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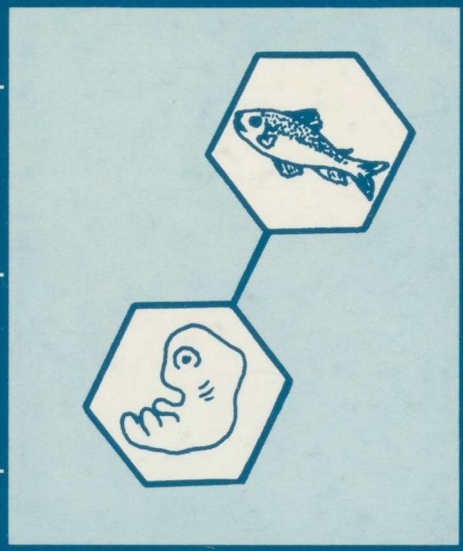
Cause-Effect Linkages 2: Symposium Abstracts, September 27-28, Grand Traverse Resort, Traverse City, Michigan

Michigan Audubon Society

Steve Schneider

Michigan Audubon Society

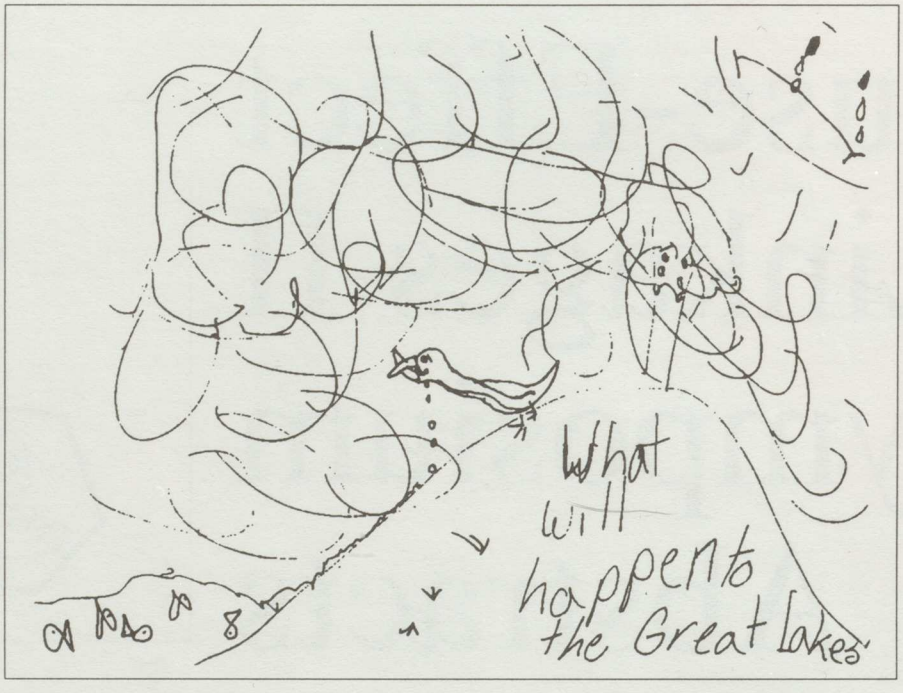


Cause-Effect Linkages II Symposium Abstracts

September 27–28, 1991
Grand Traverse Resort, Traverse City, Michigan

"What we are doing to the Great Lakes we are doing to our children."

– Joe Seamans,
Great Lakes, Fragile Seas
National Geographic Society



Drawing by Emily Campbell, age 8

The Michigan Audubon Society is dedicated to the protection and preservation of wildlife and natural resources through public education, research, and sound environmental policies. In support of these goals MAS is sponsoring the Cause-Effect Linkages II Symposium. The Society wishes to provide an open forum where Great Lakes basin bioeffects researchers can present current data to the general public and government representatives on the relationship between trace toxic contaminants and wildlife and human bioeffects. While the Society supports the symposium and will present a summary statement to the International Joint Commission, the organization does not necessarily support the individual comments and recommendations of the participants.

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"To whom do I owe my loyalty?...I believe you must be loyal to the public interest directly and always be guided by a very simple precept. Do what is right for the resource regardless of the political consequences."

– Dr. James P. Ludwig
Testimony to the International Joint Commission
Hamilton, Ontario, 1989

Preamble and Acknowledgements

The rationale and methodology of cause-effect linkages has developed over the past one hundred thirty years. The first postulates, outlined by Robert Koch, addressed the relationship between disease and microbes. Those principles were applied to causality between disease and environmental contaminants including smoke and other non-infectious agents in the 1960s. In 1989 during the first Cause-Effects Workshop, the Council of Great Lakes Research Managers adopted the criteria for determining causality between persistent toxic contaminants and observed effects in fish and wildlife.

Over two years have passed. Much new research has been pursued, innovative analytical techniques developed, and intriguing new observations accumulated. The Michigan Audubon Society, with concurrence of the International Joint Commission, is sponsoring Cause-Effect Linkages II to provide a public forum for presentation and discussion of the new data. In addition, the symposium will explore the relationship between human health effects and toxics, and the development of biomarkers that are indicators of impairment of health within a given species. It is believed that these advancements will result in a new paradigm, amended regulatory guidelines, and revised public policies to address Great Lakes environmental quality.

These abstracts and comments are published to provide accurate information to the general public, educators, and policy makers. This information will also be presented during the Biennial Meeting of the International Joint Commission in Traverse City, Michigan, September 30 to October 2, 1991. The full proceedings of the symposium will be published by the Michigan Audubon Society in early 1992.

A great many people associated with the *Michigan Audubon Society* (MAS) and its contractor for conference preplanning, *Ecological Research Services, Inc.* (ERS), have contributed to the production of these abstracts and the program.

Program planning was the responsibility primarily of *Heidi J. Auman* (ERS), who worked tirelessly with *Sally Cole-Misch* and *Rita Kerner* of the International Joint Commission (IJC) to complete logistical arrangements for speakers and attendees. *Loraine M. Campbell*, president of MAS, worked closely with ERS staff and MAS volunteers and to accomplish many tasks. *Rick Campbell* (MAS) served as publisher of these abstracts, lending his expertise in printing and design. *Mike Gilbertson* of the IJC deserves special recognition for inspiring and producing the first Cause-Effects Linkages meeting in March 1989, and later suggesting this follow up symposium to MAS officers. *Matthew E. Ludwig* (ERS) developed the computerization of abstracts. *James P. Ludwig* (ERS) helped coordinate appearances of speakers and panelists. *Stephen Schneider*, who is editing the symposium proceedings, volunteered his time and expertise in editing the abstracts as received from authors.

The authors themselves deserve the lion's share of credit for their efforts. Scientific symposia are very rarely open to public participation. The scientists who came to this symposium to give their reports and views are unusual in their willingness to expose their nascent work to public view. We commend them for their diligent research and their efforts to ensure the success of this open forum. Finally, the Directors of Michigan Audubon Society are to be commended for their leadership in bringing this issue forward directly through this open public symposium.

-Loraine Campbell, President MAS
-Theo Colborn, Symposium Co-chair
-James P. Ludwig, Symposium Co-chair

1.0 Introduction

1.1 Welcome

Good morning. On behalf of the members and Board of Directors of the Michigan Audubon Society, I welcome you to Traverse City and The Cause-Effect Linkages II Symposium. It is over two years since the Council of Great Lakes Research Managers called Cause-Effects I to order and, with studied scientific objectivity, examined the relationship between persistent toxic chemical contamination and the resulting bioeffects in fish and wildlife. Much has been accomplished in the interim. The Great Lakes Basin bioeffects researchers have worked closely together, coordinating their research efforts, their techniques, and their observations. Scientists have spent three more field seasons gathering new information and observations. Papers have been published. A clearer picture is emerging of the complex relationships between the physical environment, the flora, fauna, and human populations of the Great Lakes ecosystem. New and challenging questions are being asked. It is time to formally review the progress and the questions. It is also time for us to meet the challenge placed before us. We must be willing to change how we live and do business in the Great Lakes Basin as an international community. We must accept our responsibility as an agent of change in the environment, and as a species that is directly affected by the changes we have initiated.

To that end, this meeting differs in one significant aspect from the first Cause-Effects Workshop. That was a closed meeting. It afforded scientists and regulators the opportunity to share their data, techniques, and candid concerns regarding the issues *among themselves* in an atmosphere that was protected. It was an opportunity for them to outline the scientific criteria they would follow in establishing causal relationships between phenomena observed and the exacting analytical measurements now possible.

But, the public has also become increasingly informed and concerned in the last three years. Newspaper and magazine articles, TV specials, and news accounts have focused on the Great Lakes, on toxics, on deformed birds, injured wildlife populations, and the hazards of consuming Great Lakes fish. The popular press has made daring statements. They have told alarming stories with incredible social and political repercussions. Some believe the stories, others do not. Some are overwhelmed by the complexity of the issue, the convoluted chemistry and the economic implications. Many say they don't know who or what to believe.

Therefore, Cause-Effect Linkages II is designed to be open to the general public—to businessmen, sportsmen, environmentalists, and those who make and administer policy in the United States and Canada. It is open to women who eat fish and who are concerned about the health of their babies. It is open to educators who must be able to answer the questions students put before them in an accurate but clear fashion. Cause-Effect Linkages II is an open forum for each of us, with our divergent points of view. It is an opportunity to listen to the scientists, to ask questions, and to contemplate the consequences of the answers.

There are two goals for this symposium. The first is to present objective scientific data. The second is to translate those facts into a form each of us can use in our own lives and as members of biological, social, and political communities. As we begin, we must bear in mind the complexity of nature. It is not an easy task to establish cause-effect relationships. Grandmothers still admonish us not to get chilled or we'll catch a cold. How long did it take us to learn that temperature change *per se* does not cause colds? How much work still must be done in the medical community to establish links between viruses, diseases, and the impact of environmental factors on our immune system. We must remember that sheer volume of effort does not always translate into clear answers. Seeking the cure to the common cold ought to be a warning that this process will be lengthy and full of pitfalls, and for some, pratfalls.

As we begin, it is important that we review the criteria for establishing a cause-effect relationship. These rules, first stated by Sir Karl Popper and put forth at Cause Effect Linkages I are:

1. **The consistency of observations**; repeated observations under different circumstances should be consistent.

2. **The strength of association**; a more intense exposure should result in a more severe effect. In other words, there should be a dose/response relationship.

3. **Specificity of the association**; the outcome should be well defined and a recognizable entity. For instance, it should not be claimed that all sorts of illnesses, from headaches to cancer, are all caused by 2,3,7,8,-tetrachlorodibenzo-p dioxin (TCDD).

4. **Time sequence**; in investigating a poisoning episode, a definable population should have been exposed, and the exposure must have occurred before the illness.

5. **Coherence with other information**; if in one study a particular observation is made and in another study, totally different results or negative findings are obtained, then it must be determined if

the initial observation was a spurious association.

To these five, Michael Gilbertson has added a sixth rule:

6. **Predictive power of the relationship**; a reputed association should predict, with reasonable accuracy, the expected effect.

Please note, that these criteria and the abstracts of most of the presentations you will hear today and tomorrow are contained in the publication included in your folder. Additional copies of Symposium Abstracts are available at the registration desk for $4.00 per copy. Complete proceedings of Cause Effects II will be available through the Michigan Audubon Society Bookstore in early 1992. You may preorder your proceedings today at a savings. Copies of the proceedings from Cause Effect I are also available for $12.00.

GILBERTSON, M., Great Lakes Water Quality Board, International Joint Commission, 100 Ouellette Ave., Windsor, Ontario N9A 6T3, Canada.

1.2 Keynote Address

The Forensic Approach to Great Lakes Toxicology: Have We Invented a New Science for Regulation of Persistent Toxic Substances?

Welcome to the second Cause-Effect Linkages Workshop. As many of you must be aware, the first Cause-Effect Linkages Workshop held in Chicago in March of 1989 caused a considerable tension within both the scientific and the regulatory communities. One of my interests during the past 2 1/2 years has been in studying this tension because I believe it can provide us with meaning and guidance for what we are doing and with a basis for resolving differences and reconciling the protagonists.

J.B. Sprague, in a thoughtful piece about the regulation of the pulp and paper industry, formulated a three-part strategy for pollution control. Tactic 1 was based on end-of-the-pipe controls using best available technology economically achievable; Tactic 2 was the setting of water quality objectives and enforcing them through chemical analytical surveys; and, Tactic 3 was to undertake biological field surveys to verify that the first two tactics had been successful and that damage to aquatic resources was restricted to socially acceptable levels. These tactics have been, more or less, at the heart of the strategy for pollution control under the U.S. Clean Water Act and under the Canadian Fisheries Act for the past two or three decades.

In the 1960s and 1970s, through lethality or other experiments using sensitive laboratory species, water quality objectives and effluent limitations were set either on the basis of bioassays using aquatic species or on the basis of results of chemical analyses of samples. Permits were written following procedures under the U.S. National Pollution Discharge Elimination System (NPDES) or will be written under the Ontario Municipal and Industrial Strategy for Abatement (MISA). The whole process has proved enormously successful for the control of conventional pollutants such as sewage, phosphorus, BOD, colour, and total suspended solids as well as toxic substances such as zinc, copper, and nickel. The policy assumed that wastes that had been "suitably treated" could be released into the waterways of the two nations, since the streams, rivers, and tributaries had an assimilative capacity for receiving and detoxifying these treated wastes. The policy contained in the U.S. Clean Water Act and subsequently in the 1978 version of the Great Lakes Water Quality Agreement was that the discharge of toxic substances in toxic amounts was prohibited. The corollary is that the discharge of toxic substances in nontoxic amounts is permitted and this has been the *modus operandi* for about 20 years in both Canada and the United States.

Slowly, the quality of the waters of the two nations improved. Wastewater treatment plants were built and the most visible and obvious signs of gross pollution began to recede. This whole approach spawned an enormous bureaucracy including toxicologists involved in experimentation with laboratory animals, analytical chemists determining concentrations of chemicals in samples, engineers constructing and operating treatment plants, politicians appropriating money and writing legislation, and bureaucrats writing standards and regulations and interpreting the law. An entrenched tradition had been created and progress was being made.

Until recently, there was very little to disturb this harmony even though there was a growing scourge of persistent toxic substances. The seeds of dissent came from an unlikely source—the bird watchers. Charles Broley documented the decline of the Florida bald eagle during the critical years of the introduction of DDT and other organochlorine insecticides in the mid-1940s and by the early 1950s was making definitive statements about the causal relationship. In the early 1960s, Joe Hickey at the University of Wisconsin was studying causes of reproductive failure in Lake

Michigan herring gulls with his student Tony Keith and contractor Jim Ludwig. Rachel Carson in 1962 had published her book *Silent Spring*, which chastised the agricultural chemical industry for peddling organochlorine chemicals such as DDT and dieldrin and she in turn was chastised for mentioning the subject. In Canada, the Pest Control Products Act was updated and most uses of DDT and dieldrin were cancelled. In the United States, the Federal Insecticide, Fungicide and Rodenticide Act was moved from the Department of Agriculture to the newly formed Environmental Protection Agency. But none of this seriously challenged the regulatory community involved in water pollution control. To change anything that was occurring would require resorting to the courts. Thus, it became clear in the late 1960s that to control organochlorine chemicals, the detective work required to infer a causal relationship between the injury and these chemicals would need to be sufficiently comprehensive as well as reliable to be able to make a legal case. This was the beginning of what I have called the forensic approach and which corresponds with Sprague's Tactic 3 involving biological field surveys that show that, even though everyone is in compliance with all regulatory requirements, something has gone seriously awry.

Two of the original forensic sleuths were Dick Aulerich and Bob Ringer who showed that outbreaks of kit mortality in ranch mink in the late 1960s were related to PCBs in Great Lakes fish. Joe Hickey and his student Dan Anderson became concerned about eggshell thinning in birds and this phenomenon, which had been experimentally linked by U.S. Fish and Wildlife Service biologists Richard Porter and Stan Wiemeyer to contamination with DDE, was documented in aquatic birds dependent on Great Lakes fish. But the regulatory response in the mid-1970s was still to assume that these compounds could be treated like other toxic wastes through development of water quality objectives to limit discharges or ambient concentrations. Despite the fact that it was already known that these compounds were persistent, bioconcentrated in Great Lakes food webs, and dispersed far from the scene of release as well as being toxic, it was assumed that organochlorine compounds could be "suitably treated" and that there was an assimilative capacity in waterbodies to detoxify and satisfactorily reduce exposures to these substances.

The harmony was a long time in being disturbed despite a growing body of knowledge that for these persistent toxic substances the system was not working, and I may have had a small hand in disturbing that harmony. In 1970, after working for 6 years on peregrine falcons in Northern Ireland and on various sea bird colonies around the British coastline, I started working on colonies of common terns in Hamilton Harbour and found an anomaly. There was an unprecedented incidence of congenital abnormalities such as crossed beaks as well as a high incidence of embryo mortality and growth retardation. Perhaps the most enduring image from that time was Karl Heinz Himmer's 1972 photographic study of a Caspian tern on Pigeon Island with a fledgling chick with a crossed beak. The photograph is symbolic of the anomaly and subsequent scientific crisis precipitated by undertaking field surveys and finding that somehow some source of pollution had not been controlled and that a teratogenic compound(s) had been released on, what was presumed to be, a massive scale. At the time, there was a frantic but unsuccessful search being undertaken by Gerry Bowes of the Canadian Wildlife Service for dioxins and dibenzofurans in Lake Ontario herring gull eggs. Field studies and laboratory incubation of samples of eggs confirmed the high embryo mortality, congenital abnormalities, and growth retardation. Based on the existing poultry literature, this led me to a diagnosis of an outbreak of "chick-edema disease." This was reported by the Water Quality Board in 1975 when it recommended a water quality objective for PCB to the International Joint Commission. In addition to the high levels of PCBs found by Lincoln Reynolds in the eggs in the early 1970s, Ross Norstrom and Doug Hallett eventually found high levels (3,000 ppt) of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) retrospectively in frozen samples when chemical analytical technology improved in the early 1980s. Other researchers, notably Tom Erdman and Tim Kubiak, found similar lesions in the late 1970s in Green Bay colonies of Forster's and common terns, and in 1983 attributed these to the coplanar PCBs. Later, Jim Ludwig extended their work to other species, notably the double-crested cormorant and to other locations in the upper lakes, and through his collaboration with John Giesy, showed the dose-response relationship between embryo mortality and dioxin equivalents.

A characteristic feature of all of this work was that it was received with extreme skepticism, and in some cases, with questions about the investigator's scientific competence and integrity. Many scientists are uncomfortable with even the mentioning of the subject, but I believe that there is much to be learned by studying this point of tension and trying to cast it in a sociological context. Great Lakes scientists are not the first to be vilified for their unconventional ideas, but if there is some way of comprehending what is occurring and how it will likely be resolved, the experience may be less disturbing for those who find themselves involved and the process may be more productively controlled.

Perhaps the most useful book that has been published on this topic is the volume by Thomas S. Kuhn entitled *The Structure of Scientific Revolutions*, published in 1962. The title gives a clue to the process that we are involved in. I would contend we are in the middle of a scientific revolution. Briefly, Kuhn stated that most scientists are involved in the elaboration of existing paradigms or bodies of knowledge, do not aim to produce major conceptual or phenomenal novelties, and that everything except the most esoteric detail of the result of their work is known in advance. However, in carrying out this "normal science," scientists may become aware of anomaly and recognize that nature has somehow violated the expectations derived from the existing paradigm. In 1970, the existing paradigm included eggshell thinning caused by DDT and metabolites and embryo and adult mortality due to dieldrin. When I went to the islands in Hamilton Harbour in 1970, I had an intuition that I was dealing with something quite different from anything that had been seen before in wild bird populations. The intuition came from the extensiveness of the nesting failure and from the presence of congenital abnormalities in the common tern chicks.

Kuhn noted that as well as not aiming to invent new theories, scientists are often intolerant of those invented by others. However, the persistence of anomaly leads the science to crisis, which in turn leads to a period of pronounced professional insecurity, characterized by resistance, before the emergence of new theories leads to a major paradigm shift. Kuhn has suggested that resistance serves a useful function to ensure that an existing paradigm is not surrendered too easily, that scientists are not lightly distracted, and that the research on the anomalies that leads to paradigm change will penetrate existing knowledge to the core.

However useful resistance may be, it can be a shattering experience for those who propose new theories or who report anomalies. The 1973 proposal of a diagnosis of chick-edema disease in the eggs of fish-eating birds led to the prediction of a series of associated pathologies including subcutaneous edema, pericardial edema, peritoneal edema, liver necrosis, hepatomegaly, and porphyria, all of which were confirmed in 1974. When, however, the same lesions were not found in 1975 by independent observers and no traces of dioxins or furans were detected by the analytical chemists, the diagnosis was discredited until TCDD was found 6 years later in 1981. A similar resistance is in progress at the moment concerning the use of dioxin toxicity equivalents for coplanar PCBs, thus slowing down acceptance of the significant correlations that Giesy and coworkers have established with respect to egg mortality on a colony basis. Resistance may serve a useful function, but for a scientific discovery with serious human health implications to take 20 years to penetrate the community would seem to be counterproductive and dangerous.

If we are involved in a scientific revolution, Kuhn's writings would indicate that the scientists committed to the competing paradigms would start to examine the rules that limit the nature of acceptable solutions and the steps by which they are to be obtained. To a large extent, Tom Muir and Ann Sudar working at the Canada Centre for Inland Waters at Burlington precipitated the scientific crisis with their review of what they believed toxic chemicals had done to a variety of organisms in the Great Lakes basin, including mink, otters, bald eagles, cormorants, various gulls and terns, snapping turtles, lake trout, and even beluga whales in the Gulf of St. Lawrence. But the most shocking evidence that was included in their review concerned the Jacobsons' evidence of perinatal effects on infants who had been exposed in utero because their mothers had eaten quite

small amounts of Lake Michigan lake trout prior to pregnancy. David Egar, the Canadian Co-chair, must be credited with bringing the question of how to link causes with effects to the Council of Great Lakes Research Managers, thereby starting the process that led to the first Cause-Effect Linkages Workshop in Chicago in March 1989. But it was Glen Fox of the Canadian Wildlife Service who was responsible for supplying the Council's workshop with the rules for inferring causality, and these were derived from the writings of Sir Karl Popper and from epidemiology. Briefly, the six main criteria were: time order; strength of association; specificity; consistency on replication; coherence; and predictive performance.

The ballots at the first Cause-Effect Linkages Workshop indicated a fair degree of skepticism among the participants about whether a causal linkage had been demonstrated for several of the case studies. This was probably not surprising since the material was intellectually complex and difficult to communicate in the time allocated. During the past $2^1/_2$ years, more people have become familiar with the wealth of evidence linking the developmental injury found in fish, wildlife, and humans to persistent toxic substances and particularly PCBs. At this second Cause-Effect Linkages Workshop, others will embrace this new paradigm and, over the next few years, it will become the accepted norm among scientists. Freed from the constraints of examining the rules, the research will likely proceed more efficiently. This will be a period of reconciliation between scientists who have previously been at odds over what to believe. There will inevitably be a few who will not make the transition and these will tend to be written out of the science.

New paradigms are accepted primarily because they can solve problems that the old paradigm was unable to do. They are also accepted because they enable predictions of previously unsuspected phenomena and finally because they are just aesthetically "prettier" than the old paradigm. In the research on the Great Lakes, the new scientific paradigm concerns the relationship between PCBs, 2,3,7,8-TCDD, and developmental abnormalities and embryo mortality in fish and wildlife, and growth retardation and behavioural and cognitive deficits in infants. The new paradigm can first explain what has occurred, and use of the scientific method has yielded reliable statements of causality. Second, the new paradigm can cut through the myriad of compounds known to be in commerce in the Great Lakes basin (probably well over 30,000) and the compounds that have been detected by environmental chemists (probably about 1,000) and isolate the critical compounds that have done the damage (probably about half a dozen). Third, this evidence can then be used as a basis for guiding future regulatory action and the amendments to correct the inadequacies of existing legislation. Finally, it can be used to predict that when all the sources of the critical compounds including DDT, dieldrin, HCB, PCB, and 2,3,7,8-TCDD are controlled, including those from outside the Great Lakes basin, these particular kinds of pathologies will no longer occur and species such as the lake trout will again be able to reproduce and sustain their populations and bald eagles will again be able to reestablish and breed on Great Lakes shorelines. More particularly, the contaminants in the fish, and specifically PCBs, will no longer cause behavioural and cognitive deficits in offspring of women who eat Great Lakes fish prior to pregnancy. These are the primary reasons why scientists will embrace the new paradigm in the next few years.

If the research has been difficult for the scientific community to accept, it has posed a much more profound shift for the regulatory community that was founded on Sprague's Tactic 1 (end-of-the-pipe) and Tactic 2 (water quality objectives). First, the forensic research based on biological field studies or epidemiological surveys has shown that despite implementation of costly conventional waste treatment, there has been actual injury to fish and wildlife populations and to human perinatal development. Second, by elucidating the scientific rules for inferring causality, risk assessment techniques that led to statements about potential effects have had to be replaced by injury or damage assessment techniques that lead to statements about actual effects and their causes. Third, the research has shown the priority of teratogenesis as a pathological process in the Great Lakes compared with the existing priority accorded carcinogenesis and risk assessment. It

is these factors, derived from the new paradigm, that shattered the harmony of the regulatory agencies responsible for water pollution control. Their science is mandated since it derives from a body of law that lays down how they should act, what the existing scientific and regulatory paradigm is, and how to conduct their regulatory procedures.

If there was a scientific crisis it is in the process of resolution. But for the regulatory community the crisis is only just beginning and it is only now starting to feel the effect of the new scientific paradigm that must penetrate to the politicians, the legislatures, the bureaucracies, and finally the industry. The pressure has largely been fuelled by a knowledgeable citizenry which has at last informed itself, through people such as Theo Colborn, about the forensic evidence, and which has organized itself for action such as was experienced at the 1989 Biennial Meeting of the International Joint Commission in Hamilton. Regulatory officials, who had spent entire careers in cleaning up the Great Lakes and were justifiably looking for an acknowledgement of their contribution, were instead badgered for the lack of progress in coming to terms with the policy contained, since 1978, in the Great Lakes Water Quality Agreement that stated that "the discharge of any or all persistent toxic substances be virtually eliminated." This policy, that should have derived its rationale and evidence from the forensic science, was unimplementable using existing water pollution control legislation in the two countries and will necessitate an overhaul of the mandate. The forensic scientists were looked upon by the existing regulatory institutions as radical and disruptive, which they are. And because the forensic evidence could not penetrate the existing pollution control institutions, it tended to be released to the public and to the media, further alienating the forensic scientists from the regulatory officials.

The forensic evidence has now penetrated these institutions and the methodology for inferring causality has left no space for ambiguity, negotiation, or manoeuvre. The regulatory authorities are now confronted with a comprehensive complaint about injury to health and resources caused by certain persistent toxic substances. This evidence implies a fiduciary duty of care on the part of the regulatory authorities and the comprehensiveness of the case leaves no basis for inaction without a concomitant compensible liability. It may be a long time before the forensic scientists will be forgiven by the regulatory community for precipitating this crisis with no familiar solution in existing legislation.

This second Cause-Effect Linkages Workshop through the efforts of the Michigan Audubon Society, is being held in association with the Sixth Biennial Meeting of the International Joint Commission. There is an enormous new body of knowledge that has been generated in the $2^1/_2$ years since the first workshop. The proceedings of the first workshop, thanks to the efforts of Steve Schneider, are available for everyone to examine and, if they feel so inclined, to critique. The regulatory officials responsible for pollution control are here and are eager to listen to the evidence. For the speakers at this workshop, there is an enormous opportunity to convert skeptical fellow scientists and the regulatory community to the new paradigm and thereby not only bring the seriousness of the situation regarding persistent toxic substances to their attention but also to legitimize J.B. Sprague's Tactic 3 involving biological field surveys and associated forensic research and to institutionalize it as a normal and necessary part of the pollution control processes of the two nations.

What, then, should I give you as a charge? First of all, cheer up! We are in the middle of a scientific revolution and at the beginning of a regulatory revolution. It is simultaneously uncomfortable and exhilarating, but out of it will come a more appropriate water pollution control regime for our two nations and what more challenging or worthwhile career could there be? Second, be open to perceptual change and to the possibility of commitment to the new paradigm and thereby to reconciliation with your former sparring partners. Third, remember that this science is the basis for political change in both countries and thus it must not only be reliable but also it must be publicly accessible. Finally, in the interests of Canadian official bilingualism, *Vive la Revolution Scientifique!*

FOX, G.A., Canadian Wildlife Service, National Wildlife Research Centre, Hull, Quebec K1A OH3, Canada.

1.3 Biomarkers: What Are They and What have They Told Us About the Effects of Contaminants on the Health of Great Lakes Wildlife?

The Great Lakes basin is a contaminated ecosystem. Since the early 1970s, monitoring data have shown the presence of hundreds of toxic chemicals in the air, water, sediments, and biota. There is a need to determine what, if any, effects the ambient levels of contaminants measured in various environmental media and animal tissues have on biological systems of sentinel wildlife populations. This can be determined through biological monitoring—the regular application of biological techniques and methods to assess the quality and condition of biological systems.

Biological monitoring allows us to (1) identify and quantify bioavailable contaminants, (2) determine if the integrated exposure to the host of chemicals in the environment results in a biological response and if the response has exceeded homeostatic capacity, (3) assess when contaminant-induced stress has detrimentally affected individuals, populations, or the structure and function of ecosystems, and (4) establish priorities for site clean-up and measure success of remedial actions.

In biological monitoring we deal with the concepts of stress, health, and disease. Stress may result from accentuation of some preexisting environmental influence or the introduction of some foreign agent into the system which results in the activation of homeostatic processes. Depending on its severity, sublethal stress may limit physiological systems, reduce growth, impair reproduction, predispose the individual to infectious diseases, and reduce the capacity of the organism to tolerate subsequent stress. We call any failure of normal homeostatic processes at any level of biological organization "disease." In studying the effects of stress and disease in biological systems it is apparent that there is a gradient, starting with normal health or homeostasis, progressing along scales of impairment and disability, ultimately to failure or death. Measures of impairment are more sensitive to contaminant effects than are measures of disability. Thus, monitoring impairment of physiological and behavioral responses will clearly provide early warning of the potential onset of disabilities and provide an understanding of the specific mechanism(s) by which health was impaired. In the past, we have often managed in a reactive fashion, responding to overt disease/disabilities such as mortality, reproductive abnormalities, deformities, and other gross manifestations of homeostatic failure. To be proactive, we must intervene early in the disability sequence. We must use more sensitive measures of impairment and sublethal endpoints. We call these sensitive sublethal indicators of impairment "biomarkers." Any change that is qualitatively or quantitatively predictive of health impairment resulting from exposure can serve as a biomarker. We have measured a number of physiological/biochemical biomarkers in Great Lakes wildlife, and in particular, birds.

Mixed function oxidase (MFO) enzymes add oxygen to a wide variety of compounds, rendering them more water soluble and thus more readily excreted. Their normal physiological role is metabolism of endogenous compounds such as steroids and prostaglandins. Exposure to PCBs and other contaminants may cause enzyme induction which may lead to the increased or altered metabolism of these endogenous compounds and other contaminants with production of more toxic metabolites. Elevated MFO activities have been found in the livers of embryos or newly hatched chicks of herring gulls (Lake Ontario and Saginaw Bay, 1981), Forster's terns, common terns, and black-crowned night herons (Green Bay and Saginaw Bay, 1983-85), and double-crested cormorants (Green Bay, 1988).

Deregulation of heme biosynthesis results in alterations in the size and/or composition of the porphyrin pool. Alterations in porphyrin patterns have been observed in livers of herring gulls

from colonies throughout the Great Lakes as recently as 1985. Alterations were most severe in Green Bay, Saginaw Bay, and Lake Ontario. In Lake Ontario, the alterations were more severe in 1974 than 1985.

Vitamin A (retinol) is necessary for vision, reproduction, and maintenance of differentiated epithelia and mucous secretions. Retinoid homeostasis (uptake, storage, and mobilization) is a highly regulated process but exposure to a variety of environmental contaminants is known to alter these processes in mammals and birds. Measurement of retinoids in herring gull livers as early as 1982 suggested that levels in the Great Lakes were generally lower than those in birds from the Atlantic coast. Retinoid levels are most abnormal in gulls from western Lake Erie, the Detroit River, and Lake Ontario. Depletion of retinoids was greater in 1985 than in 1982/83 in colonies in Lake Ontario, Saginaw Bay, and Green Bay.

The thyroid plays a very important role in many physiological processes, particularly those involving metabolism, development, differentiation, and growth. Herring gulls collected throughout the Great Lakes in 1974-1983 suffered from thyroid enlargement (goiter) and varying degrees of epithilial hyperplasia. Thyroids of gulls collected from western Lake Erie were most enlarged. Epithelial hyperplasia was not observed in the thyroids of gulls from the Atlantic coast but occurred in 11 of 12 collections from the Great Lakes and was most prevalent in gulls from Saginaw Bay, Green Bay, Lake Ontario, and western Lake Erie. It was most severe in the 1974 collection from Lake Ontario. Goiter is present regardless of whether the free iodide content of the plasma is similar to that of gulls from the Atlantic coast, suggesting that the endemic goiter in Great Lakes gulls is a result of exposure to goitrogenic substances rather than iodide deficiency. In Lake Ontario, the severity of goiter has decreased in more recent collections.

Chemicals capable of altering the integrity of DNA are present in the Great Lakes environment. Three measures to assess chemically-induced damage to genetic materials have been applied to the tissues of fish-eating birds from the Great Lakes. These are the frequency of sister chromatid exchange, the amount of DNA strand breakage, and the degree of DNA methylation. The only significant finding was that DNA from livers of young herring gulls collected from Lake Ontario in 1987 contained a significantly lower percentage of 5-methyl deoxycytidine than the livers of gull chicks from the Atlantic coast. However, these genetic studies were initiated after the peak of chemical contamination in the mid-1970s and after most gross health and biological effects were observed. It is therefore impossible to assess the role of genotoxic chemicals in the pathophysiological lesions observed in Lake Ontario in the early 1970s.

The prevalence of congenital malformations (birth defects) in fish-eating birds serves as a sensitive indicator of the levels of developmental toxins in the food web. Congenital malformations have been reported in the young of 10 species that are partially, if not totally fish-eaters. Several lines of evidence suggest that this is a chemically-, rather than a nutritionally-, microbially-, or genetically-mediated phenomenon. The maximum prevalence rates and the largest numbers of affected species have been in locations heavily contaminated with PCBs and related compounds, particularly Lake Ontario, Green Bay, and Saginaw Bay. Young with crossed beaks and other malformations continue to be found.

At least 11 wildlife species in the Great Lakes basin have experienced reproductive or other problems and/or population decreases since the 1960s that have been attributed to chemical contaminants. The list includes two mammals, eight species of birds, and one reptile. All are long-lived, fish-eating species.

Our studies of wildlife populations suggest that although conditions are improving and gross manifestations of contaminant toxicity are observed less frequently, biochemical changes indicate the presence of sufficient amounts of contaminants such as PCBs in forage fish to influence the

physiology of herring gulls over much of the Great Lakes basin. Our experience with wildlife suggests that more emphasis should be placed on studying effects on embryonic development, biochemical processes, and reproduction in humans who consume significant amounts of Great Lakes fish and wildlife.

LEATHERLAND, J.F., University of Guelph, Guelph, Ontario N1G 2W1, DONALDSON, E.M., Department of Fisheries and Oceans, West Vancouver, DOWN, N.E., B.C. Ministry of the Environment, Fort St. John, FLETT, P.A., University of Guelph, MOCCIA, R., University of Guelph, MUNKITTRICK, K.R., Department of Fisheries and Oceans, Burlington, SONSTEGARD, R.A., Tufts University, and VAN DER KRAAK, G., University of Guelph

2.1 Field Observations on Reproductive and Developmental Dysfunction in Introduced and Native Salmonids from the Great Lakes

The title of the presentation indicates that the paper will contain observations on the reproductive biology of both native and introduced salmonid species in the Great Lakes. In fact, there are only a few salmonid species that are *bona fide* native species; these include lake char (trout), brook char (trout), cisco, and whitefish. Because of habitat destruction, pollution, and overfishing, many of these populations have been severely depleted in recent years, particularly in the lower Great Lakes region, and stocks are maintained by artificial culture and restocking programs. The Atlantic salmon, which used to be native to Lake Ontario, has not been found in the Great Lakes in the last decade or more, and attempts to re-introduce the species have been largely unsuccessful. Since there have been no systematic studies of the reproductive physiology of these species, data from native species are not available.

The most numerous salmonid species, in terms of biomass and number of species, are introduced (exotic) species such as rainbow trout, brown trout, and the Pacific salmon (coho, chinook, pink, chum). The success of the introduced species has depended on expensive and extensive fish culture and stocking procedures; even these have failed to maintain sufficient numbers of certain species in some lakes, and the stocking programs have been disbanded. Most of our work has focused on coho salmon, and to a lesser extent chinook and pink salmon.

Since their introduction into the Great Lakes beginning in the early 1960s, Pacific salmon have exhibited a variety of endocrine and reproductive problems, including massive thyroid lesions, low egg thyroid hormone content, high prevalence of sexual precocity, poor expression of secondary sexual characteristics, low blood estrogen and progesterone levels in females, low rates of egg fertility, and low rates of survival to hatch, often associated with high embryonic deformity.

Thyroid lesions have been found in every adult Pacific salmon taken from the Great Lakes since 1974 when we began the study. In the 1970s, it was argued that the lesions represented an iodine deficiency condition because of the land-locked condition of the introduced stocks. Subsequent work in our laboratory has shown that the iodide content of most Great Lakes salmon stocks is similar to that of fish taken from the Pacific Ocean, and that waterborne goiterogenic factors are the most likely cause of the condition. Organochlorine xenobiotics do not seem to be etiologic agents in the salmon, although these xenobiotics did induce thyroid lesions in rodents that were fed diets containing Great Lakes salmon. The role of the thyroid dysfunction of Great Lakes salmon on the reproductive problems outlined below remains unresolved.

Salmon eggs contain large amounts of thyroid hormone that appears to be of maternal origin. It has been speculated that the hormone is important for the control of embryogenesis, particularly

during the period prior to the secretion of thyroid hormones by the embryo itself. Coho salmon eggs of Lake Erie, and to a lesser extent Lake Ontario stocks, contain considerably lower levels of thyroid hormone than comparable stocks from British Columbia. The significance of these low levels in terms of embryonic development, survival to hatch, and survival to adult stages has yet to be fully explored.

The causes of sexual precocity (sexual maturation, particularly of the males, one or sometimes two years before the expected time of maturation) of salmonid stocks is still not understood. From a management perspective, precocious maturation reduces the number of available adult males, and the fish are undesirable as a sportfishing resource. There is evidence to suggest that there is a genetic basis for the problem. However, the evidence is often less than convincing, and environmental etiologies have been proposed. To put the Great Lakes situation into some perspective, returns of precociously mature males in stocks of salmon in British Columbia are of the order of 2-5%; in Lake Erie, the returns since 1980 have been in excess of 40%, and as high as 90%. These values are difficult to explain on the basis of a genetic etiology alone, especially since the salmon stocks introduced into Lake Erie are from the same stocks that are found in Lake Michigan.

Some loss of secondary sexual characteristics, particularly by males, is a feature associated with fish breeding programs. Lack of conspecific competition for mating partners tends to reduce the expression of the hooked jaw ("kype") of the male that is used for territorial fighting in the spawning areas, and the brightly colored flanks of the males that appear to be sexual attractants used by the females for mate selection. The hormonal control of the formation of secondary sexual characteristics in salmon is not understood; it is assumed to be regulated by male sex steroids, such as 11-ketotestosterone, although such a simple relationship does not appear to be true, and further work on the basic biology of the species is needed to explain the phenomenon. This loss of sexual dimorphism is particularly marked in the Great Lakes stocks, even in the self-reproducing stocks studied in Lake Erie. While there is certainly a genetically-controlled component (possibly operating via an endocrine pathway), the severe loss of sexual dimorphism in some of the Great Lakes stocks strongly suggests an environmental etiology.

Recent studies by two of my collaborators (Munkittrick and Van Der Kraak) have shown marked changes in blood sex steroid hormone levels in white suckers taken from areas adjacent to paper mill effluent discharge; such measurements provide a sensitive marker for environmental monitoring. Adult female coho salmon from some Lake Erie stocks have extremely low plasma estradiol and dihydroxyprogesterone levels; these might explain the extremely low egg fertilities, marked prevalence of over-ripe eggs still attached to the ovarian membranes, and general ovulatory problems of the stocks. Similarly, the high prevalence of embryonic abnormalities may well be also related to the array of ovarian problems expressed by these females.

The paper will a) describe the reproductive (or developmental) problem, b) examine the evidence for and against a possible environmental etiology, c) outline deficiencies in current informational levels, and d) outline future research needs in this discipline.

Egg hatchability was negatively correlated with total PCBs ($R^2 = 0.48$, N = 24). Using TCDD equivalents rather than total PCBs resulted in a weaker correlation ($R^2 = 0.27$). No other measure of reproductive success correlated with any expression of PCB concentration or with egg lipid content.

Our results show that maternally transferred PCBs reduce the hatchability of eggs of feral Lake Michigan lake trout. Surprisingly, total PCB concentration accounted for a greater percentage of the variation in hatching success than TCDD equivalents. Analysis of individual PCB congeners, particularly those that are potent inducers of AHH activity, was expected to improve interpretation of PCB residue data and its toxicological significance. Two factors may be responsible for the current shortcomings of using TCDD equivalents. First, the TEF values that we use which assign the potency of individual congeners in calculating TCDD equivalents are based on activity of these compounds in mammals and birds. The responses of fish to various congeners may be different. For example, enzyme induction did not occur in scup (*Stenotomus chrysops*) exposed to congeners 105 and 118, two of the most potent mono-ortho substituted PCBs. The other factor is that the measures of reproductive success used in this paper are for the most part nonspecific. Distinct periods of mortality were observed within the embryonic development of eggs, most notably an early mortality that occurred through epiboly stages, and a second period that occurred just before hatching and was associated with ruptured egg membranes. Also, different types of terata were observed that included spinal deformities and yolk-sac edema (blue-sac disease). Efforts to examine these specific responses in relation to PCB residues is currently underway.

Fry survival in lake trout is controlled by a severe mortality syndrome that affects fry just before swim-up. The timing of this syndrome and the associated behavioral signs are similar to those observed in salmonids in the 1960s and attributed to DDT poisoning. DDT has declined dramatically in fish from the Great Lakes and we could find no evidence of PCB involvement with this mortality.

GIESY, J. P. Department of Fisheries and Wildlife, Pesticide Research Center, Institute for Environmental Toxicology, Michigan State University, East Lansing, Michigan 48824 and TILLITT, D. E. National Fisheries Contaminant Research Center, U.S. Fish and Wildlife Service, Columbia, Missouri 65201

2.3 The Role of Planar Halogenated Hydrocarbons in the Reproductive Problems of Fish-Eating Waterbirds of the Great Lakes

Certain populations of fish and wildlife living in the Great Lakes region have had demonstrable reproductive problems. Although these are not the only symptoms observed in Great Lakes fish and wildlife, these are the symptoms which have alerted biologists to investigate the possible causes of the continued problems. Fish-eating waterbirds are one such group. Fish-eating waterbirds of the Great Lakes have exhibited a variety of reproductive anomalies, including embryotoxicity and teratogenesis. The teratogenic symptoms at elevated levels in certain populations of Great Lakes waterbirds include: crossed-bill; clubbed foot; exophthalmia; gastroschisis; subcutaneous edema; pericardial edema; and hydrocephaly. The spatial distribution of these anomalies and their rates of occurrence closely follow the occurrence and degree of environmental contamination by persistent planar halogenated hydrocarbons (PHHs). PHHs include, among others, polychlorinated biphenyls (PCBs), dibenzo-p-dioxins, and dibenzofurans. Although no longer legally manufactured in North America and banned from use in the late 1970s, PHHs remain in the Great Lakes ecosystem. Due to the controls placed on production and use of PHHs in North America, concentrations of PHHs (in particular PCBs) have decreased in the biota

of the Great Lakes. This decrease in the concentrations of PCBs has proceeded according to a first-order exponential decay curve, and currently they are at or near the asymptote of the curve. Therefore, the concentrations of PCBs in the biota are not expected to decrease substantially for quite some time. The problem, however, is that many of the PHH-associated, dioxin-like symptoms in fish-eating waterbirds continue to persist. This paper discusses our studies to define the occurrence of these reproductive anomalies in Great Lakes fish-eating waterbirds, the role of PHHs in the anomalies, and the reason for the persistence of the symptoms.

LUDWIG, M.E., LUDWIG, J.P., AND AUMAN, H.J., Ecological Research Services, Inc., 2395 Huron Parkway, Ann Arbor, Michigan 48104; and JONES, P., GIESY J.P., and VERBRUGGE, D., Michigan State University Center for Environmental Toxicology, East Lansing, Michigan 48824.

2.4 Differences in Contaminant Levels in Great Lakes vs. Inland Fish Species

Significant differences in concentrations of planar polychlorinated hydrocarbons (PCHs) have been recorded between gamefish species living in Great Lakes influenced waters and those occurring in the inland land-locked waters of Michigan. Fish from the Muskegon, Manistee, and AuSable rivers were sampled above and below hydroelectric dams to determine concentration differences between similar species. Dams on each of these rivers serve as barriers, and separate Great Lakes-influenced fish populations from those occurring above the dams. This study was designed to quantify the differences in mercury, pesticide organochlorine, and planar halogenated hydrocarbon concentrations between similar species of fish above and below dams on three Michigan rivers.

Electrofishing was used to collect five individual fish of similar lengths (within 5 inches) of the same or closely related species from isolated inland vs. at-large Great Lakes populations. Field collections were completed between 12 September, 1989 and 15 May, 1990. Three species from each location were analyzed except for the AuSable River where four species were analyzed. Manistee River species/genus sampled included steelhead/brown trout, northern pike, and white sucker. Muskegon River sampled species included carp, yellow perch, and walleyed pike. AuSable River sampled species included chinook salmon/brown trout, white sucker, northern pike, and walleyed pike. Since chinook salmon and steelhead do not occur above dams on the AuSable and Manistee rivers, brown trout were substituted. Once matched samples were obtained from each location they were selected for analysis. Five individual matched fish were homogenized by passage though a large commercial meat grinder. Homogenization aliquots were placed in chemically-cleaned jars and stored at -20°C until analysis. Samples were separated into three identical sets, two for analysis and one for long-term archive.

Samples were analyzed using both the Ethoxy-resorufin-o-deethylase (EROD) bioassay and by PCB congener-specific analysis. Additionally, pesticide scans for organochlorines and mercury were performed. The bioassay measured the ability of the extracted samples to induce cytochrome P 450 1A1-associated EROD activity in H_4IIE rat hepatoma cells as compared to 2,3,7,8-TCDD (dioxin). The H_4IIE, bioassay-derived TCDD equivalents (TCDD-EQ) were used to compare the induction ability of each set of matched, above and below dam samples. Congener-specific PCB analysis utilized direct measurement of specific congeners, particularly the planar-halogenated congeners for comparison purposes between matched species. Wet weights of specific PCB congeners were recorded and then converted to toxic equivalents using factors derived by Safe (1990). Both the EROD bioassay and congener-specific PCB analysis methods were utilized in tandem to attempt to overcome the shortfalls of a singular analysis protocol. Pesticide scans revealed wet weight concentrations for any other organochlorines and mercury present in the samples.

Fish occurring below the dams in Great Lakes influenced waters showed significantly greater concentrations of specific PCB-congeners than similar species occurring above the dams. Great Lakes-influenced waters also showed higher EROD enzyme induction rates than their upstream counterparts. Manistee River fish showed EROD enzyme activity ratios from above- to below-dam species ranging from at least 1 to 6.6 in white suckers to at least 1 to 26.8 in salmonids. Muskegon River fish EROD induction ratios ranged from at least 1 to 7.6 in perch to at least 1 to 26.8 in carp. AuSable River fish revealed EROD induction ratios from at least 1 to 4.3 in white suckers to at least 1 to 13.3 in salmonids. Concentrations of DDT, other pesticides showing up in minute amounts, and mercury were not significantly different between fish species occurring above and below dams.

Higher concentrations of organochlorines, particularly planar halogenated hydrocarbons (co-planar PCBs) from below-dam (Great Lakes-influenced) fish may suggest a possible explanation to differences in piscivor reproductive ability. Eagles and mink occurring on Great lakes influenced waters have shown much lower reproductive rates than those living on landlocked "inland" waters. Differences in organochlorine contamination in fish occurring above and below dams seem to parallel top picsivor reproductive ability in sensitive species of mammals and raptors.

TILLITT, D.E., National Fisheries Contaminant Research Center, U.S. Fish and Wildlife Service, Columbia, Missouri 65201, GIESY, J.P., and JONES, P.D., Department of Fisheries and Wildlife, Pesticide Research Center, Institute for Environmental Toxicology, Michigan State University, East Lansing, Michigan 48824 and, ANKLEY, G.T., U.S. Environmental Protection Agency, Environmental Research Laboratory - Duluth, Duluth, Minnesota

3.1 TCDD-EQ in Colonial Waterbird Eggs from Great Lakes Colonies (1986-1990) as Measured by the H_4IIE Bioassay and Chemical Residue Analysis

Planar halogenated hydrocarbons (PHHs) continue to be of concern in the Great Lakes aquatic ecosystem. Concentrations of PHHs in the Great Lakes biota have decreased to an apparent steady state with the expectation of little further reduction. However, the dioxin-like symptoms associated with PHHs in Great Lakes fish and wildlife have persisted. The determination of the role of PHHs as a causal agent of these symptoms is confounded due to the changes in the chemical patterns among locations and species, along with a lack of a complete understanding of the toxicological interactions among PHHs and their modulators.

Our investigations have been into the spatial and temporal trends of certain PHHs in fish-eating colonial waterbird eggs taken from the Great Lakes and measured by a novel *in vitro* bioassay, the H_4IIE rat hepatoma cell bioassay. The results of this bioassay are given as 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents (TCDD-EQ) and incorporate the interactions of synergism, antagonism, and/or additivity which are known to occur among PHHs and the modulators of their toxicity.

An alternative method for the calculation of TCDD-EQ is that of chemical analysis and an additive model of toxicity. Comparison of these two methods for the determination of TCDD-EQ can be a useful way to obtain a better understanding of the toxicology of complex mixtures and the relative importance of individual compounds in these mixtures. Studies presented here detail the temporal and spatial trends in TCDD-EQ in Great Lakes waterbird eggs, compare the two methods of TCDD-EQ determination, and discuss the toxicological implications of these measurements to Great Lakes fish-eating waterbirds.

LUDWIG, J.P., LUDWIG, M.E., AND AUMAN, H.J., ECOLOGICAL RESEARCH SERVICES, INC., 2395 Huron Parkway, Ann Arbor, Michigan 48104; and Jones, P., Giesy J.P., and Verbrugge, D., Michigan State University Center for Environmental Toxicology, East Lansing, Michigan 48824

3.2 Uptake of Toxic Chemicals from Great Lakes Fish by Double-crested Cormorant and Herring Gull Chicks

Chicks captured in eight colonies (seven double-crested cormorant, one herring gull) spread geographically from western and eastern Lake Superior, across Green Bay, northern Lake Michigan, and four Lake Huron sites extending from the Straits of Mackinac to Saginaw Bay were tested for their pattern and levels of uptake of toxic chemicals during 1989. This research was designed to measure differences in the local patterns of toxic chemical contamination and to determine whether cormorants were useful conservative models of fish to consumer biomagnification processes and rates. Waterbird uptake patterns were then used to monitor the local flows of planar toxic contaminants through fish to warm-blooded vertebrates using the conceptual model of biomagnification from forage fish to birds. Toxicity was estimated two ways on the same samples of forage fish recovered from birds, bird eggs, and four age-classes of chicks: Ethoxy-resorufin-o-deethylase (EROD) enzyme induction by planar halogenated hydrocarbons was estimated as TCDD (dioxin) equivalents in ppt wet weight and PCB congeners were measured to 0.1 to 0.01 ppb levels using carbon column and advanced GC/MS technologies on the same samples. Nineteen enzyme-inducing PCB congeners were measured and the sum of their toxicities was then calculated using Safe's (1990) TCDD toxic equivalency factors for PCBs. The average forage fish to bird egg biomagnification factor (BMF) was estimated at 23.2 by EROD and 23.7 by PCB congeners. Cormorants averaged a BMF of 21 by both methods for seven colonies, but the herring gull BMF was near 41 for a Saginaw Bay colony. Congener-specific analysis suggested that cormorants have a much greater ability to degrade certain very toxic PCB congeners, especially PCB 77, than gulls, a finding consistent with other data sets for members of the gull and tern family (Laridae).

In their first 5 weeks of life, large cormorant chick uptake rates measured by EROD bioassay varied 11 fold from the Apostle Islands area (a whole body concentration of 14.1 TCDD-EQ) to Green Bay (162.4 TCDD-EQ); Saginaw Bay herring gulls accumulated 384.7 TCDD-EQ. Chick uptake curves, calculated from four sizes of chicks collected at each site, were exponential over time in the more contaminated areas (Green Bay, Saginaw Bay, and northern Lake Michigan), but linear elsewhere. Forage fish samples recovered from bird regurgitations at five colonies in Lakes Michigan and Huron varied from 5.6 TCDD-EQ to 24.4 TCDD-EQ by bioassay, while large cormorant chicks at an average age of 35 days had accumulated from 28.3 to 162.4 TCDD-EQ in the same colonies, an average BMF of 5.6. Because cormorants have cold-blooded chicks which are altricial until 10-14 days post hatch and are large-bodied birds with relatively lower standard metabolic rates, it is believed that the cormorant chick uptake rates represent the minimal uptake rates for all Great Lakes colonial waterbird chicks. By comparison, the 38-day-old herring gull chicks, which are warm-blooded and precocial, from a Saginaw Bay colony ate fish with 13.4 TCDD-EQ and accumulated 384.7 TCDD-EQ, an average BMF of 28.7. This BMF probably reflected their smaller size, greater standard metabolic rate, precocial habit, and their lower capacity to degrade some of the more toxic congeners, especially PCB 77, when compared to cormorants.

From these accumulation data and previous studies on the relationship of egg mortality in Great Lakes cormorant colonies and their egg toxicity burdens as TCDD-EQ, the number of days required to accumulate a lethal concentration for 50% of eggs (LCE 50) laid due to planar toxic contaminants in each of the eight areas sampled was calculated to range from 1,438 days in the Apostle Islands in Lake Superior to 128 days in Green Bay in cormorants and just 54 days in

Saginaw Bay herring gulls. The calculations for Saginaw Bay herring gulls matched the field observations in Saginaw Bay Caspian terns from 1988 where 48-day exposure to local fish resulted in eggs laid which developed 48% grossly abnormal embryos and 60% egg death during incubation. These field observations confirmed that cormorant chicks are both a sensitive barometer of local contamination and a very conservative species to use to model impacts on other homeothermic wildlife species. Smaller bird and mammal species with much greater standard metabolic rates should accumulate toxic residues much more rapidly.

From six years of field studies on egg mortality rates, published egg injection studies, and EROD bioassay data, a model of uptake and expected waterbird egg mortality was constructed based on measured TCDD-EQ (ppt wet weight) in the forage fish the birds ate and eggs they laid. These data suggest that the lowest levels found in Great Lakes forage fish (5.6-24.4 TCDD-EQ) would cause waterbirds with a BMF of 21 to produce eggs with 118 ppt TCDD-EQ (eggs of Lake Superior cormorants were measured at 141-165 TCDD-EQ in 1986-1989) upwards to 512 TCDD-EQ (Saginaw Bay herring gull eggs in 1989 were measured at 557 TCDD-EQ; 1987 year late-clutch Caspian tern eggs were measured at 628 TCDD-EQ). Further, if bald eagles consumed a 90% fish/ 10% Great Lakes waterbird diet with the mean levels found in this study and a BMF of 21, then eagles would produce eggs with 921 TCDD-EQ (two bald eagle eggs were recently measured at 560 and 1,065 TCDD-EQ). For reference, the measured concentrations of actual TCDD that produced a 50% death of white leghorn chicken eggs was 140 ppt, and 1,000 ppt killed 100% of white leghorn eggs. Further, the finding that a dietary concentration of about 3.6 TCDD-EQ per kg per day in ranch mink (a lowest observed effect level) leads to deleterious reproductive effects confirms that the least contaminated of Lakes Michigan and Huron forage fish tested at 5.6 TCDD-EQ are a likely hazard to reproduction of most wildlife species. Since no-effect levels are typically projected at one-tenth of the lowest observed effect level, a TCDD-EQ below one in forage fish must be achieved in order to protect wildlife, but the lowest forage fish toxicity found was five fold above this critical level. The average whole body level reported in Great Lakes fish of a variety of trophic levels at 18 TCDD-EQ is more than an order of magnitude over the no-effect level for sensitive wildlife.

HEATON, S.N., AULERICH, R.J., BURSIAN, S.J., Department of Animal Science and Institute for Environmental Toxicology; GIESY, J.P., Department of Fisheries and Wildlife and Institute for Environmental Toxicology; RENDER, J.A., Department of Pathology; Michigan State University, East Lansing, Michigan; TILLITT, D.E., U.S. Fish and Wildlife Service, Columbia, Missouri 65201; and KUBIAK, T.J., U.S. Fish and Wildlife Service, East Lansing, Michigan 48823.

3.3 Reproductive Effects of Feeding Saginaw Bay Source Fish to Ranch Mink

Planar polychlorinated hydrocarbons (PCHs) have been implicated as a cause of the declining populations of wild mink and otter throughout the Great Lakes region. The objective of this study was to determine the effects of PCB-contaminated prey on the reproductive performance and survival of mink. Carp collected from Saginaw Bay, Michigan, containing 8.4 ppm total polychlorinated biphenyls (PCBs), were substituted for marine fish at levels of 0, 10, 20, or 40% of the diet. The H_4IIE rat hepatoma cell bioassay was used to determine TCDD equivalents (TCDD-EQs) of the complex mixture of PCHs in the carp to predict the relative potency of the dietary PCHs.

Experimental diets were fed to both male and female mink beginning 7 weeks prior to breeding and continuing until the young were approximately 7 weeks old. Males continued to be fed the treated diets for an additional 4 weeks. The quantities of PCBs and TCDD-EQs ingested by the

female mink fed 0, 10, 20, or 40% carp over the first 85 days of the trial were 0.004, 0.16, 0.30, and 0.39 mg PCBs/mink/day and 0.27, 4.23, 7.86, and 12.13 ng TCDD-EQs/mink/day, respectively.

Feed consumption for both sexes and body weight gains for females were inversely proportional to the PCB content of the diet. Expressed as a percentage of brain weight, the weight of the liver, spleen, and lungs of females significantly increased in a dose-dependent manner. Histopathologic examination of the livers showed periportal and vacuolar hepatocellular lipidosis in the females fed 40% carp. Liver hypertrophy was attributed to an increase in lipid content within the liver cell cytoplasm. In addition to splenomegaly, the animals in the 20% dose group had spleens that were irregular in shape and more firm than in controls.

Total hepatic PCB concentrations in the carp-fed groups were significantly different in a dose-dependent manner from control values. However, a similar dose-related increase in hepatic TCDD-EQs was not observed. Total PCB concentrations in the mink livers ranged from 0.07 µg PCB/g in controls to 10.6 µg PCB/g in the 40% carp group. The hepatic TCDD-EQ ranged from undetectable for controls to 950.1 pg TCDD-EQs/g in the 40% carp group.

The diets containing Saginaw Bay carp caused impaired reproduction and/or kit survivability. Females fed 40% carp whelped the fewest number of kits, all of which were stillborn or died within 24 hours. Kit survival and average kit body weights in the 10 and 20% carp groups were significantly reduced at 3 and 6 weeks of age compared to the controls. Four stillborn kits from the 40% carp group had localized edema in the head and tail region. In addition, the mandible and maxilla of one of these kits were reduced in size. Brain, liver, spleen, kidney, heart, and lung weights in 6-week old kits decreased in a dose-dependent manner.

A Lowest Observed Adverse Effect Level (LOAEL) of 0.134 mg PCBs/kg body weight/day (3.60 ng TCDD-EQs/kg body weight/day) was determined. The relative potency (the ratio between TCDD-EQs and total PCBs) of the control diet was approximately twice that of the 10, 20, and 40% carp diets (68.7, 27.0, 26.2, and 31.6, respectively). However, in the liver, potencies were 55.5, 239.0, 151.7, and 130.6 in the control, 10, 20, and 40% carp groups, respectively, suggesting that the animals were retaining the more toxic congeners. Interestingly, there was a significant inverse relationship between dietary carp content and hepatic relative potencies.

The results of this study confirm the extreme sensitivity of mink to PCBs and lend support to the suspicion that PCBs are responsible for the marked decline in mink populations in certain areas adjacent to the Great Lakes.

BÉLAND, P., St. Lawrence National Institute of Ecotoxicology, 3872 Parc-Lafaontaine, Montréal, Quebéc H2L 3M6, DEGUISE, S., St. Lawrence National Institute of Ecotoxicology and Faculté de Médecine Vétérinaire, St-Hyacinthe, CP 5000, St-Hyacinthe, Québec J2S 7C6, GIRARD, C., Faculté de Médecine Vétérinaire, St-Hyacinthe, CP 5000, St-Hyacinthe, Québec J2S 7C6, LAGASE, A., Faculté de Médecine Vétérinaire, St-Hyacinthe, CP 5000, St-Hyacinthe, Québec J2S 7C6, MARTINEAU, D., St. Lawrence National Institute of Ecotoxicology, 3872 Parc-Lafaontaine, Montréal, Quebéc H2L 3M6 and New York State College of Veterinary Medicine, Cornell University, Ithaca, New York 14853, MICHAUD, R., St. Lawrence National Institute of Ecotoxicology, 3872 Parc-Lafaontaine, Montréal, Quebéc H2L 3M6, MUIR, D., Fisheries and Oceans Canada, Freshwater Institute, 501 University Crescent, Winnipeg, Manitoba R3T 2N6, NORSTROM, R., Environment Canada, Canadian Wildlife Research Center, 100 boul. Gamelin, Hull Québec, PELLETIER, É., INRS-Océanlolgie, 310 allée des Ursulines, Rimouski, Québec G5L 3A1, RAY, S., Toxicology Section, Science Branch, Fisheries and Oceans Canada, P.O. Box 5667, St John's, Newfoundland A1C 5X1, and SHUGART, L., Environmental Sciences Division, Oak Ridge National Laboratory, P.O. Box 2008, Oak Ridge, Tennessee 37831

3.4 Toxic Compounds and Health and Reproductive Effects in St. Lawrence Beluga Whales

An epidemiologic study was carried out over a period of 10 years on an isolated population of beluga whales *(Delphinapterus leucas)* residing in the St. Lawrence estuary (Québec, Canada). More than 100 individual deaths were aged, autopsied, and analyzed for toxic compounds. Arctic belugas and other species of whales and seals from the St. Lawrence were examined as controls. The St. Lawrence population was also surveyed for size and structure.

Population Dynamics: A mortality curve drawn from aged deaths showed an increase in mortality past age 14. Pregnancy rates of autopsied females were much lower than in the Arctic. Surveys during the study showed that the overall population size appeared to be stable. Modelling showed this stable pattern to result from low calf production and/or low survival to adulthood.

Toxicology: Various tissues were analyzed for mercury, lead, cadmium, PCBs, DDT, mirex, dioxins, furans, PAGs, benzo[a]pyrene metabolites, and PAH metabolites. St. Lawrence belugas had much higher levels of all compounds than Arctic belugas except for dioxins, furans and PAH metabolites, which were equivalent, and cadmium which was much lower. The interspecies comparison in the St. Lawrence showed that levels of PCBs and DDT were inversely related to specific body size in cetaceans. Major contributing factors are thought to be metabolic rate, diet, and trophic position, compounded by length of residence in the St. Lawrence basin. St. Lawrence belugas had much higher levels of total organochlorines than predicted from their body size. Levels increased consistently with age in both males and females, even though unloading by females through the placenta and/or lactation was evidenced by high burdens in some calves. Contrary to what was expected on the basis of mirex levels, no PCDDs and only low levels of some PCDFs (all not fully substituted at the 2378- positions) were detected in St. Lawrence belugas. The pattern of PCB congeners and homologs in belugas differed considerably from a mixture of Aroclor standards. Proportions of toxic non-ortho (coplanar) PCBs were low relative to other species. Tetra-, penta-, and hexachloro congeners predominated in Arctic animals, whereas hexa- and heptachloro congeners predominated in St. Lawrence males. In the latter, four congeners substituted at the 2,4- and 2,4,5- positions on both phenyl rings accounted for a higher proportion of total PCBs. Differences may reflect greater atmospheric transport of more volatile congeners as well as higher metabolic activity in St. Lawrence males due to MFO induction. At least ten different PCB methylsulphone metabolites were detected in St. Lawrence belugas. Levels of B[a]P adducts to DNA in St. Lawrence belugas approached those found to be carcinogenic in small laboratory animals. No B[a]P adducts were detected in Arctic belugas, which had, however, similar levels of aromatic DNA adducts as measured by 32_P post labelling.

Pathology: Although blubber thickness varied considerably, St. Lawrence belugas were not emaciated, as blubber to total body weight ratios were the same as in animals killed in the Arctic. Major findings from 45 necropsies of St. Lawrence belugas were a high incidence of tumors (35% of animals examined) including nine malignant neoplasms; a high incidence of lesions to the digestive system (53%), to the mammary glands (45% of adult females) and to other glandular structures (11%); there was frequent evidence of immuno-suppression; tooth loss and peridontitis were common; two animals had severe ankylosing spondylosis and another animal was a hermaphrodite. No such lesions were observed in 36 necropsies of other marine mammals belonging to six species of seals and cetaceans found dead in the St. Lawrence, and of five belugas killed in the Arctic.

The high incidence of various severe and chronic lesions readily distinguish St. Lawrence beluga whales from other cetaceans and pinnipeds from the same basin and from elsewhere. Although it has not been demonstrated clinically, a direct cause and effect relationship is likely to exist between specific toxic compounds present in tissues and the health and reproductive status of this whale population. Among them, B[a]P is the only potent carcinogen, while reproductive impairment, glandular dysfunction, and immuno-suppressive problems could originate from a mixture of organochlorines. It would seem that the future of St. Lawrence belugas is linked to that of the Great Lakes basin. The more abundant toxic compounds found in the whale tissues are all on the short list of "critical chemicals" in the Great Lakes. A mass balance based on mirex levels shows that at least half of the total organochlorines in the beluga population originate directly from the Great Lakes through migrating eels.

KUBIAK, T.J., U.S. Fish and Wildlife Service, East Lansing, Michigan, 48823

3.5 A Review of Bird Egg Toxicity Studies with Planar Halogenated Hydrocarbons

In the decade of the 1980s, better methodology was developed for the analysis of Planar Halogenated Hydrocarbons (PHHs). These PHHs include three families of structurally similar compounds, the polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans, (PCDFs), and the polychlorinated biphenyls (PCBs). Analysis of these compounds from environmental samples, especially Great Lakes fish-eating colonial waterbirds, was undertaken by the U.S. Fish and Wildlife Service and other organizations. A number of "dioxin-like" congeners (those congeners whose chlorine substitution and effects which are similar to 2,3,7,8-tetrachlorodibenzo-p-dioxin or TCDD) were identified and quantified in eggs of birds nesting on the Great Lakes. The purpose of this review is to 1) summarize what is known about the dioxin-like congeners of these families relative to toxicological and biochemical effects and relative potencies in birds and mammals and 2) provide a comparison with concentrations that have been found in Great Lakes bird eggs.

Although the field data are not abundant, there is clear evidence that PCB congeners 126 (3,3',4,4',5-pentachlorobiphenyl) and 77 (3,3',4,4'-tetrachlorobiphenyl) are the most important congeners relative to toxicological threats to avian reproductive success on the Great Lakes. This appears to be the case when the relative hazard is expressed as a "Hazard Quotient" (the wild bird egg concentration divided by the No Observable Adverse Effect Level or NOAEL; or the wild bird egg concentration divided by the LD_{50}, or the lethal dose to 50% of the test population) on an individual congener basis. When these congener concentrations are converted into a common "unit of measure" termed 2,3,7,8-TCDD Equivalents (TCDDeq), and compared to the NOAEL for actual 2,3,7,8-TCDD either individually or when summed together assuming an additive model of toxicity, the results show a similar degree of hazard.

Examples of Great Lakes bird egg contamination with these compounds exist for the Forster's, common, and Caspian tern, the double-crested cormorant, and the ultimate predator, the bald eagle. Using the above Hazard Quotient approach, the PCB contamination is clearly excessive when when viewed relative to the most sensitive endpoint (embryotoxicity) in the most sensitive species (white leghorn chicken).

SUMMER, C.L., AULERICH, R.J., AND BURSIAN, S.J., Department of Animal Science and Institute for Environmental Toxicology; GIESY, J.P., Department of Fisheries and Wildlife and Institute for Environmental Toxicology; RENDER, J.A., Department of Pathology, Michigan State University, East Lansing, Michigan; and KUBIAK, T.J., U.S. Fish and Wildlife Service, East Lansing, Michigan 48224.

3.6 Preliminary Analysis of Reproductive Effects from Feeding White Leghorn Chickens Saginaw Bay Source Fish

Recent studies indicating reduced reproductive success of colonial waterbirds nesting in the Great Lakes basin link polychlorinated hydrocarbon (PCH) contamination to increased incidence of terata and decreased hatching rates in these species. These observed effects are correlated with measured PCH concentrations and 2,3,7,8-tetrachlorodibenzo-p-dioxin equivalents (TCDD-EQs) in the adult birds. This study was designed to determine the reproductive effects of feeding Saginaw Bay source fish, an environmentally-derived PCH mixture, to white leghorn hens as a surrogate species for colonial waterbirds.

Sixty hens, divided into three treatment groups of twenty hens each, were fed diets containing 35% fish formulated to provide 0, 20, and 200 ppt TCDD-EQs. Saginaw Bay carp containing 8.4 ppm polychlorinated biphenyls (PCBs) was incorporated into the diets at levels of 0 (control), 3.5, and 35%. "Uncontaminated" ocean fish was added to the control and 3.5% carp diets at 35 and 31.5%, respectively. The control diet was fed to all birds for 2 weeks after which the birds were placed on the appropriate treatment diet for 8 weeks.

Egg production and feed consumption were monitored daily and hen body weights were recorded every 2 weeks. Eggs were collected daily and incubated weekly. Before incubation, samples of eggs from each dose group were taken for subsequent residue analysis. The remaining eggs were candled at 5 and 11 days of incubation to determine fertility and embryo viability. At 11 days of incubation, egg samples were taken for subsequent vitamin A analysis. At 18 days of incubation, viability was determined with an embryo viability detector. Eggs not containing a viable embryo were opened and assessed for the stage of development and the presence of any gross anomalies. A sample of live 18-day embryos and all 21-day hatchlings were examined for gross abnormalities, weighed, and necropsied. Weights of the brain, liver, heart, spleen, and bursa were recorded. Tissue samples were taken for histopathologic examination, determination of aryl hydrocarbon hydroxylase activity (liver), and PCH residue analyses (liver and brain). Those eggs which did not hatch by day 21 but contained a viable chick were incubated for up to an additional 4 days to determine if delayed hatching occurred. At the end of the feeding trial, the hens were killed and necropsied. Brain, heart, spleen, and liver weights were recorded.

Feed consumption was initially variable but stabilized during the last 5 weeks of the study with hens in the 35% carp group consuming the most feed. Control hens were consistently heavier throughout the study but they produced the fewest and smallest eggs.

Teratogenic effects similar to these reported in colonial waterbird populations occurred in a time- and dose-dependent fashion. Common terata observed in the embryos and chicks included edema in the regions of the head, neck, abdomen, and back. Occasionally, edema of the brain was noted as were incidences of microencephaly. Hemorrhages were most frequently observed in the neck

musculature. Beak deformities included crossed beaks and reduced upper beaks. Eyes were occasionally missing or variable in size. Limb deformities consisted primarily of talipes (clubfoot) and occasional malformed femurs. Unabsorbed and fluid-filled yolk sacs were observed in a number of the hatchlings. Grouped into six categories and tabulated across all treatment groups, head/neck edema was the most prevalent teratogenic effect (64%), followed by abdominal edema (15%), foot/leg deformities (14%), miscellaneous abnormalities (2.9%), skull/brain malformations (2.6%), and yolk sac anomalies (1.6%). Of the 2,081 embryos and chicks examined for abnormalities, 29% displayed teratogenic effects. The control, 3.5% carp, and 35% carp groups had a 17, 24, and 40% incidence of deformities, respectively. Multiple deformities were more common in the 35% carp group, particularly during the latter part of the study.

Liver weights were significantly greater in the 35% carp 18-day embryos. In the hatchlings, body, liver, brain, and heart weights were also significantly greater in the 35% carp group. Histopathologic examination of the 18-day embryos revealed no consistent lesions. Chemical residue, vitamin A, and enzyme induction analyses have not been completed.

At necropsy, the hens in the control and 3.5% carp groups had clinical signs (fatty and hemorrhagic livers) and histopathologic lesions indicative of fatty liver hemorrhagic syndrome which may have contributed to the unexpected patterns in feed consumption, egg production, egg weights, and terata rates.

The results of this study indicate that the contaminants contained in Saginaw Bay carp are capable of producing teratogenic effects in white leghorn hens and, as such, this species may be an appropriate avian model for studying the effects of environmental contaminants on colonial waterbirds. The results of this study also support the suggestion that declining populations of piscivorous avian species occupying the Great Lakes basin are, in part, due to problems associated with reproduction resulting from contamination of their food supply.

WESELOH, D.V, and BISHOP, C.A., Canadian Wildlife Service - Ontario Region Box 5050, Burlington, Ontario L7R 4A6, and NORSTROM, R.J., and FOX, G.A., Canadian Wildlife Service National Wildlife Research Centre, Ottawa, Ontario K1A OH3

3.7 Monitoring Levels and Effects of Contaminants in Herring Gull Eggs on the Great Lakes, 1974-1990

In 1974, the Canadian Wildlife Service initiated a surveillance project which has been in effect to this day to monitor chemical contaminants and their potential effects in what was felt to be the best single indicator species, the herring gull (*Larus argentatus).*

In this study, the levels of 16 toxic chemicals were analyzed in herring gull eggs collected annually from two colonies in each of the four Great Lakes that border Canada and one colony in Lake Michigan. Reproductive success and eggshell thickness of herring gulls was also measured during the 1970s and early 1980s. Over the years, colonies and analytical protocol and refinements have been made in the biological effects monitoring. In 1978, the study became a joint effort of the Canadian Wildlife Service and the U.S. Fish & Wildlife Service. Today eggs are routinely collected from 14 sites including the St. Lawrence, Niagara and Detroit rivers, concentrations of dioxins, furans, and non-coplanar PCB congeners are determined, and we are attempting to incorporate the use of biochemical measures of stress into the program.

From 1974-1985 all eggs were analyzed individually. Beginning in 1986, the eggs from one lake only (determined on a rotational basis) were so analyzed; the remaining eggs were tested as colony pools. Temporal and spatial trends were determined from the log-normalized wet weight concentration of the contaminant.

RESULTS

The concentrations of "major" contaminants, the predominant compounds determined at the beginning of the study, have decreased dramatically since 1974. The maximum decreases at any site for these compounds in 1990 were as follows: PCBs - 180 to 18 ppm, DDE - 33 to 4.0 ppm, mirex - 7.4 to 0.68, dieldrin - 0.90 to 0.32, and HCB - 0.60 to 0.03. Dioxins (TCDD and others) and furans were discovered in Great Lakes herring gull eggs in 1980. Since then we have analyzed archived eggs from the CWS Tissue Bank and determined that from the early 1970s to 1990 on the most contaminated lake, Lake Ontario, the levels of 2,3,7,8-TCDD have decreased from over 1,500 ppt to less than 100.

Among the colonies and compounds originally monitored and for the years 1974-1985, 67% of the comparisons have shown a significant temporal trend. Sixty-five percent have decreased, 2% have increased (oxychlordane and a-HCH at one site each), and 33% have shown no significant change in concentration. PCBs, DDE, mirex, and HCB decreased significantly on all colonies by 35-94%.

Trophic half-lives for contaminants, i.e., a measure of the compound's availability in the food web, in herring gull eggs tended to be shortest on Lake Michigan (minimum of 2.6 years for pentachlorobenzene) and longest on Lake Erie (15.2 years for PCBs). Shorter trophic half-lives tended to be associated with sites that had the highest contaminant levels initially and showed dramatic decreases over the years. Longer half-lives were associated with those sites that have shown relatively lower decreases in residue levels usually because the initial levels were lower.

Trophic half-lives of the six major contaminants compared between the 1970s and 1980s ($N = 48$) showed that 83% were significant during the 1970s and only 44% were significant during the 1980s. Overall, half-lives were shorter ($X = 4.4$ years) during the 1970s and longer ($X = 6.5$ years) during the 1980s. This indicates that these contaminants were decreasing more slowly during the '80s than they were in the '70s.

Of the original colonies monitored, the two from Lake Huron and the site in western Lake Erie had the lowest overall mean levels for both major and minor contaminants. The Lake Ontario colonies had the highest levels for mirex, HCB, PCBs, tetra- and pentachlorobenzene, and b-HCH. The Lake Michigan colony consistently had the highest residue levels for DDE, dieldrin, HE, and oxychlordane.

In the early years of the study, reproductive success of herring gulls was poor (less than 0.5 young per pair) on the colonies in Lake Ontario and eastern Lake Erie. Eggs were not hatching and young gulls were not surviving to fledge. By 1978, this pattern had changed; since then there have been no signs of contaminant-induced reproductive failure in gulls among the sites we have monitored, i.e., the gulls appear to be reproducing normally.

The traditional field methods of measuring biological (ecological) effects of contaminants appear outdated as they apply to the herring gull and we no longer use them on a routine basis. Contaminant levels are too low, and herring gulls are too resistant, to show the effects that were obvious in the early 1970s. Two alternative approaches to bioeffects monitoring are being considered: 1) we are attempting to apply more sensitive biological (physiological/biochemical) tests to herring gulls on a routine, lake-wide basis and, 2) we are holding extensive expert panels to consider changing to one or a suite of different biological indicator species (bald eagle, osprey, mink, and/or otter) whose reproductive biology and ecology are more sensitive to the lower (current) levels of contaminants in the Great Lakes.

Since its inception in 1974, the Great Lakes Herring Gull Monitoring Program has acquired one of the most complete and continuous databases in the world for contaminant residues in an indicator species found in an area seriously contaminated with toxic organic compounds. An extensive and cumulative database is necessary in order to statistically determine long-term trends in residue

levels, and to evaluate the changing environmental health of an area such as the Great Lakes. The contaminant data for herring gull eggs have allowed us to observe the overall decline in residue levels in all the Great Lakes without perturbation by transient fluctuations. As well, residue levels found in the eggs of herring gulls throughout the Great Lakes have been used as a standard against which to compare and corroborate contamination levels and trends in other biota on the Great Lakes.

BOWERMAN, W.B., Department of Fisheries and Wildlife, Pesticide Research Center, Institute for Environmental Toxicology, Michigan State University, East Lansing, Michigan 48824, BEST, D., KUBIAK, T., USDI-Fish and Wildlife Service, 1405 S. Harrison Road, Room 301, East Lansing, Michigan 48824 POSTUPALSKY, S., 1817 Simpson Street, Apartment 201, Madison, Wisconsin 53713, TILLIT, D., USDI-Fish & Wildlife Service, National Fisheries Contaminant Research Center, Route 2, 4200 New Haven Road, Colombia, Missouri 65201

3.8 Bald Eagle Reproductive Impairment Around the Great Lakes: Association with Organochlorine Contamination

Concentrations of organochlorine compounds in bald eagles (*Haliaeetus leucocephalus*) have traditionally been measured by collecting and analyzing abandoned eggs. Problems with this method include the inability to collect samples from areas of interest and the small number of eggs collected due to scavenging prior to collection. It has been argued that blood plasma collection could be used to supplement this method. We report here on the results of both methods and relate organochlorine concentrations to reproductive impairment of bald eagles that nest within 8.0 km of the Great Lakes.

Up to 10 cc of blood was collected from 46 nestling bald eagles from Michigan in 1987 and 1988 and 118 nestlings from Michigan, Minnesota, Ohio, Ontario, and Wisconsin in 1989. Eggs were collected from 36 breeding locations from 1986-90 in Michigan, Ohio, and Alaska.

Concentrations of PCBs and p,p'-DDE were observed in blood plasma of all nestling eagles sampled. Mean concentrations of PCBs, 183.3 ppb, and p,p'-DDE, 60.9 ppb, in plasma of nestlings from the Great Lakes breeding areas were significantly greater ($P = 0.001$, PCBs; $P = 0.001$, p,p'-DDE, Kruskal-Wallis) than mean concentrations in plasma of nestlings from more interior areas, 23.7 ppb PCBs, 20.0 ppb p,p'-DDE.

Concentrations of PCBs, p,p'-DDE, and dieldrin were observed in all addled eggs sampled. Mean concentrations of PCBs, 33.1 ppm, p,p'-DDE, 9.3 ppm, and dieldrin, 0.7 ppm, in eggs from Great Lakes breeding areas were greater than mean concentrations in eggs from interior areas of the Great Lakes basin and Alaska, respectively, 8.3 ppm, 1.4 ppm, PCBs; 2.9 ppm, 0.5 ppm, p,p'-DDE; and 0.2 ppm, 0.01 ppm, dieldrin.

The relationship between egg concentrations and productivity of eight eagle sub-populations was explored from Lakes Erie, Huron, Michigan, and Superior, interior Ohio, Upper Peninsula of Michigan, Lower Peninsula of Michigan, and Alaska. Significant relationships were found between sub-population productivity and egg concentrations for total PCBs, $R^2 = 0.802$, p,p'-DDE, $R^2 = 0.626$, and dieldrin, $R^2 = 0.821$.

The concentration of total PCBs and TCDD equivalents (converted to congener specific data and derived from the H_4IIE bioassay) in two addled bald eagle eggs collected near Lakes Michigan and Huron were respectively, 83 and 98 µg/g total PCBs, 21,369 and 30,894 pg/g as TCDD equivalents from congener specific data, and 1,065 pg/g and 560 pg/g as TCDD equivalents derived from the H_4IIE bioassay. In an examination of chicken feeding studies, conversion of Aroclor/congener

concentrations in bald eagle eggs are far greater than known effect levels in poultry experiments, either by total PCB concentration or by conversion of individual PCB congeners.

From 1986-1990, reproductive levels of bald eagles nesting within 8.0 km of a Great Lake were lower (0.71 young/occupied breeding area) than interior areas (1.11 young/occupied breeding area) in Michigan. Bald eagles breeding within 8.0 km of Lakes Michigan and Huron fail to reproduce above 0.6 young/occupied breeding area and are reproducing at a rate lower than is necessary to maintain the population at a stable level (0.70 young/occupied nest). Bald eagle productivity is negatively correlated with concentrations of PCBs, DDE, and dieldrin, with PCBs believed to be the primary cause of lowered reproduction and fitness of eagles who nest near the Great Lakes.

SWAIN, W.R., EcoLogic International, 2395 Huron Parkway, Ann Arbor, Michigan 48104

4.1 A Review of Research on the Effects of Human Exposure To Orally Ingested Polychlorinated Biphenyls: Reference Dose and Exposure Assessment

Three sets of studies of the impacts of human exposure to PCB-contaminated fish from the Great Lakes basin- the Michigan Sports Fisherman Cohort, the Michigan Maternal/Infant Cohort, and the Wisconsin Maternal/Infant Cohort- have been evaluated utilizing accepted epidemilogical criteria. The studies were compared against each other, and against comparable data from other geographic locales. Seven major categories of exposure sequelae were evaluated. These ranged from the effects of primary exposure to contaminants upon maternal health status, to effects from secondary intrauterine fetal exposure, including alterations in birth size and gestational age, changes in neonatal health status, and effects persisting into early infancy. Results of the evaluations suggest that the causal hypothesis may be strongly affirmed for the relationship between PCB exposure and the transplacental passage of PCB molecules.

The relationship between PCB exposure and alterations in both neonatal health status and in health status in early infancy may be affirmed with reasonable certainty. While the evidence from the Michigan Maternal/Infant Cohort related to maternal exposure to PCB and infant size at birth and gestational age is suggestive of causality, studies from other geographic locales tend not to support this conclusion. Analytic differences are likely responsible for these differences, but epidemiologically, the composite rating must be regarded as indeterminate. The relationship with observed alterations in maternal health status, composite activity ranking, and McCarthy Memory Scale deficits were also classified as indeterminate. The relationship with observed alternations in maternal health status, composite activity, and McCarthy Memory Scale deficits were also classified as indeterminate. No evidences of obvious negation were seen, although one portion of a study was disqualified because of incoherence.

Based upon maternal PCB burdens associated with deficits observed in infant visual recognition memory among children born to exposed mothers, a No Adverse Effect Level (NOAEL) was estimated. This NOAEL was used to derive a reference dose (RfD) for the visual recognition memory portion of neurobehavioral function. This RfD was 2.7×10^{-3} µg/kg/day. A comparative market basket survey of foodstuffs consumed in the U.S. conducted by the United States Food and Drug Administration estimated the daily dietary intake of PCBs in 1982 was 3×10^{-3} µg/kg/day. This value is analytically indistinguishable from the reference dose. Of further interest is the fact that both the RfD and the market basket daily intake value for PCBs are approximately an order of magnitude above USEPA quantitative risk estimate for the *de minimis* 1×10^{-6} carcinogenicity dose of 1.3×10^{-7} mg/kg/day (1.3×10^{-4} µg/kg/day). In comparison with other sensitive species,

the human RfD is an order of magnitude below the similar endpoint for rodents. The current USEPA RfD for PCB based upon low birthweight in rhesus monkeys is 1.0×10^{-4} mg/kg/day (1.0×10^{-1} µg/kg/day). This value is two orders of magnitude above the RfD for visual recognition memory.

In the Michigan Maternal/Infant Cohort, maternal exposure of infants *in utero* was primarily a function of PCBs acquired from fish consumption. Since the body burden of PCB in fish may be related to ambient water concentration, it was reasoned that an "idealized" water quality objective for PCB could be developed to reduce the fish contamination, and hence, eliminate the occurrence of visual recognition memory deficits as a function of consumption of contaminated fish.

To this end, a series of calculations were made utilizing the Human Threshold Criterion (HTC) employed by the State of Wisconsin, and retrospective analyses (hindcasting) of the U.S. Fish and Wildlife Service's historical data set for PCBs in Lake Michigan fish. The ranges of values from these calculations were in generally good agreement between and among the various methodologies employed. Depending upon the distribution coefficient selected, the extremes of the values ranged between approximately 0.1 and 10 picograms PCB per liter. The central range of values spanned an order of magnitude, 0.6 and 7.0 pg/l. From this range a compromise value of 1.0pg/l was established as the target of "idealized" water quality objective for PCB required to protect maternal and infant health. This objective is presently two to three orders of magnitude below existing PCB concentrations in the waters of the Great Lakes, and is also one to two orders of magnitude below the 79 pg/l value established by USEPA for it *de minimis* 1×10^{-6} carcinogenicitgy criterion.

Ideally, water quality objectives for PCBs should be based upon definable measures of toxicity. The current best approximation of PCB toxicity is the use of enzyme induction and TCDD-equivalent toxicity for the various individual congeners of PCB and for environmental mixtures of these compounds. Although our knowledge base is rapidly expanding, regrettably, there are insufficient data available at present to use this approach for development of water quality objectives for planar chlorinated hydrocarbons, including PCB congeners. Thus, it can be reasonably expected that an "idealized" water quality objective developed for "total PCBs" will undoubtedly change as additional information on planar chlorinated hydrocarbons becomes available. The direction and magnitude of this change will be a function of the complex interrelationships between and among congeners, including additivity, synergism, antagonism, and competition for binding sites.

HUMPHREY H.E.B., Michigan Department of Public Health, Lansing Michigan 48090

4.2 The Human Population - A Receptor for Aquatic Contaminants

Humans who eat sport caught fish are exposed to variety of chemical contaminants found in the aquatic environment. Regular consumption can result in elevated human body burdens of these contaminants and accordingly, possible toxicological risks. Beginning with a Lake St. Clair mercury exposure study in 1970, public health officials in Michigan have developed and sustained epidemiological cohort studies in order to monitor human exposure and evaluate potential health outcomes from such exposures. These studies have served as the basis for innovative collaborative research aimed at exploring subtle as well as obvious toxic effects. Work with experts in psychology, immunology, and enzymology is breaking new ground in our understanding of contaminant manifested biological differences between exposed populations and unexposed populations.

JACOBSON, J.L., JACOBSON, S.W., Department of Psychology, Wayne State University, Detroit, Michigan 48202

4.3 Follow-up on Children from the Michigan Fish-eaters Cohort Study: Performance at age 4

Two hundred and thirty-six children, recruited at birth on the basis of maternal consumption of Lake Michigan fish, participated in a longitudinal study of the neurotoxicity of polychlorinated biphenyls (PCBs) and related contaminants. Prenatal exposure was indicated by umbilical cord serum PCB level; breast-feeding exposure was indicated by maternal mild PCB level and weeks of breast-feeding. Serum samples were also obtained from the children at age 4 years. A broad range of potential confounding variables was assessed, including prenatal exposure to alcohol, maternal smoking during pregnancy, maternal age, prenatal medical complications, and quality of intellectual stimulation provided by the parents (HOME Inventory). All control variables even weakly related to each measure of exposure were controlled statistically in all analyses of the effects of that exposure on developmental outcome. None of the staff members performing the outcome assessments were aware of any of the biological or fish consumption measures of PCB exposure.

PCB level was evaluated by packed column gas chromatography at the Michigan Department of Public Health. Total PCBs were quantified using Aroclors 1016 and 1260 as reference standards. At the time of the study, congener specific analyses could not be performed by means of the analytical methodologies available. It is important to recognize that the effects found to be associated with PCB serum and milk levels may have been due to exposure to other, related contaminants in the fish these mothers ate. It may be most appropriate to consider the serum and mild PCB levels to be marker variables for an environmental exposure whose precise chemical composition has yet to be determined. Nevertheless, we refer to this exposure in terms of PCB levels, because PCBs are the substances that were measured.

About half the children had detectable PCB levels in the serum samples obtained at age 4, with levels approaching those found in their mothers' sera. Four-year serum PCB level was unrelated to cord serum level, presumably because the quantity of PCBs transferred prenatally was too small to be detected given the growth in body size since birth. The principal determinants of 4-year serum PCB levels were maternal milk PCB level and weeks of breast-feeding, which jointly explained 60% of the variance in the 4-year PCB levels.

Cord serum PCB level was associated with lower weight and smaller head circumference at birth; lower weight, shorter stature, and smaller head circumference at 5 months; and lower weight at 4 years. This evidence of growth retardation is consistent with effect observed in more highly exposed Japanese and Taiwanese (*Yusho* and *Yucheng*) children and also with a general population study from Japan. The absolute magnitude of the physical growth deficit is small, however, and probably not clinically significant.

Cord serum PCB level was associated with poorer visual recognition memory in infancy and poorer performance on the McCarthy Memory Scale and the Sternberg visual search and short-term memory paradigm at age 4. On the McCarthy, deficits were seen in verbal memory, which includes memory for words, sentences, and a story, and numerical memory, which assesses the child's ability to recall strings of numbers both as dictated and in reverse order ("forward and backward digit span"). All these effects were dose dependent. Although much larger quantities of PCBs were transferred to the child postnatally via breast-feeding, neither the physical growth nor short-term memory deficits were associated with postnatal exposure from breast milk. The only deficit associated with postnatal exposure was a slightly depressed level of activity as assessed in independent ratings by the child's mother and the examiner.

In summary, these data indicate a reasonably consistent pattern of physical growth and short-term memory deficits that appear to be related specifically to prenatal exposure, even though much larger quantities of PCBs are transferred postnatally to the child by breast-feeding. Possible mechanisms for this increased prenatal vulnerability include the high degree of sensitivity of migratory cells and cells undergoing mitosis to toxic insult and the relatively late development of the blood-brain barrier and drug-metabolizing capacities. It should be emphasized that the magnitude of the deficits observed is modest. There is no evidence of mental retardation or gross impairment. Nevertheless, the effects are sufficiently robust to disrupt short-term memory function in different domains and different modalities: verbal and quantitative auditory memory on the McCarthy Scales at age 4 and visual memory for pictures on the Fagan test in infancy and the Sternberg paradigm at age 4. These deficits, although subtle, could have a significant impact on acquisition of reading and arithmetic skills in later childhood.

DALY, H.B., Department of Psychology, State University of New York, Oswego, New York 13126

4.4 Laboratory Rat Experiments Show Consumption of Lake Ontario Salmon Causes Behavioral Changes: Support for Wildlife and Human Research Results

Fish living in the Great Lakes are contaminated with a large number of persistent toxic chemicals. People are eating these fish despite recommendations to restrict their consumption. Are these chemicals causing physiological and behavioral changes? The easiest way to answer this question is to select randomly a large number of people, and divide them randomly into two groups. One group would be forced to eat a certain amount of Great Lakes fish, and the other group would be forced to eat an equivalent amount of fish raised in a clean body of water. We would then measure if there are physiological and/or behavioral differences between the groups. Assigning subjects to groups randomly assured that the two groups were equal to begin with. By manipulating only one factor between the groups (contaminated vs. uncontaminated fish), we can conclude that this one factor caused any observed behavioral or physiological differences.

The study described above cannot be done. For the same reason we cannot prove that smoking *causes* lung cancer in humans, we cannot prove that eating Great Lakes fish *causes* physiological or behavioral changes in humans. For ethical reasons we cannot force a random sample of people to eat Great Lakes fish and force others not to. Therefore, the task of determining if the consumption of Great Lakes fish causes physiological or behavioral changes in humans who consume these fish is made extremely difficult. Probably the best ways to proceed is to use converging evidence from a variety of approaches. These include: (1) observation of changes in the wildlife living around the Great Lakes and consuming Great Lakes fish; (2) large-scale epidemiologial studies that measure fish-consumption levels and archival data such as birth-weight records, infant morality, etc.; (3) correlational research that measures the association (relationship) between mother's fish-consumption and, for example, scores on a behavioral test given to the children (e.g., tests of infant intelligence); and (4) research with sub-human animals using the experimental method outlined earlier (i.e., random assignment of subjects to groups).

The first three approaches are examples of correlational research. The reason one cannot conclude that there is a cause and effect relationship using these approaches is that there may be one or more confounding variables, i.e., it is possible that what caused people to consume Great Lakes fish may also be the reason for the observed behavioral or physiological differences. For example, poorer people may eat the Great Lakes fish for economic reason, and poorer people may also not be able to provide as rich an environment for their children (e.g., educational toys). It may be the lack of an enriched environment that caused the behavioral differences, not the toxic chemicals in the fish eaten. One can measure a large number of these potentially confounding variables and statisti-

cally remove their influence, but one can never be sure that all confounding variables have been accounted for. However, when one finds a significant correlation between amount of Great Lakes fish eaten and scores on behavioral or physiological tests, the correlation may be due to a cause and effect relationship. This is especially true if the criteria outlined in the proceedings of the first Workshop on Cause-Effect Linkages are met. In addition, evidence of a strong correlation can be used to *predict* differences in future fish-eaters. Other speakers will review evidence from these research approaches. Their results clearly indicate a cause for alarm. The Jacobsons have shown a number of behavioral changes in the children of mothers who consumed Lake Michigan fish. Behavioral changes (e.g., parental attentiveness) have also been reported by Kubiak and others in Forster's terns eating Lake Michigan fish (Green Bay).

The experimental method is used to determine cause and effect relationships, and can be used with subhuman animals who are fed Great Lakes fish. We have used this method with laboratory rats. All results point to one conclusion: RATS FED LAKE ONTARIO SALMON ARE HYPER-REACTIVE TO UNPLEASANT EVENTS, but react normally when "life is pleasant."

Our procedures involved feeding adult rats a diet consisting of 30% ground Lake Ontario salmon fillets and 70% ground rat chow for 20 days. Control rats are fed a 30% mixture of Pacific Ocean salmon (shown to have far lower levels of persistent toxic chemicals), or a no-salmon rat chow diet. They are then removed from their fish diets and given behavioral tests. To measure their reaction to a mild electric shock, we first trained them to run down an alley (210 X 10 X 14 cm) to receive a large food reward. Then we gave them an extremely mild shock (0.1 mA, 1 sec) just before they reached the reward. The time to reach the end of the alley to obtain the reward increased from 4 to 27 seconds after the shocks were introduced in the group previously fed Lake Ontario salmon, but from 4 to only 12 seconds in the group fed Pacific Ocean salmon. In a similar study, rats were shifted form a large (555 mg) to a small (37 mg) food reward, and the group fed Lake Ontario salmon showed a much larger contrast effect: they took much longer to run down the alley to obtain the small reward than the grouped fed Pacific Ocean salmon or no salmon (similar results were obtained when the rats were fed a 10% diet of Lake Ontario salmon for 60 days). In the first study the rats fed Lake Ontario salmon were hyper-reactive to a fearful event (shock), and in the second study they were hyper-reactive to a frustrative event (smaller than expected food reward).

In other studies rats fed Lake Ontario salmon showed: (1) greater avoidance of unpredictable frustrative nonreward; (2) greater suppression of lever-press responding when mild shocks were introduced; (3) lower activity levels in an unfamiliar environment; (4) less resistance to extinction (gave up sooner when the food reward was no longer given; (5) faster re-aquisition when rewards were re-introduced; and (6) more lever-press responding on a progressive ratio schedule (the rat is required to press a lever two times, then four times, then six times, etc. to receive a food reward, an increase in two responses after each reward). All results, even these last two counter-intuitive results, were predicted by DMOD (a mathematical model), given the assumption that rats fed Lake Ontario salmon have an increased reaction to negative but not positive events. The pattern of results rules out a large number of alternative interpretations (e.g., changed activity levels, physical illness, task aversion, sensory impairments, memory changes, deprivation level).

We have recently shown the same BEHAVIORAL CHANGES IN THE OFFSPRING of rats fed Lake Ontario salmon (progressive ratio schedule and contrast effect). The female rats were fed the fish diets from the day they were placed with the male rats until the rat pups were 7 days old. The pups were allowed to nurse until weaning (21 days of age). The behavioral changes occurred when the offspring were tested as juveniles and as adults, even though these rats have never been fed the fish.

We have never found differences between the two control groups fed Pacific Ocean salmon or no salmon, indicating that it is not a fish diet, per se, that causes the behavioral changes in the rats fed Lake Ontario salmon. The differences are always between the group fed salmon raised in Lake

Ontario and the control groups. The obvious difference among these diets is the higher level of persistent toxic chemicals present in Lake Ontario salmon. At this point we do not know which chemical(s) is (are) causing the behavioral changes, but it is possible that the behavioral changes are due to additive or synergistic effects among a number of chemical.

Can the results with laboratory rats be generalized to humans? To answer this question, we have begun a four-pronged approach at the Center for Neurobehavioral Effects of Environmental Toxins at the State University of New York at Oswego:

	Adults	Offspring
(correlational) HUMANS		
(experimental) RATS		

We are testing adults and offspring of humans and rats who have eaten Lake Ontario fish. Their patterns of behavior are being compared with controls who have not eaten these fish. If the same behavioral changes found in the laboratory rats (experimental design) are found with humans (correlational approach), then we will have a firmer basis for concluding that the correlational results found in humans are based on cause and effect relationships.

DAVEY, P.G., 425 East Washington St., Ann Arbor, Michigan 48104

4.5 Human Multiple Chemical Sensitivity Syndrome: A Clinician's Approach

Chemical sensitivity offers a challenging puzzle for physicians, scientists and public-policy decision makers. The pieces of the puzzle include observations of possible inciting or triggering substances and health effects, and discovering reasonable physiologic mechanisms, diagnostic methods, and treatments. People who claim to be "chemically sensitive" experience acute symptoms to low levels of chemicals commonly found in our homes, work places, schools, and in other surroundings. Although a conclusive and exact picture will be fulfilled in time, at present, there is adequate evidence to conclude that chemical sensitivity does exist as a serious health and environmental problem and that action is warranted at both state and federal levels.

PROTHRO, M. U.S. Environmental Protection Agency, Washington D.C. 20460

5.1 EPA's Water Quality Criteria Program: How Criteria are Developed.

Water quality criteria are scientific documents that compile toxicity studies and other information on specific chemicals. They also contain formulas that take these data and develop bottom line water quality numbers, which may be used by states in developing water quality standards, which then serve as a basis for state water quality regulatory programs. Historically EPA's water quality program has focused on chemical specific human health and aquatic life criteria: EPA has developed 100 human health criteria and 35 aquatic life criteria. The purpose of human health criteria is to estimate the ambient water concentration of a pollutant that does not represent a significant risk to the public. Ambient water quality criteria for human health are primarily based on two types of biological endpoints: carcinogenicity and toxicity. In some cases, criteria have been based on organoleptic effects. For the purpose of deriving ambient water quality criteria, carcinogenicity is regarded as a non-threshold phenomenon. The information needed for carcinogens is an estimate of the carcinogenic potency of the compound. For non-carcinogens,

EPA uses the reference dose (RfD) as the parameter in calculating the criteria. The next step is exposure assessment to determine safe exposure levels. In this process, EPA characterizes the human populations exposed to the chemicals, the environmental transport and fate pathways of those chemicals, and the frequency, magnitude, and duration of the exposure dose. To calculate the water quality criteria for carcinogens, EPA next determines the water concentrations estimated to cause a lifetime, upper bound carcinogenic risk of 10^{-5}, 10^{-6}, and 10^{-7}. The data used for quantitative estimates are lifetime animal studies and epidemiological studies. With regard to aquatic life criteria, generally separate criteria are calculated for fresh water and salt water. Criteria are now calculated for acute and chronic effects. In addition to these chemical-specific human health and aquatic life criteria, more recently EPA has begun developing other types of criteria, such as sediment criteria, biological criteria, and wildlife criteria.

LUDWIG, J.P., Ecological Research Services, Inc., 2395 Huron Parkway, Ann Arbor, Michigan, 48104, and COLBORN, T., W. Alton Jones Foundation and World Wildlife Fund, 1250 24th St. N.W., Washington, D.C., 20003

5.2 The Human Cancer Paradigm Reviewed: An Inadequate Means to Protect Piscivorous Great Lakes Wildlife and Humans

Cancer, mutation, and acute toxicity studies have driven standard regulatory decisions on toxic chemicals and their permissible discharge levels for several decades. The regulatory community has been forced to project environmental impacts based on the assumption that only these possible outcomes are of concern: acute or lethal toxicity to the exposed (test) organism, or the potential for the induction of cancers based on extrapolations from laboratory induced malignancies. Thus, the assumptions underlying standard decision-making process are generally based on the outcomes of only two types of studies, neither of which are related to the real world biological effects of long-term, chronic exposure of living organisms to the array of interactive chemicals which contaminate all ecosystems. For many of the chemicals that contaminate aquatic ecosystems, the more important and relevant endpoints are due to long-term, chronic exposure that eventually affect offspring of exposed individuals, e.g. reproductive failure or depression, altered development, immune system suppression, elevated rates of disease, wasting syndromes, and both adult and neonatal behavioral changes. In short, there are broad physiological and developmental effects which are not being addressed under the present scheme of risk assessment. The papers at this and the previous Cause-Effect Linkage Symposium provide sufficient evidence that a regulatory structure primarily dependent on the cancer paradigm does not address wildlife health. It is also doubtful that this traditional approach can protect human health. Well controlled laboratory studies cannot adequately duplicate or predict the potential impact of phenomena such as variable biomagnification factors, selective bioconcentration congener sorting in the environment, and related phenomena which render even traditional toxicity tests invalid after contaminants pass through the food web. A false sense of security is engendered with respect to human health impacts when only a cancer outcome is considered.

Great inertia exists in the regulatory community because it is adept at and comfortable with conventional studies and protocols. It takes a great deal of effort to adopt new ways of thinking. In order to use nonconventional, real-world bioeffects endpoints in a rigorous fashion to establish low or no adverse effect exposure levels for toxic contaminants. New testing that tracks several generations' reproductive outcomes of impacted human and wildlife populations is needed. Just as effluent toxicity testing spawned a large analytical infrastructure and unique sets of test protocols, monitoring the bioeffects of persistent, bioaccumulated compounds in real populations requires the development of unique skills and a supporting community. Unfortunately, neither

universities, nor most resource agencies, nor industries possess the trained personnel to do such studies, particularly for wildlife populations. Regardless, an aroused and much more sophisticated public now demands a real world, holistic approach to ecosystem regulation. Ultimately, the false comfort inherent in the cancer paradigm and acute toxicity studies must be replaced by the realization that actual risks are posed by complex exposure to many toxic chemicals interacting in a dynamic ecosystem. While these risks cannot currently be predicted in a mathematical way, it is apparent that unborn generations are at risk. This risk varies greatly in degree and expression depending on when in the life cycle exposure occurs. The societal implications of ignoring bioeffect outcomes are enormous. A vastly expanded regulatory system to meet the challenge will take time and significant investment.

Underlying any decision to broaden regulatory strategy beyond cancer, in order to embrace the ignored nontraditional endpoints, deep senses of commitment, responsibility and stewardship are required. Not only do these judgements depend on science, but more on nonscientific valuing exercises in the realm of ethics and philosophy. For example, do recreational experiences and a Great Lakes sport fishery outweigh the risks of changed behavior, lost human reproductive and mental potential, terata, and decreased immunocompetence in offspring? These and other effects are associated with children of fish-eating women, and other groups of human and animal fish consumers. Do humans have the right to inflict the impacts of toxic chemicals on wildlife? These questions are not resolvable by science. Personal and professional ethics, social contracts, and religious beliefs are the only means available to make comparisons between such grossly dissimilar concepts as cancer risks and intelligence. Risks cannot be quantified until society agrees on which endpoints are relevant and in what order of importance. Without consensus on endpoints, good faith attempts to address such issues as fish advisories will continue to be viewed as unreasonable by many interested groups.

It is possible that the costs of increased health care requirements in children born to exposed mothers will be determined. However, quantification of such putative endpoints as lost potential (or test performance), such as occur as a result of embryonic and neonatal exposure to agents like PCBs, alcohol and lead, will be exceptionally difficult and contentious exercises. As a society we must be willing to ask such agonizing questions as, "What does a downward shift in intelligence of babies born to exposed mothers and the related health care costs mean to society? Do the recent general societal patterns of increased female and male human infertility, merging of human sexual identities, increasingly poor performance in schools by recent generations, increases in autoimmune diseases, etc. have anything at all to do with exposures to persistent lipophilic toxic chemicals? If so, where in society should the costs of these damages be borne?" Even posing such questions offends many people. These kinds of questions are similar to those that plagued the smoking debate for decades. As this workshop has demonstrated, science will continue to provide more and more evidence on causal relationships, but it will take society-at-large through the political process will resolve the debate on limits to new or expanded regulatory strategies.

All parties must now begin to translate cause-effect linkages and similar scientific results into good faith risk assessments in spite of the personal and professional risks involved. All of us should resolve to have patience and to explore novel ideas at the risk of appearing foolish or vulnerable. We must learn to embrace our errors and gain from experience. We must listen to those who point out gaps in our knowledge and the unwarranted assumptions of existing regulations. Although the task will be difficult, contentious, and time-consuming, it is imperative to demand assessments that include multigenerational developmental effects, as well as cancer, if our society is to address these problems effectively.

How the Public Should View and React to the Data Produced by Wildlife and Human Bioeffects Scientists

Sorting the wheat from the chaff to reach responsible use of complex data.

by Loraine Campbell, Michigan Audubon Society

We have heard many words during the course of this symposium: biomarker, teratogen, dioxin equivalents, bioassay, and enough acronyms to make a bowl of alphabet soup. But of all the words, terms, phrases, and concepts you have learned , three should be outstanding: CAUSE, EFFECT, and most important LINKAGE. "Linkage" is the key. The concept of linkages is essential if we are to understand the entire network of biological and social relationships tied to the issue of persistent toxic contamination. And ultimately working with linkages is only the beginning of real actions we must follow if we are to truly resolve this most complex and damaging problem and restore the environmental integrity of the Great Lakes.

How does the public sort true from false, risk from damage, threat from injury, paradigms from reality, the wheat from the chaff? We must develop a fundamental understanding of linkages.

The most basic linkage of all is that we are linked to our environment. Over recorded history, the human species has fought that reality. Man has held up his bible in defiance of that link, claiming he was created to subdue the earth. As he evolved, man used his technology to distance and insulate himself from the linkage. He's even tried to buy his way out of it following the philosophies, "I can soil my bed as long as I am willing to pay the price of living in filth," or more often, "I can move away from my filth." But he has never broken the link, nor can he escape it. In the past, individuals have reminded us of our linkage with very poignant words. The legendary Saint Francis called the sun and water his brothers. Centuries later, a native American named Seattle stated simply yet eloquently, "Man is a part of the earth...What man does to the earth he does to himself." The environmental movement of the 1960s and its resurgence today shows that some segments of the world community are rediscovering and accepting that linkage. The first Cause-Effects workshop and the 1989 IJC Biennial demonstrated that a ground swell of public interest in the wildlife of the Great Lakes Ecosystem exists. Cause-Effects Linkages II and the impending IJC Biennial meeting here in Traverse City will intensify that interest and commitment to restoring the resource, and will dictate that we include our own species as part of that resource.

We too are part of the resource. We are part of the cycle of nutrients and the food web. And that linkage extends to our children, and those children they will bear. Every woman in this room carries her life's complement of eggs as I carry mine. Our eggs were fully formed at birth. Our bodies will not manufacture more. Any chemical burden I inherited from my mother, plus the chemicals I accumulated growing up in Detroit, I have passed on to my two daughters. They are five and eight years old. They carry their eggs, my chemical burden and the burden they will acquire growing up. They will pass it on to their children. This linkage between generations renders my grandchildren vulnerable to the environmental desecration of the 1950s, 60s, and 70s and the bioeffects described in the presentations of Helen Daly, and Sandra and Joseph Jacobson. Human bioeffects may not be as obvious as the crossed bills or clubbed feet of cormorants and terns described by Chip Weseloh and Jim Ludwig. Rather they involve subtle changes in behavior, in the ability to learn, grow healthy, and reproduce successfully. It is another linkage ignored, but only at great risk to our own children.

We have been stumbling over our environmental blunders for years. Traditionally, we follow the same steps to identify, study, and solve each issue. It is the scientific process to isolate and define the hazard, often *ad nauseum*. It is an economic process to determine the financial impact of the

problem as it exists, if it is left untreated, and the costs incurred by society to resolve the problem It is a political process to take the information of the scientists and the economists, balance those data with our own personal values, and legislate change. The total process is really another series of links in a chain of political events we can label 'the system.' Yet for the hazard of PCBs and similar compounds, the old system is obsolete. The chain has become incredibly tangled, partly because we do not agree on the meaning of the science as the recent reevaluations of dioxin toxicity illustrate.

In his opening comments, Mike Gilbertson remarked that the issue of persistent toxic chemicals has cast the science into a revolution. There is growing concensus that the traditional process for dealing with water pollution does not work with PCBs, dioxins, and furans. Dilution is not a solution. PCBs *are* dilute, trace molecules measured in water as parts per trillion or quadrillion. How much or little is that? If we walk down to the bay and scoop up a handful of water, can I say with any certainty how many trillion trillion water molecules I hold and how many of those molecules are PCBs? And what threat does that handful of molecules really represent? Traditional regulatory guidelines state that toxic substances can be released in nontoxic amounts. But parts per trillion of PCB congeners physically and chemically sort in the air, water, and sediments. The more toxic congeners are drawn to fat molecules in living systems like iron shavings to a magnet. They bioaccumulate. And parts per trillion in water are concentrated forty to fifty million times in the tissues of fish and birds. Bioaccumulated PCBs are linked to birth defects, failed reproduction, wasting syndrome, and immunosuppression in wildlife. Additional research presented at this symposium suggests a linkage between toxics and impaired health and development in our own children. Parts per trillion of PCBs in Great Lakes water is still far too much. Conclusion: PCBs do not exist in nontoxic amounts in the environment.

Mike concluded that the scientific revolution will give rise to a new paradigm that establishes the relationship between PCBs and dioxin-like compounds and observed bioeffects in wildlife and humans. These effects are reproductive and developmental as opposed to carcinogenic. But the revolution will also yield a new set of linkages between institutions and disciplines. To determine exactly how organochlorines such as PCBs behave in different media, and in living systems, chemists, geologists, biologists, ecologists, psychologists, and human physiologists must continue to share their expertise and their combined understanding. Only a blending of traditional scientific disciplines holds an answer.

Further, if society is to take the information of the scientists and implement new regulatory processes, then governmental institutions must establish new effective linkages. Within the Great Lakes Basin there are US and Canadian federal governments with separate divisions and cabinets. These federal agencies are broken down into a myriad of individual offices, each charged with addressing a particular area or concern: air or water quality, various kinds of pollution, hazardous wastes, wildlife, fisheries, commerce, enforcement, and human health and services. Below these federal labyrinths are eight Great Lakes state governments and one provincial government. Their departments and ministries are at least as complex as the divisions of federal government. Further, there are literally hundreds of county and municipal governments in the basin. Traditionally, linkages between these layers of federal, regional, and local governments have been few and often not cordial. Cross linkages between the various levels of the two countries are even more remote. This is a perfectly acceptable and efficient system for many issues and duties. But if the Great Lakes Basin is to implement an ecosystem approach to environmental quality, linkages must extend between governmental levels and between the countries. Further the linkages must include the general public. This process has been initiated through the RAP process, and through environmental town meetings involving local governments, business and industry, environmental organizations and the public. This symposium is another example. The first Cause-Effect Linkages Workshop was a closed meeting, offering the scientists a chance to share information and formulate the guidelines for establishing cause-effect relationships. However, Cause-Effects II

has welcomed the general public, environmentalists, businessmen, sportsmen, educators, and representatives from government agencies. It is the belief of the Michigan Audubon Society and my own personal belief that the level and scope of interaction must increase and be carried beyond plans for cleanups and actual cleanup processes. The interaction must also continue through monitoring programs and include ongoing dissemination of new information. Our understanding of the behavior and effects of toxic chemicals and the success or failure of remediation is a new and emerging interdisciplinary science. The rules will change as the information changes. No doubt even the new paradigm that is a direct result of the data accumulated by the scientists here will change over time. It has changed subtly as a result of this symposium. The science, chemistry, and ecosystems are just that complex and dynamic. But if we forge these new linkages, it is probable that we will be able to anticipate the results of our actions on the system, and modify them accordingly as long as we monitor our results. We can avoid the old pattern of labeling the problem and determining damage control after injury to the system and living things has occurred.

Ah yes, you nod, that sounds good, but think of the cost. Yes, cost is the final linkage. In our society we link cost to value. Some values can be measured and assigned a numerical or dollar measurement. Others cannot. According to biologist Eric Ashby, there are four types of values, including market value, value as defined by usefulness, intrinsic value, and symbolic value. We can assign dollar figures to market value. The fish, water, and harvestable resources of the Great Lakes ecosystem have a cash value. Similar prices can be assigned to cleanup and education programs. Dollar amounts or quantitative measurements can also be assigned to values as usefulness. For example, there is a measurable cost to reduce the number of metric tons of carbon monoxide released by cars over a specific period of time. The value gained is cleaner air. In the same way, we can determine the cost of dredging one mile of toxic sediments from a river bed. But we cannot place a dollar value on the beauty of the river or Grand Traverse Bay, the sound of loons wailing at sunset, or the tranquility of wilderness. These are intrinsic values. And if we attempt to quantify these as dollar values, part of value of the resource is automatically lost. The final value is one of symbols. Symbols, by the fact that they represent something precious, sacred, or treasured cannot be measured.

There are incredible costs to be paid to cleanse the Great Lakes ecosystem of toxic contamination. It will take many billions of dollars. It will also take time, energy, and much effort on the part of individuals, businessmen, governments, regulators, and the public. Do we have faith that the values gained over time will exceed the costs? Moreover, some of the values to be regained cannot be measured. The new paradigm that uses biological monitoring to determine the effects of contaminants on the system and targeted wildlife species ensures a real-world ecosystem approach to environmental quality. It moves us closer to that time when we change our actions to prevent environmental disasters, rather than focusing energy on cleanups. Renewed and healthy Great lakes fisheries ensure a viable resource for future generations. They preserve a way of life as well as quality of life. Progressive industries have already demonstrated that environmentally sound technologies and processes are cost effective, and in many cases more profitable than old procedures that generated toxic wastes. In addition, employees feel a renewed sense of pride working for firms that mandate environmentally responsible policies. Other savings are also realized by our governments. If they do not allow pollution through the old permit process, they need not regulate it, dispose of it, recycle it, or find the funds to clean it up. Most importantly, values are gained in our own lives. *A shift in life style now is an investment for our children. It is their health and their quality of life that we gain.*

But how can we tell if and when the cause-effect linkages approach really works? How will we know we have succeeded in restoring the Great Lakes ecosystem? The fourth value described by Ashby is one of symbolism. I suggest to you that when the bald eagle, the symbol of the United States and a universal symbol of strength, successfully breeds along the Great Lakes shores we

will have truly repaired the system. When immature bald eagles raised in Green Bay and Saginaw Bay fish along the bluffs and hang on the wind over the lakes and return six years later to nest, we will know in our souls we have succeeded.

As we leave this conference, let us resolve first to have yet another Cause-Effect Linkages meeting as new data and understandings are produced by the scientific community. We must address the coming generations of people, wildlife, and political systems. Second, let us resolve to carry our new understandings into the political process in partnership with all segments of society: the general public, special interest groups, and, *yes*, industry too. The way out of this mess is to convince both political and industrial leaders to be serious in our shared commitment to clean up the Great Lakes. Realize this is a process of creating *new cross linkages* or networks. The process involves changing the personal values of the people in control of the political and industrial institutions of our society. And finally, to echo Mike Gilberton's call at the beginning of this symposium, let us resolve to implement the revolution in our scientific understanding of Cause and Effects into the values and ethics of wise stewards of the resources of the Great Lakes.

Appendix – Authors and Panelists

Ed Addison
Wildlife Research Section
Ontario Ministry
of Natural Resources
P.O. Box 5000
Maple, Ontario L6A 1S9
416-832-7124

Dr. Richard Aulerich
Animal Science Dept.
Michigan State University
Anthony Hall
East Lansing, MI 48824
517-336-1393

Dr. Pierre Beland
St. Lawrence National Institute
of Ecotoxicology
3872 Parc-La Fontaine
Montreal, Quebec H2L 3M6
514-524-8711

Dave Best
USFWS
Manly Miles Bldg.
1405 S. Harrison
East Lansing, MI 48824
517-337-6651

William Bowerman
Dept. of Fisheries and Wildlife
Pesticide Research Center,
Institute for Environmental
Toxicology
Michigan State University
East Lansing, MI 48824
517-353-9420

Loraine "Rainy" Campbell,
Michigan Audubon Society
1051 Avon Manor
Rochester Hills, MI 48307
313-852-1969

Dr. Theo Colborn
The Conservation Foundation
1250 24th St. N.W.
Washington DC, 20037
202-778-9643

Dr. John Cooley
Fisheries and Oceans
Canada Centre
for Inland Waters
P.O. Box 5050
Burlington, Ontario L7R 4A6
416-336-4670

Helen Daly
Dept. of Psychology
State University of New York
Oswego, NY 13126
315-341-3459

Dr. Paula G. Davey, M.D.
Environmental Medicine
425 E. Washington, Suite 201
Ann Arbor, MI 48104
313-662-3384

Dr. Gary Dawson
Consumers Power Co.
1945 W. Parnall Rd.
Jackson, MI 49201
517-788-2432

Carlos Fetterolf
Great Lakes Fishery Commission
1451 Green Rd.
Ann Arbor, MI 48105
313-994-3331

Dr. Glen Fox
Canadian Wildlife Service
National Wildlife Research Centre
Hull, Quebec K1A 0H3
819-997-6076

Dr. John Giesy
Dept. of Fisheries and Wildlife,
Pesticide Research Center
Michigan State University
East Lansing, MI 48824
517-353-2000

Michael Gilbertson
Great Lakes Water Quality Board
International Joint Commission
100 Ouelette Avenue
Windsor, Ontario N9A 6T3
519-250-7821 or
313-226-2170

Dr. Andrew Gilman
Great Lakes Health
Effects Program
Environmental Health Dir.
Bureau of
Chemical Hzds, Rm 241
Health and Welfare Canada
Ottawa, Ontario, K1A 0L2
613-957-1867

James Grant
Surface Water Quality Division
Michigan DNR
Box 30028
Lansing, MI 48909
517-335-4124

Chris Grundler and
Connie Hunt
Great Lakes National
Pgm Office
USEPA
230 S. Dearborn
Chicago, IL 60604
312-886-4014

Sylvia Heaton
Dept. of Animal Science
Anthony Hall
Michigan State University
East Lansing, MI 48824

Dr. Harold Humphrey
Michigan Dept. of Public Health
P.O. Box 30195
Lansing, MI 48824
517-335-8335

Drs. Joseph and
Sandra Jacobson
Wayne State University
Dept. of Psychology
Detroit, MI 48202
313-577-2813 or
313-577-2822

Tim Kubiak
USFWS
310 Manly Miles Bldg.
1405 S. Harrison Rd.
East Lansing, MI 48824
517-337-6651

Dr. John Leatherland
University of Guelph
Dept. of Zoology
Guelph, Ontario N1G 2W1
519-824-2719

Dr. James P. Ludwig
Ecological Research Services Inc
2395 Huron Parkway
Ann Arbor, MI 48104
313-677-0050

Matthew E. Ludwig
Ecological Research Services, Inc
2395 Huron Parkway
Ann Arbor, MI 48104
313-677-0050

Dr. Mike Mac
USFWS Ann Arbor
National Fisheries Research
Center - Great Lakes
Ann Arbor, MI 48105
313-994-3331

Dr. Ross Norstrom
Wildlife Toxicology and
Surveys Branch
Canadian Wildlife Service
Ottawa, Ontario K1A 0H3
819-997-1411

Martha Prothro
Office of Water Regulation
and Standards, USEPA
401 M Street, S.W.
Washington, D.C. 20460
202-382-5400

Dr. Richard Seegal
Dept. of Public Health
New York State
Albany, NY
518-473-4738

Cheryl Summer
Dept. of Animal Sciences
Michigan State University
East Lansing, MI 48824
517-224-2821

Dr. Wayland Swain
EcoLogic International
2395 Huron Parkway
Ann Arbor, MI 48104
313-973-2780

Dr. Don Tillitt
National Fisheries Contaminant
Research Center
USFWS Missouri
Route 2, 4200 New Haven Rd.
Columbia, MS 65201
314-875-5399

Dr. D.V. Weseloh
Canadian Wildlife Service
Ontario Region Box 5050
Burlington, Ontario L7R 4A6
416-336-4968