the organisms leave the biphenyl nucleus untouched and less chlorinated PCBs are formed. Although this dechlorination represents actual biodegradation of highly chlorinated PCBs, it is being distinguished here from processes that do attack the biphenyl ring, resulting in potential mineralization of the PCB. Such a process has recently been reported by Rheo and coworkers. 24,25 In this work, bacterial populations from Hudson River sediments were reported to anaerobically degrade the lightly chlorinated congeners in PCB mixtures. No metabolites were identified, and no evidence for the dechlorination process described earlier was observed by these authors. Although the dechlorination and biodegradation processes reported here both utilized sediments from the Hudson River, CO₂ was provided to the dechlorinating systems as bicarbonate, but it was absent in the biodegradation studies. It is interesting to speculate that CO₂ may be important in determining the type of anaerobic activity observed on PCBs. It is possible that in the absence of CO₂, a selection is imposed favoring organisms capable of degrading PCBs to obtain CO₂ and/or low molecular weight metabolites as electron acceptors.

E. Summary

This environmental dechlorination of PCBs has now been observed in a large number of contaminated anaerobic sediments. Sites include many locations in the Hudson River (New York), Silver Lake (Pittsfield, Massachusetts), New Bedford Harbor (Massachusetts), Escambia Bay (Pensacola, Florida), Woods Pond (Massachusetts), the Housatonic River (Connecticut), the Sheboygan River (Wisconsin), Waukegan Harbor (Illinois), and the Hoosic River (North Adams, Massachusetts). 26 The widespread occurrence of this natural process indicates that it is a general phenomenon.

These changes observed in PCB-contaminated anaerobic river sediments led to the proposed microbial reductive dechlorination of PCBs. 27,28 This process has now been confirmed in a number of laboratories with sediments from many distinct aquatic systems. 29-31 Some of the most significant findings from current anaerobic dechlorination experiments follow.

1. Dechlorination has been observed in a large number of sediments and the process is widespread in the environment.
2. Although congener preferences are demonstrated, in general the organisms present in Hudson River sediments exhibit broad dechlorination activity on the more highly chlorinated PCBs.
3. These anaerobic microorganisms are capable of dechlorinating even the previously recalcitrant, highly chlorinated PCB congeners contained in Aroclor 1260. 32 All results to date involve primary cultures and pure PCB dechlorinating strains have not yet been isolated.
4. Dechlorination selectively removes meta and para chlorines, significantly reducing any toxicity associated with PCBs. 33 The less chlorinated congeners that are produced are known substrates for aerobic bacterial systems.

IV. CONCLUSIONS

This paper has focused on recent progress (since 1985) in the aerobic biodegradation of PCBs and the new anaerobic dechlorination process recently discovered. The large number of researchers cited have repeatedly demonstrated that PCBs, commonly believed to be indestructible, are degraded by a number of diverse microorganisms.

Two separate and complementary biological systems have been the focus of this report on the biodegradation of PCBs. Anaerobic bacteria, present in river and lake sediments, remove chlorine from even the most highly chlorinated PCBs. This process is relatively broad, attacking a large array of highly chlorinated PCBs. The resultant lightly chlorinated compounds are less toxic, and are known substrates for aerobic bacterial biodegradation. Such aerobes have been identified in nearly all PCB-contaminated areas and are widespread in the environment. The obvious complementarity of these biological processes leads to the combined treatment scheme shown in Figure 9 (only one PCB congener is shown as an illustration). Successful application of this sequential treatment may enable the bioremediation of nearly all types of PCB contamination.

REFERENCES
