Ecosystem Health and Sustainable Agriculture 2

Ecology and Animal Health

Editors:
Leif Norrgren and Jeffrey M. Levengood

Editors:Leif Norrgren and Jeffrey M. Levengood

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Preface

The Baltic Sea and the North American Great Lakes are influenced by many different and similar problems affecting its environmental status. The Baltic Sea was classified by the United Nations International Maritime Organization (IMO) as a Particularly Sensitive Sea Area (PSSA) in April 2004. Both the Baltic Sea and the Great Lakes are of ecological, socioeconomic, cultural or scientifically importance. Discharge of nutrients from agriculture and waste-water treatment plants, as well as from industries, transportation and other human activities leads to eutrophication and other forms of pollution. The Ecosystem Health and Sustainable Agriculture project aims at updating knowledge in the field of rural development, sustainable agriculture and animal health pertaining to the Baltic Sea Region and to some degree also the Great Lakes region.

The agricultural activities are often based on individual producer´s decisions and on their attitudes, knowledge and level of technology. It is however also based on political and economic considerations, attitudes and opinions from the society. Thus, continuously updated scientifically based knowledge, both from an environmental, social and economic view, need to be disseminated and applied with a much increased ambition. Technological facts may be well known, but still strong social and economic reasons and pressure from outside to make short term profits hinders the appropriate application of relevant measures. This is the reason why we have all parts of the sustainability concept covered in our texts: the ecological, the social, the economical, and the institutional/juridical.

The books produced as part of this project include:

1. Sustainable Agriculture
2. Ecology and Animal Health
3. Rural Development and Land Use

These three books are based on experience from the Baltic Sea and Great Lakes Regions and written by prominent experts and scientists from the two regions. Two networks have been involved in the production of the books, The Baltic University Programme (BUP) and the Envirovet Baltic Networks. The BUP is a network of approximately 220 universities in the drainage basin to the Baltic Sea that cooperates on sustainable development, studies of the region, its environment and its political changes. The program, founded in 1991 at Uppsala University, Sweden, operates by producing courses, holding conferences and seminars. In 2010, the BUP network delivered courses at more than 100 universities serving nearly 9,500 undergraduate and graduate students. The Envirovet Baltic, a network of environmental health scientist/ educators from USA and the nine countries bordering the Baltic Sea, was founded in 2001 on an initiative of the College of Veterinary Medicine, University of Illinois and the Centre for Reproductive Biology in Uppsala, Swedish University of Agricultural Sciences with scientists from universities in the Baltic Sea Region (BSR). Courses are delivered separately by each university in the networks in both cases. Preferably all the three books should be studied to give a comprehensive overview on actual experience and research findings. We have chosen to use the ecosystem health concept to understand and prevent problems for the future. It is our aim that the books will strengthen knowledge on ecosystems and its interaction with human activities in a wider sense. The texts presented in this book should deepen and update our knowledge on all aspects of rural development and sustainable agriculture. Our aim is furthermore to provide explanations of the problem complexity and examples of problem solving.

Target groups are students, teachers, experts and people working in government offices, ministries, municipalities and as agricultural advisors and managers of different natural resource based activities in rural areas.

The Baltic Sea Watershed with its population of more than 85 million people contributes to an ongoing environmental disaster. The disaster is well documented and can be summarized as:

1) Eutrophication from heavy contamination of excess Nitrogen and Phosphorus – sources are diffuse sources, point sources and atmospheric downfall
2) Excess fishing, distorting the marine ecosystem; diminishing cod population being the most drastic example
3) Contaminants other than nutrients, mainly PCB, heavy metals, oil spill, human and animal drugs, etc.
4) Distortions of the biodiversity of the sea, e.g extinction of many indigineous species, and introduction of alien species which distort the ecological balance drastically
5) Threats connected to climate change

Sustainable Agriculture and Sustainable Rural Development

"Sustainable agriculture" has become a popular way of expressing that what society wants is an environmentally sound, productive, economically viable, and socially desirable agriculture. However, the concept of agricultural sustainability does not lend itself to precise definition. Agriculture is practiced in so many climates and in different cultural contexts, so "sustainable agriculture" cannot possibly imply a special way of thinking or of using farming practices.

Sustainable agriculture is an approach to securing the necessary resources for safeguarding global food production, biodiversity reserves, recreation needs, water quality and well developed rural areas and wildlife areas. It can also be an effective means of poverty reduction and of achieving the Millennium Development Goals, as well as means of mitigating climate change. It is also about health, welfare, respect and ethics regarding animals and man, as well as quality of food and feed. In other words they are truly transdisciplinary and represent a new holistic outlook on ecosystem health and sustainable agriculture.

We have chosen to give agriculture a wider meaning than the traditional. It is common to understand agriculture as being the activity which is securing food supply. But as we shall see in the chapters on historical trends, activities on the countryside have always been complex and integrated with other activities in rural areas, such as small business, forestry, fishing, and other activities. Also several hundred years ago, the farmer combined biological and technical knowledge with economical and organisational skill, and very often he or she made money from different other jobs like carpentry, timber and coal production, horse- and oxen driving, cheese production, food conservation etc. Today the diversity in income generation is even higher and in fact, every farming enterprise has at least one or two side activities. Our definition of agriculture as main activity would today be: to produce and manage biomass. Some examples, explained in more detailed in our books, are (besides food production):

- public goods in the form of natural and cultural amenities to benefit the ecosystem (such as management national parks and of landscape for a certain type of desired biodiversity)
- fish production
- biomass for timber and fibre products
- biomass for energy production

- social caretaking of, for instance, sick people or people who needs rehabilitation from criminal or other lives
- tourism (including views, maintenance of tracks, camping sites, buildings, etc)
- recreation (such as horseback riding and horse racing, golf- and soccer fields, fishing sites, hunting etc)
- animal raising and caretaking (dog- and cat kennels etc)

A widespread definition of sustainability of agriculture, with which we sympathise, is the FAO (Food and Agricultural Organisation of the UN) statement:

> *”Overall objectives of technological interventions for sustainable agriculture can be summarized as food security and risk resilience, environmental compatibility, economic viability, and social acceptability.”*

An addition to the FAO definition would need, as we see it, the following phrasing: *Sustainable agriculture is not linked to any particular technological practice. Sustainable agriculture should have adaptability and flexibility over time to respond to demands for biomass production but it should also have ability to protect the soil, the genetic resources and the waters.*

We also want to recall the definition done for the specific Baltic Sea Region, as expressed by the Baltic Agenda 21:

Sustainable agriculture is the production of high-quality food and other agricultural products and services in the long run, with consideration taken to economy and social structure in such a way that the resource base of non-renewable and renewable resources is maintained. Important sub goals are:

- *The farmers` income should be sufficient to provide a fair standard of living in the agricultural community;*
- *The farmers should practice production methods which do not threaten human or animal health or degrade the environment, including biodiversity, and a the same time minimise the environmental problems that future generations must assume responsibility for;*
- *Non-renewable resources gradually have to be replaced by renewable resources, and that re-circulation of non-renewable resources is maximized;*
- *Sustainable agriculture will meet the needs of food and recreation, and preserve the landscape, cultural values and historical heritage of rural areas, and contribute to the creation of stable, well-developed and secure rural communities;*
- *The ethical aspects of agricultural production are secured.*

It is our firm belief that agriculture and related activities can, if well managed, be positive for the ecosystems and for biodiversity. Contrary, agricultural activities can also give disastrous effects to the environment including humans living in the area. Often national goals are not only to sustain a food security of the country, but also to keep an open landscape and rural lifestyle. A sustainable rural development has its backbone in a sustainable agriculture.

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Christine Jakobsson, Leif Norrgren, Ingrid Karlsson, and Jeffrey Levengood

Part A

Wildlife and Ecosystem Health

Authors: Val Richard Beasley and Adriane Michelle Adkesson

Coordinating Author: Val Richard Beasley

Wildlife and Ecosystem Health

1

Val Richard Beasley
Adriane Michelle Adkesson
University of Illinois at Urbana-Champaign, Urbana, IL, USA

Introduction

Evolution is a slow process, involving stochastic changes in DNA, followed by trial through survival and reproduction. Mutations in long-refined DNA codes are unlikely to make an organism more fit. However, over millions of years, innumerable mutations gave rise to increasingly competitive, biodiverse life forms that made the Earth's ecosystems liveable, resilient and fascinating. Thus, in healthy ecosystems of the 'real world' to which we evolved, natural biotic and abiotic resources produced a self-renewing symphony of energy and nutrient transfers, growth, reproduction and decay to enable microbial, plant and animal life forms to thrive together for the long term. Increasingly, however, we live in the artificial world of human heavy-handed modifications – one which is increasingly unstable and which has begun to fail to provide the regulated climate, clean air and water, fertile water-retentive soils and natural controls on disease that have made our development and that of a multitude of other species possible, sustainable and delightful.

For most of human history, small groups of *Homo sapiens* lived as hunter-gatherers. As with other omnivores, our ancestors presumably helped cull sick and weak animals and dispersed the seeds of food plants, while they also made temporary clearings for homes that were employed by 'edge-specialist' species. Accordingly, it seems likely that their net effect was to enhance biodiversity. Today, however, humans are rapidly depleting biodiversity, but continuation of downward trends is not inevitable. Because we live in a time of vastly degraded habitats and depleted native biota, with relict populations of a number of species still present in the wild, and because there are new tools at hand for more astute research and management, we have splendid opportunities to foster ecological recovery. Done well, ecological recovery would support not only improved health of wildlife, domestic animal and human populations, but also economic wellbeing. The question is how we can best apply our knowledge, wisdom, effort, innovation, governance, and investment to counteract ecological destruction, which in our view is the greatest crisis in recorded history.

The Problems – Human and Environmental

Despite ecological resilience, five major extinction events have prompted marked reductions in biodiversity. None of these were related to the activities of man. The current sixth extinction event, however, began with the development of modern hunting cultures, accelerated with the agricultural revolution and proceeded even faster with the ongoing industrial revolution.

Twelve thousand years ago, North America could be thought of as a 'super-Serengeti' with amazing megafauna (more than Africa). Condor-like birds with a 5-meter wingspan, ground sloths as big as hippos, elephant-like mastodons and mammoths, giant beavers, giant armadillos, "American cheetahs", several other species of big cats, giant wolves, giant bears, peccaries, antelope-like

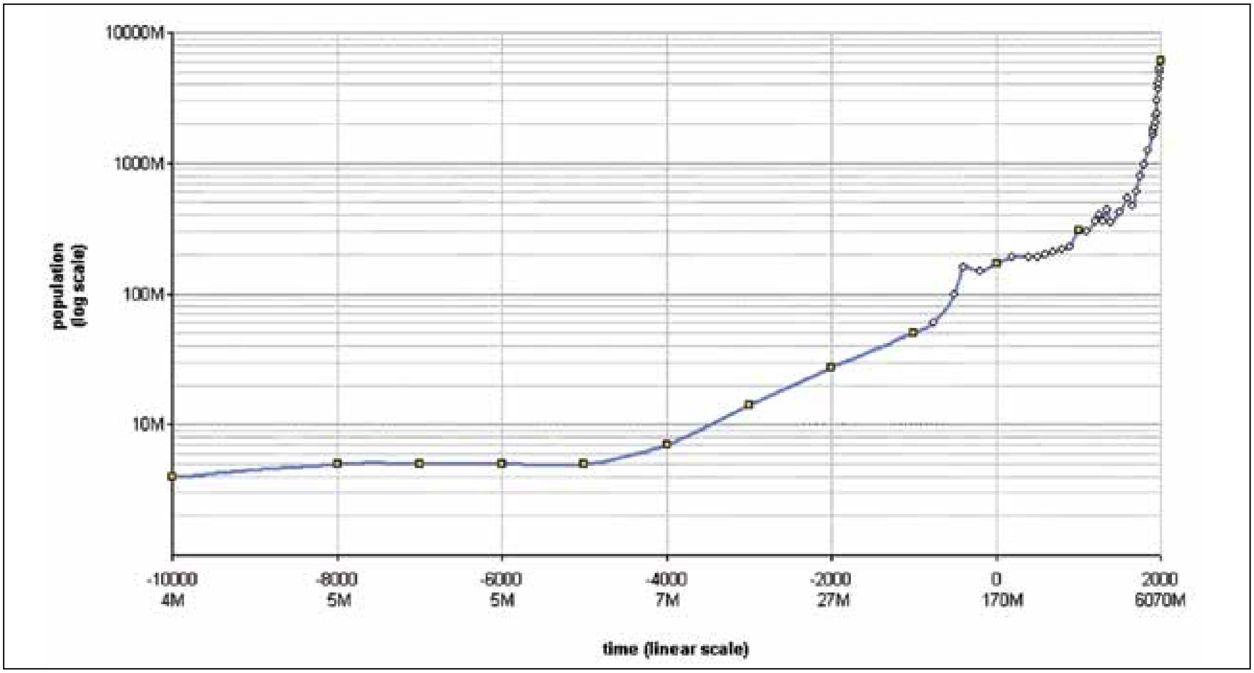


Figure 1.1. Estimated global human population growth from 10,000 BC to 2000 AD. Source: El T, (Wikipedia, 2010).

pigs, assorted camelids, several species of pronghorn antelopes, deer, stag moose, shrub-ox, Harlan's muskox, bison, tapir, and horse species, roamed across the continent. About eleven thousand years ago, roughly 95% of the mega-fauna were wiped out. There are two principal theories cited to explain this phenomenon: i) global warming with hotter summers and sometimes colder winters at the end of the Pleistocene, which would have contributed to off-schedule reproduction in plants and animals, and ii) efficient stone-age hunting by Paleo-Indians, who were widely employing arrows, spears and fire for hunting the larger animals of North America.Changes arising from the overkill of mammoths and mastodons would have contributed to expansion of forest cover and related food deprivation for grazing species and their predators. Eventually, humans began burning large areas of the landscape to increase grazing lands for species such as bison. Fires, of course, remove understory vegetation which deprives browsing species of food resources. A third theory relates to widespread diseases, some of which may have been transferred from Asia to North America via traffic of humans, their dogs, and perhaps their chickens across the "land bridge" made possible by the trapping of ocean water in glacial ice. A fourth theory involves an asteroid-induced or massive comet-induced inferno. In the authors' opinion, a combination of factors likely caused many of the extinctions.

Humans evolved in the presence of many species of African megafauna, many of which presumably had sufficient time to evolve defensive behaviours and thus to avoid extirpation by increasingly powerful homonids. However, even in Africa today, megafauna represent only about 70% of the genera that existed in the mid-Pleistocene era. About 50 genera disappeared from Africa forty thousand years ago, coincident with maximal advancements of Stone Age hunting cultures. By contrast, megafauna outside of Africa are believed to have been caught defenceless by a newly-arrived effective hunting culture, and the people happening upon strange new species may have been more startled by and thus more lethal to them.

After the elimination of a number of large species around the world that followed the spread of 'modern hunting cultures' across the globe and the devastating loss of important key species, extinction events became increasingly common. These were exacerbated by the domestication of crop plants and animals, leading to the spread of agriculture. More recently, the process accelerated further due to human population growth related to increased food supply, limits imposed on warfare, improving health care and inadequate birth control (Figure 1.1). The vast expansion in human numbers, coupled with greatly increasing resource consumption per individual related to the industrial revolution, the rapid development of modern marketing, and globalisation of the world economy are currently devastating the Earth's life support systems.

Wildlife cannot manipulate ecosystems and isolate themselves from ecosystem conditions in the manner of human beings. Instead, they integrate environmental change and, as a result, serve as vital sentinels of ecosystem health. At present, the signals they are sending are disturbing indeed. We are currently in the midst of an accelerating 'Global Biodiversity Deficit,' in which extinction rates have been estimated to be 100-1,000 times faster than before humans existed. Of groups evaluated across the planet, 32% of fish, 30% of amphibian, 28% of reptile, 21% of mammal, 27% of coral, 35% of crustacean, 12% of bird, 26% of insect, and 45% of mollusc species are threatened with extinction (IUCN, 2010).

Summary of Problems

According to Albert Einstein, modern societies are guilty of "Perfection of means and confusion of ends." This infers that despite our industrial, agricultural, economic, and cultural innovations, societies need to refocus on sustainability and ecological renewal: i.e. ends that make sense. Specific 'Big Drivers' that are increasingly recognised as interacting to aggravate the Earth's sixth extinction event (equivalent to Wilson's bottleneck below) are societal factors that:

1. Promote human population growth.
2. Stimulate resource consumption per individual (the aspiration bomb).
3. Enable over-harvest, such as through over-fishing and poaching.
4. Introduce destructive alien and invasive species.
5. Undermine ancient hydrology (e.g. via dams, locks, embankments, drainage canals), and deplete freshwater (e.g. via inefficient irrigation, industry, and residential use).
6. Establish vast areas of production of monoculture crops for food, fibre and timber that fragment, degrade and eliminate natural habitat (contributing to widespread deforestation).
7. Prompt under-grazing, over-grazing, and desertification.
8. Pollute water, soil and air with toxic chemicals from mining, power generation, industry, businesses, homes, transportation, and pest control, as well as nutrients from agriculture and human wastes that contribute to toxigenic algal blooms in freshwater, estuarine and marine systems.
9. Cause microclimate and global climate change.
10. Degrade natural mechanisms of infectious disease prevention and control.
11. Cause manipulation, militarism and terrorism, which waste resources rather than learning from past mistakes and using resources in ways that reflect compassion for all life.

Of importance is that the forms of environmental stress just listed often interact. Their harmful effects may be antagonistic, additive, or synergistic. Fortunately, human foibles, including individual and collective ignorance, corruption, inertia and denial – which have enabled these stressors to persist – can be countered through energetic education and astute governance.

Human Population Growth, Security and Isolation from the Environment

Because of laudable concern for one another, we have been zealous in efforts to control human diseases, increase food supply and control warfare, but we have been incredibly ineffective in much of the globe in our efforts to check associated human population growth. The agricultural and industrial revolutions were/are intended to increase food security, safety, and comfort for growing human populations. But, unfortunately, the methods most widely used have decimated vast numbers of native

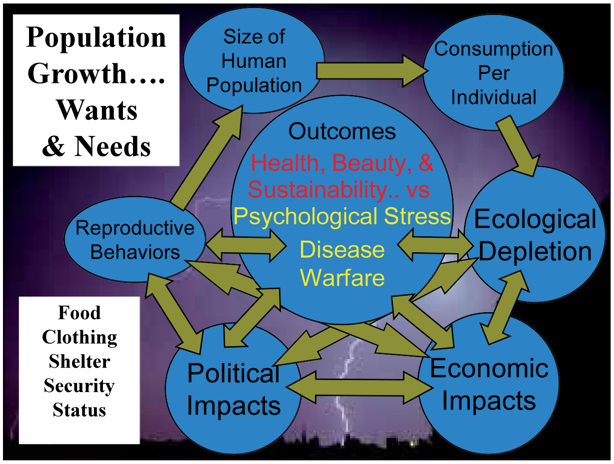


Figure 1.2. An illustration of interactions among human population growth, consumption rates, conflict, ecosystem impacts and health risks. A great deal depends on human choices as to how to address human needs and wants. Source: Author.

species in nearby and distant environments and thereby caused significant ecosystem disease. For example, urban sprawl and habitat conversion to agriculture coupled with inadequate management of wildlife with abnormal competition for feeding, loss and degradation of breeding and nesting habitats, predator-prey imbalances, and inbreeding have led to genetic impoverishment with related low fertility in wildlife, as seen in remnant populations of cougar (*Felis concolor*, Florida panthers). In the words of Dr. Milton Friend, founder of the USGS National Wildlife Health Center, "We live in an era that is creating wildlife ghettos" (Dolan, 1986). In this pattern of land and water management, wildlife are in a situation similar to that of humans crowded into the degraded cities and villages of medieval times, and it should be no surprise that they experience the effects of repeated plagues. *An important consideration is that, with our largely indoor lifestyles, people are no longer witness to the calamities that we are imposing on other organisms near and far.*

Excessive Consumption

Resource use by modern societies, which is generally highly inefficient, has vastly accelerated over the last two centuries. Since 1900, fossil fuel consumption has increased more than 10-fold and industrial production has increased more than 50-fold. Since 1950, the demand for grain, beef and mutton has tripled. Domestic cattle, sheep, goats and other livestock have often been allowed to overgraze, which erodes land and favors plants that are toxic. Other places that had historically been grazed are now undergrazed, leading to failure to build soil. Since 1950, fresh water use has tripled and thus solutes in rivers are less diluted. Indeed, parts of the Rio Grande at the Texas-Mexico border are so saline that fish die. Moreover, the Colorado River and Yellow River in China are often totally dry before reaching the sea (Navarro, 2006). Since 1950, the harvest of fish and other seafood has more than quadrupled, there are many international conflicts over fishing rights, and persistent contaminants threaten the offspring of fish-eating wildlife and human populations in many areas (Navarro, 2006).

Exotic and Invasive Species

Introduction of exotic and invasive species is another fundamental mechanism of ecosystem disease. Accidental introductions of problem plants have included neurotoxic and teratogenic poison hemlock (*Conium maculatum*), and deliberate, but short-sighted introductions of harmful plants have included nephrotoxic and calcium-depleting halogeton (*Halogeton glomeratus*) as well as highly invasive water hyacinth (*Eichhornia crassipes*).The latter holds water that can be utilized for reproduction by mosquitoes which carry diseases including malaria, and it forms such dense growths on the surface that insufficient sunlight reaches plants in the water column, thereby interfering with photosynthesis and the production of dissolved oxygen needed by fish. In general, exotic invasive plants cause problems because they take space, air, water, and nutrients needed by native species and/or they are highly toxigenic. Accidental introductions of invasive animals have included sea lampreys, which helped decimate lake trout in the Great Lakes of North America, and Asian tiger mosquitoes, which carry arboviruses to a host of vertebrate species including humans. Other accidentally-introduced species include Japanese beetles that harm plant communities in the United States, zebra and quagga mussels that clog water intakes and set the stage for harmful algal blooms in the US Great Lakes, brown tree snakes that caused multiple extinctions of endemic birds in Guam, and cane toads (*Bufo marinus*) that harmed local amphibian communities and poisoned other wildlife as well as domestic dogs and people in Florida, Hawaii and Australia. Deliberate but short-sighted introductions of animal species that became invasive pests have included exotic crayfish that prey on multiple native invertebrates and vertebrates, carp that destroy aquatic plant communities, and starlings that drive even woodpeckers from their nests. The introduction of exotic species may also bring disease organisms to immunologically naïve native biota, as has been the case with chytrid fungi that have decimated amphibian communities after infected African clawed frogs used in pregnancy tests were released to the wild around the world.

Altered Hydrology

Because of deforestation, horizon-to-horizon agriculture and urbanisation, the landscape cannot hold, filter or adequately purify rain water, and thus we are witnessing excessive flash floods, mud slides, soil losses and toxicant-, silt-, temperature- and pathogen-induced impacts on plants and animals in streams, estuaries, coral reefs, and other coastal zones. Grading of urban and suburban areas and placing drainage tiles under farm fields increases flashiness of streams, causing flooding. Urban use further depletes surface waters and aquifers. Irrigation leads to losses of water via evaporation and transpiration. During subsequent dry weather, due to the reduced ground water recharge, flow in streams is greatly reduced, such that they become hot which causes low dissolved oxygen, potentially harming or killing native fish and other aquatic animals. Mudslides and silt from erosion in agriculture, construction, and forestry degrade breeding grounds and harm fish gills. In human-altered environments, rain also results in pulses of hot and cold water carrying pollutants from roads, parking lots, roofs, construction sites, homes, lawns, and local industries. The damage to fish and shellfish communities is aggravated by factory-scale fishing.Dam-building, coastal development, filling estuaries and dredging reduce filtration and eliminate nursery areas needed by fish. Such changes also expose coral reefs to warm water, nutrients, metals, organic contaminants and particulates. Excess CO_2 released by fossil fuel burning and deforestation also equilibrates with carbonic acid to lower the pH in water, which may ultimately undermine the capacity of crustaceans, bivalves, and coral to precipitate calcium needed for rigid exoskeletons, shells, and reefs.

Increased use of retention ponds in recent decades has offered some benefits by reducing flooding and increasing groundwater recharge, but generally they have not been situated in corridors or populated by sufficient numbers of native species as to compensate for losses of biodiverse wetlands. These are too often placed near roads, which greatly undermines their value in conservation of wildlife, such as amphibians, reptiles, birds and mammals.

Monoculture Agriculture and Forestry

Monoculture agriculture, clear-cutting and monoculture forestry are increasingly leading to habitat simplification and loss. Fewer species are supported in the areas managed for human purposes, in part because simplified ecosystems provide few niches needed by wildlife specialists. Micro-climates are altered or lost, giving rise to adverse impacts even in adjacent 'wild' areas. The removal of biodiverse plants to produce genetically similar plants that are susceptible to attack by fungi and herbivorous insects also triggers the use of toxic pesticides. In addition, monocultures create a need for fertilisation. Fertilisation and pesticides often degrade soil biodiversity and related soil building as well as water retention capacity. The associated runoff of free nutrients commonly leads to harmful algal blooms.

Pollution

Simple spills from chemical transport such as oil tankers, trains, trucks, cars and pipelines, can be catastrophic. Areas with cold climates are especially vulnerable to petroleum spills because there is much slower removal through volatilization and biodegradation than in tropical, subtropical or even temperate locales.

Elemental mercury is still used to capture gold particles from deposits in streams in the Amazon, Africa, and Asia, leading to local environmental contamination, and impacts on animals and people. However, the greatest source of environmental loading of mercury is from burning coal. Mercury moves up the food chain as methyl mercury. Conversion to methyl mercury is achieved by ubiquitous sulphur reducing bacteria in sediments. Methyl mercury biomagnification causes toxicosis, leading to a classical ataxic stance in poisoned cats, with widely-spaced forelegs and knuckling over of the hind limbs. Toxic impacts of mercury in cats, fish-eating birds,

Figure 1.3. Photo of a frog with supernumerary limbs due to infection by the trematode Ribeiroia prior to metamorphosis. Photo: Author.

and fish can warn of risks to other species including humans. However, such sentinels can serve animal and human well-being *only if they are routinely monitored and diagnoses are efficiently established*. Mercury exposure of humans through consumption of contaminated fish causes functional deficits and teratogenesis, including major cerebellar/motor lesions, such as reduced size and hypomyelination of the corpus callosum and status marmoratus. Severe lifelong disabilities and deaths have occurred in people from mercury toxicosis.

Chemicals deliberately produced by industry to solve problems (e.g. pesticides, drugs, construction chemicals, household chemicals) contaminate the environment. They are accompanied by accidentally produced by-products in final formulations and in gaseous, particulate, liquid and solid waste streams that leave production facilities. In some instances, accidental releases are catastrophic. For example, the discharge of methyl isocyanate, a precursor in the synthesis of the carbamate insecticide carbaryl, into a highly populated area of Bhopal, India, caused the deaths of thousands of people, and many others were severely injured. Exposure to complex mixtures of chemicals occurs at many dump sites. Moreover, in landfills and other places where wastes are deposited, spontaneous oxidative, reductive and hydrolytic reactions increase the myriad of toxicants emitted. In addition, burning rubbish in developing countries that lack adequate waste disposal systems increases the complexity of toxicant exposures.

Irrigation mobilizes elements and salts from soils into surface waters. Leaching of selenium due to irrigation and then evaporation has caused toxic concentrations of selenium with marked teratogenesis, leading to reproductive failure in waterfowl (Kesterson Wildlife Refuge in California). There, the contaminated artificial wetland was filled in to prevent further waterfowl exposures via selenium-contaminated invertebrates in the foodweb.

We believe there are still many shortcomings in the overall approach of ecotoxicology. Decisions are sometimes unduly influenced by political and economic pressures and often fail to address the greatest needs. Research can be too disconnected from the problems. Complex mixtures are rarely studied. Research has failed to sufficiently examine the indirect effects of contaminants. There are large numbers of risk assessments, but infrequent validation of the accuracy of the predictions. However, ecotoxicology also offers many advantages to society and the environment. It has helped terminate or reduce the manufacture and dissemination of some major environmental pollutants and has helped offset adverse effects by prompting the development of new, more environmentally-benign, man-made chemicals (more astute choices). In addition, it has helped basic and applied ecology become problem-driven, has helped set the stage for conservation medicine, and has begun to merge with ecological restoration (rehabilitation) science. Nevertheless, in the authors' opinion, ecotoxicology needs to be *far more humble* about unmeasured effects, respecting the precautionary principle.

Global Climate Change

Excessive reliance on fossil fuels (natural gas, petroleum, coal) and clearing of forests trigger a secondary mechanism of ecosystem disease: global climate change. Carbon dioxide arises in excess from fossil fuel burning and burning and clearing of land for cultivation. Methane is produced from fossil fuel industries, microbial fermentation in herbivores, manure, rice cultivation, biomass burning, wetlands, and water bodies. As snow cover is lost from artic tundra, methane is increasingly released. These two gasses are the major drivers of the 'greenhouse effect' where infrared is reflected back to the earth rather than escaping to space. Wildlife that are intolerant of the hot conditions (or paradoxically cold, as may occur in Northern Europe) may need to move their range, but lakes, rivers, mountain ranges, deserts, agricultural fields, cities and highways are often in the way, prohibiting essential migrations.

As glaciers melt from the land and as warming ocean water expands in volume, islands and coastal areas experience increased flooding. Coastal flooding increases waterborne diseases, including human cholera. With these changes in atmospheric gasses, climates also become erratic, prompting fires when weather is excessively dry, as well as floods from extreme precipitation events. When glaciers are finally gone, the rivers they fed run dry.

With warming, the generations per year of a range of arthropod vectors (mosquitoes, ticks) increase, so more are available to carry their diseases to hosts in established locales as well as those in higher elevations and areas farther from the equator. Also with warming, increased generations of toxigenic phytoplankton can cause increased risks of poisonings. Many of the resultant harmful algal blooms produce toxins that move up the food chain to poison carnivorous water birds, marine mammals, and humans.

Figure 1.4. Baboon with a loaf of bread taken from a tourist in a commonly visited national park of an African nation. Photo: Author.

Infectious Diseases

As human populations have increased their numbers and extended their range, placing themselves, domestic animals and wildlife under stress, one result has been an accelerated onslaught of epidemics of emerging and reemerging diseases. Abandoning a mobile hunter-gatherer existence and settling into close quarters with domestic animals and rodents gave rise to many major human epidemics including influenza and bubonic and pneumonic plague. The incidence and severity of infectious diseases are often aggravated by ecological simplification. Thus, with conversion of complex wetlands into simple farm fields, migrating ducks and geese must crowd into remnant and artificially constructed water bodies, presenting risks of influenza and salmonellosis to domestic animals and people. Moreover, the waterfowl themselves often experience mass die-offs, not only from botulism, but also from avian cholera and duck plague. Other human diseases related to mismanagement of wildlife include HIV-AIDS from non-human primates and SARS from bats and civets.

Often domestic animals bring infections to wildlife (spillover) and then the infected wildlife can transmit the diseases back to naïve domestic animals (spillback). Brucellosis arose in Yellowstone bison from cattle, and bovine tuberculosis from cattle was contracted by at least ten species of wildlife in Kruger Park, South Africa. In the Serengeti, both rabies in wild dogs and canine distemper in lions originated in domestic dogs. *Toxoplasma*, likely from domestic cats, and *Sarcocystis,* from opossums introduced to California, have harmed previously overhunted and still endangered southern sea otters off the coast of California.

In North America, shooting and poisoning of large carnivores have caused overpopulation in prey, including small carnivores and ruminants, potentially resulting in infectious disease outbreaks. It seems likely that feral cats (*Felis domesticus)*, an exotic invasive in California, and opossums would be reduced by a robust community of larger predators, but the latter were depleted by hunting and trapping in the area. Similarly, in the absence of large predators, raccoon populations have vastly increased in much of the US. Numbers of this species in many areas are regulated primarily by diseases, such as distemper and sarcoptic mange. The high numbers of raccoons present significant risks of infection in humans from both rabies and the parasite *Baylisascaris*. The absence of predator pressure may also contribute to the observed spread of chronic wasting disease in deer and elk because of the ability of large predators to select disabled prey. Absence of the large predators may give time for an increase in infective material (prion proteins) in the bodies of the affected mammals prior to their deaths. Other overpopulated prey species have become hazards to human populations, as with Lyme disease transmitted via ticks carried by deer and deer mice. Among the suggested solutions for overpopulated wildlife and the related elevation in disease risks have been "humane killing," birth control, control of food sources, public education, and re-establishment of corridors to enable the return of more robust populations of native large predators.

Warfare

Another fundamental mechanism of ecosystem disease is over-emphasis on warfare and under-emphasis on diplomacy.Diplomacy can include joint efforts to conserve mutually needed natural resources. Inadequate natural resources, such as fresh water, farmland, forests, fossil fuels, grazing land, fisheries and minerals, have – both historically and recently – culminated in widespread anger, terrorism and overt warfare. Nuclear warfare leads to blast force injury, thermal injury and radiation injury. Conventional warfare involves munitions such as bombs, artillery, rockets, mines, guns, and smokes/obscurants. Chemical warfare may involve exposures to neurotoxic chemicals, such as organophosphorous cholinesterase inhibitors and botulinus toxin, or potent mucosal irritants, emetics, and crowd control agents. Potential biological warfare include: anthrax, smallpox, plague, and many other infectious agents. Considering the human, social, and economic impacts, it can be argued that we need to do a great deal more to control warfare. Recognizing the scale of impacts of modern weaponry, there is a necessity for humane global governance with nested continental, regional and local stewardship. The alternatives include a far stronger United Nations, with global disarmament treaties, a strong world court and a multinational police force to remove despots and terrorists. Failing these or other enlightened options, we face risks of ongoing wars potentially leading to global devastation, or continuous war, potentially reducing human existence to conditions similar to that of "failed states" with feudal systems akin to Somalia. Rampant conflict and militarism should not persist when the resources wasted and degraded by warfare are needed by human, animal and plant communities as never before.

Interactions

When they are able to respond to wildlife health problems, natural resource agencies typically focus on short-term measures addressing a few species in remnant ecosystem fragments. When disease outbreaks are recognised, consultations with veterinarians may prompt diagnoses and control strategies. Although these can yield local benefits, as long as animals remain isolated in small, disconnected pieces of their former range, they are at great risk of population crashes not only from infectious diseases, but also from starvation, contaminants, inbreeding, extreme weather, predators and over-harvest. Each of these can become a terminal insult to extirpate the animals from the habitat fragment. As the process proceeds across large regions, species become threatened, endangered and ultimately extinct. Soon, co-evolved organisms that depend upon the declining or eliminated species become part of a downward spiral toward ecological simplification and reduced ecosystem services. Even more poorly understood than the direct effects of individual stressors on ecosystems are the possibilities of positive feedback loops and the potential for synergism among stressors that may accelerate environmental and health problems over time. Who among our leaders routinely addresses soil depletion, local and global climate change, loss of biodiversity, reduced binding and degradation of toxic chemicals and persistence of infectious organisms and their vectors? Yet all of these are harming society and are triggering responses involving major expenditures of work and money and/or conflict among stakeholder groups at the local to global level. Undoubtedly, when we cause large-scale degradation of ecosystems, the problems that develop within and among plants and wild animals, domestic animals and human beings are too many and too complex to anticipate, prevent or control. Because massive areas of landscapes, coastal areas and oceans are involved, we live in a time of an unparalleled global ecological catastrophe. Thus, humans and non-humans alike suffer from the effects of the imbalance between resource consumption on one hand and resource conservation and renewal on the other.

Ignorance, Inertia and Denial

Where are we headed? The current rate of elimination of wetlands is extreme, as is filling of estuaries, and 207,199 km^2 of tropical forest are being destroyed each year. If the human population doubles by 2050, it has been predicted that 90% of existing natural habitat and 25-50% of all species on Earth will be lost.

Overall, there is a deficit of creativity in accommodating human needs for food, clothing, shelter, security and status in synchronization with ecological needs. Shared regional stewardship of ecosystems is essential for wildlife sustainability, and can also help societies recognize other shared values. Unfortunately, classical approaches to environmental management have lacked systematic education, intervention, and research needed to ensure societal benefits from conservation of biodiversity in concert with robust "One Health" improvements (see Recommendations below).

An underlying and pervasive mechanism of ecosystem disease is ecological illiteracy. The world suffers from profound ignorance of ecosystem functions. Evidence for this includes rampant planning, industrial development, and marketing largely to create and fulfil human needs and wants such as food, clothing, shelter, transport, security, entertainment and status, while generally dismissing ecological impacts that result in animal and human health crises, economic losses and conflict.

Ignorance that enables persistence of ecological destruction is also revealed in the inertia of business and engineering. Both can be said to be largely stuck with heuristics. That is, if they did something before and it "sold" they will likely just try to do it that way again. An example is our transportation systems.The transportation goal is simply to get people, raw materials and products from place to place, but unforeseen (and largely unchecked) effects include trauma from vehicles, and stress from loss of habitat, obstructed movement, noise, heat, desiccation, food deprivation, and toxic particulates, heavy metals, salts, carcinogens, carbon monoxide, carbon dioxide, nitrogen and sulphur oxides, acid rain, ozone and smog. Inertia in governmental utilities is often evident in the predominance of coal fired and nuclear power plants. The goal is to provide people with electricity. The pollution from coal includes particulates, carcinogens, acids, and heavy metals, most notably mercury. Nuclear power remains a focus of developed countries, especially in Europe, and is gaining ground in the United States. Human errors in the design and operation of such plants have, however, revealed potential for area-wide catastrophe. The best known example is Chernobyl, the melt down of which caused high level pollution with short- and long-lived radioisotopes in a wide area of north-eastern Europe, causing increased cancer rates, especially in children. Many of the anticipated effects on wildlife health and sustainability of that region have gone largely unmonitored. Nuclear power plants have also become potential targets for terrorism.

Human Nature on a New Level

In an edition of Scientific American entitled 'Managing the Earth', William Ruckelshaus, the first US EPA secretary, stated: "If 80% of (people) are poor, we cannot hope to live… at peace. If the poor nations attempt to improve their lot by the methods we have employed, the result will be world ecological damage. Can we move nations and people in the direction of sustainability? Such a move would be a modification of society in scale to only two other changes: the Agricultural Revolution of the late Neolithic [Period] and the Industrial Revolution of the past two centuries. Those revolutions were gradual, spontaneous, and largely unconscious. This one will have to be a fully conscious operation, guided by the best foresight that science can provide—foresight pushed to the limit. If we actually do it, the undertaking will be absolutely unique in humanity's stay on Earth" (Ruckelshaus, 1989).

The 'Drivers' of ecosystem demise remain 'Big' because of our failure to educate students and the public at large, and because the public therefore retains incentives and disincentives that perpetuate destruction. Conversely, massive education programmes can overcome collective ignorance and denial, to ensure restructuring of incentives and disincentives to favor ecological recovery and thus sustainability of human wellbeing. The sixth extinction cannot be undone, but it can be substantially slowed, and innumerable organisms that are now depleted can regain robust populations in the wild. Thus, with sufficient creativity and effort, our generation and the next can enable widespread recovery of biodiversity and improved health among wildlife, domestic animals and human populations on a local, regional and global basis. Moreover, if this is coupled with a sizeable infusion of equity and leadership by people of good will, an era of vastly reduced conflict could also arise.

Recommendations

Fortunately, there is a better way forward. As noted in *Veterinaria Italiana* (Vol. 45, 2009), One Health, which incorporates human, animal and ecosystem health offers ways to reconcile human and ecological goals. It therefore holds potential as the framework for a far brighter future. Ecosystem health and conservation medicine, which are components of "One Health," identify and inform others as to risk factors that confront wildlife and ecosystems and how they can be overcome. Ecosystem health and conservation medicine apply expertise and assume human responsibility for all native animal populations, especially those without owners. Strengthening of the bonds among wildlife, ecological, domestic animal, and human toxicology into "One Toxicology" similarly offers opportunities for better choices going forward (Beasley, 2009). If we consistently protected pet animals, livestock, and wildlife from chemical pollutants in home and outdoor environments, we would not have to worry so much about human health impacts from toxic agents.

According to E.O. Wilson in his 2002 book 'The Future of Life', "The great dilemma of environmental reasoning stems from ... conflict between short-term and long-term values." Wilson goes on, "To combine the two visions to create a universal environmental ethic is very difficult. But combine them we must, because a universal environmental ethic is the only guide by which humanity and the rest of life can be safely conducted through the bottleneck into which our species has foolishly blundered." By illustrating both the character and severity of problems and the means by which to enable the necessary changes, veterinarians, physicians, public health specialists, wildlife biologists, wildlife managers, ecologists, governmental leaders, workers in non-government organisations (NGOs), business leaders, politicians, religious leaders, and others can help provide a more astute and humane world, one in which human populations once again collectively enhance biodiversity. Experts, who are well-schooled and experienced in work to help wildlife, have a huge role to play in moving society toward a universal environmental ethic. In the process of helping wildlife, domestic animals and human beings, these individuals gain the voice, credibility, and the audiences needed to bring the public and policy-makers toward collective wisdom and astute management choices.

Establishing a nearly "universal environmental ethic" is best accomplished by sharing the overwhelming data at hand on trends regarding the world's wild animals and other species in ways that capture attention and encourage wise intervention.While the efforts to educate the public expand, there will remain a need for experts to focus on rescue operations (the crises at hand). Of course, the proactive approach of habitat protection and rehabilitation should remain a first choice approach to sustain wildlife and ecosystem health.

Are we accountable for our failure to prevent losses in wildlife from chronic and acute infectious and toxicological diseases in recent decades? What should we do about established, re-emerging and newly emerging infectious diseases that affect wildlife because of the activities of modern societies? What should we do about the introduction into the environment of anthropogenic chemicals that disrupt wildlife health and ecosystem function? How can we contribute to safer environments for animals in the future, in areas other than infectious and toxicological disease control and prevention? It has become obvious that we can choose to continue muddling along without a plan, allowing gradual degradation/depletion of resources and resource wars, or we can learn from our past and establish flourishing landscapes, water bodies, and agricultural and business zones of the future managed in ways that ensure the largely synchronous recovery of biodiversity in concert with gains in wildlife, domestic animal, human and economic health.

Through emerging technologies, rapidly increasing collective knowledge and growth in collective wisdom, we have new options. Site-specific goals will include setting aside terrestrial and aquatic resources and implementing incentives, disincentives and regulations in order to:

1 Rehabilitate/restore small streams, rivers, wetlands, ponds, lakes, estuaries, coasts and oceans (includes mangroves, reefs, sea grasses, and coastal grasslands).
2 Re-establish connectivity of natural landscapes especially along streams and across uplands.
3 Maintain natural areas on the land large enough to sustain larger native carnivores, keystone species and other species that require large ranges.
4 Maintain areas devoted to human activities in small and large fragments linked to one another with transport systems that allow wild species to migrate and interbreed so that human-dominated landscapes once again become nested within natural ecosystems.
5 Provide ample buffers containing native plants and animals, so as to protect wild areas from noise, chemical and microbial pollution, and other forms of stress caused by human activity.
6 Harvest wild animals only at rates that allow recovery of robust, genetically diverse stocks.
7 Control introductions of exotic and invasive species, and reduce/eliminate invasive exotic species when feasible.
8 Control emerging and re-emerging diseases by a) facilitating natural disease control through protection of biodiversity, b) reducing crowding and stress in domestic species and wildlife, c) maintaining adequate distance between wildlife and human beings, and d) construction of astute food animal production systems (humane conditions and robust disease control).
9 Increase organic production to reduce the impacts of nutrient and pesticide pollution.
10 Practise agriculture and forestry using precision fertilization, methods that build soil, and methods that deplete neither aquifers nor water bodies.
11 Control chemical contaminants to below threshold concentrations for direct and indirect toxic effects (includes carbon tax to reduce CO_2 pollution).
12 Re-establish margins of safety for native biota by enabling their largely unobstructed continued evolution.

With modern mapping and coordinated landscape planning, and with implementation via public education and incentives/disincentives supported by citizens, governments and the private sector, we have the capacity to satisfy human needs and reasonable human wants within the constraints of strong trends toward healthier ecosystems. We need to ‘conserve’ land needed for natural habitats and buffers. Thus, we need incentives to develop or redevelop housing and shopping areas, away from critical habitats such as river and upland corridors, estuaries, coasts and barrier islands, and to enable human relocation to new areas when necessary.

One of the many careers in this framework will focus on working with human communities to sustain wildlife in remnant habitats and sometimes in captivity, while re-establishing viable, self-sustaining natural habitats. Astute management approaches, routinely refined over time to increase efficiency, should try to re-establish margins of safety for sustainability of a multitude of species of native plants and wildlife in largely natural, networked ecosystems. Such networked areas should be designed to surround and enclose large and small areas (fragments of human activity) devoted to agriculture, mining, manufacturing, commerce, education and housing. These fragments of human activity must be linked with one another by transit systems that enable terrestrial and aquatic wildlife to migrate as needed. For example, when a wildlife corridor encounters a major thoroughfare, bridges over or tunnels under the corridor should be constructed. Small-scale changes of this sort have already been implemented successfully. The *absolutely essential* need is to restore meta-population dynamics for wild species, even as local human communities develop and prosper. In most of the world, such benefits for wildlife are likely to accrue only when human populations are able to live in carefully-designed, highly desirable, secure ecosystem fragments.

Restoring the landscape in the near term may often rely on a ‘corridor and gate design’ such that the ‘gate’ can be closed and the ‘corridor’ controlled until a disease organism or vector or an exotic invasive species is eliminated. For example, a gate could be established in a forest ecosystem by temporarily converting it to open land with fences and perhaps guard dogs to prevent an exotic species from extending its range. In all instances, the mobility and the needs of the pathogen, the vector, or exotic invasive species would need to be deprived by the 'closed gate'.

Appropriate, site-specific buffers between human activities and wilderness areas will be needed. It may help to recall that we currently move homes and businesses to other locations to make room for super highways and shopping malls. We can do the same for conservation aims. We must not confuse the difficult with the impossible. An action that is impossible in the short term is usually fully feasible in the longer term (e.g. over the next 20-100 years). A prime example of a longer-term, rather successful effort is the Everglades National Park. A history of some of those involved in setting aside Everglades National Park is available at: http://www.evergladesonline.com/50years/forgot.htm. The local/regional/national approach used for Everglades National Park gives some pointers to draw upon today. In addition to the information provided on the above website, it is important to mention that some homes that were surrounded by the park were purchased right away, while other people who chose not to sell were allowed to live in their homes until they died. Because the land was needed for ecological rehabilitation, the homes could not be passed on to family members or sold privately. Thereafter, the government reimbursed their estates and acquired the property for the park.

The task of setting aside critical habitats in some regions is increasingly difficult, but in other places it is increasingly achievable. This is largely a function of societal attitudes and resources. With larger numbers of people moving to cities, there may be fewer left in the countryside. This may simplify negotiations needed for more logical allocation of lands to agriculture, forestry, and conservation. To assist in the reallocation process, we can rely upon new tools from the internet to radio, television, satellite images, spatial analysis, information technologies, health sciences, landscape analysis, ecological risk reduction, ecosystem economics and methods for consensus building to enable effective environmental action planning, implementation, and prioritized research. We should look toward effective NGOs to build upon their successes and learn from their frustrations.

Preventative Medicine at the Ecosystem Level

In large-scale health programmes, prevention is generally far more effective, economical and humane than treatment after the fact. Sustained monitoring of indicators of wildlife populations and health provides multiple opportunities to head off small problems before they spread. Animal population structure and health can also be rapid and sensitive indicators of the effectiveness of ecological rehabilitation efforts. Evaluating short-lived species with small ranges can provide evidence as to recent local conditions, while long-lived wider ranging species provide insights into large area, longer scale conservation success.

The framework for ecosystem health practice needs to focus on the environmental problems and then attract experts from relevant disciplines. Ecosystem health practice often begins with interactions among stakeholders (citizens, government, business/industry, NGOs) to identify target outcomes for both natural and purposefully-altered ecosystems, as well as the major threats to those target outcomes. These are the greatest environmental problems at hand and are addressed through mutually agreed countermeasures. Public education, new incentives/disincentives, product substitution, regulation, taxation, re-engineering, land-use changes and biomedical interventions should be devised and implemented as needed. Problems that fail to respond to such interventions should be addressed with alternative strategies or become the focus of new research initiatives. Because veterinarians are trained in financially-constrained, comparative biomedicine, with the emphasis on prevention, they should be included in many ecosystem health teams. Important players in this work routinely include landowners, wildlife managers, wildlife biologists, governmental representatives, and NGO workers.

Careers to Make the World Work Again

We need to aim for what is needed: vastly reduced extinction events and strong recovery of biodiversity in virtually every area of the world. To overcome human overpopulation, we must educate women to give them independence and enable them to have careers. Also, increasing security in old age will decrease the need for high birth rates in less fortunate communities. We should engage the major religions to offer existing birth control options. Moreover, we should help to develop and deploy new birth control technologies.

Teams of wildlife biologists, veterinarians, wildlife and landscape ecologists, and public health specialists can become highly effective leaders in the conservation sciences and in economic and political pursuits. The most practical approach to improve ecological recovery, productivity and resilience will often be to accommodate human nature by offering new 'package deals' that are more attractive than those chosen by human populations at present. For example, urban sprawl remains a common theme in human settlement, despite the loss of farm land, forest and habitat. Sprawl often occurs with little regard for the expense of new roads, utilities, schools, and shopping areas or the resulting chemical pollution. However, if society restructures the incentives so that developers prosper more readily by building within established communities, and if home buyers are provided greater security, attractive prices for homes with good resale value, lower energy costs (for homes and transportation), better schools, rehabilitated green spaces, and greater prestige in established urban areas, sprawl may become a bygone trend.

An important consideration is that high-rise apartment dwellings have less surface area and can be far more energy efficient than one-family homes. However, "concrete jungles" with no plant life undermine human psychological well-being. Thus, for cities to meet human needs, urban landscaping and regular visits to nearby wild areas must be provided for children, and opportunities for interactions with "green spaces" must be readily available to all ages on a regular basis.

Setting aside new regional or national parks and reserves, community conservation, eco-tourism, private-owner conservation, management-driven hunting of wildlife, sales of wildlife to restock depleted areas, and other tools can all be used humanely on a combined population/species/ecosystem level. The methods employed at the incentive, disincentive and regulation levels will rely upon energetically and creatively educating and involving all ages of the public. The public must know the problems and the opportunities at hand. This can be achieved through formal education, demonstration projects, and extensive use of the media. Moreover, successfully engaging the dominant religions so that they widely support ecological stewardship could be the tipping point that ensures more humane conditions for wildlife, domestic animals, and people, the world over.

Defeating the "race to the bottom" – where worker abuse and pollution are tolerated by corrupt governments and desperate citizens – can be facilitated by taxing depletion of non-renewable energy sources and/or carbon dioxide, stiff fines for releases of toxic chemicals, news media exposure of polluters and sweat shops, managed incentives to balance work with workforces so as to decrease the need for emigration and immigration, and taking vital natural resources off the negotiating table (i.e. setting them aside as conservation/park areas for the long-term). Global pollution standards for industry and agriculture and proper worker treatment can also be brought about by short-term required product labelling as to practices employed, followed by negotiations to prevent import and export of products that rely upon unethical practices.

There is a great need to have respect for, and provide support of, tribal cultures. To counter intrusion of corrupt global laissez faire capitalism into areas where indigenous peoples are living as subsistence hunters and fishers, assistance can be provided in wildlife and ecological management so as to guard against depletion of native biota. Such peoples can also be strong advocates for abundance, sustainability, and health of wildlife as well as prevention of over-harvesting and control of pollution. Community conservation can be supported by ensuring local benefits for individuals and groups of stewards of the region. In areas with wildlife reserves, we must acknowledge and provide compensation for wildlife-associated crop and livestock losses. Also, it is essential to prevent human injury and death from interactions with wildlife – whether related to trauma or infection.

Extensive ranching of wildlife can allow the species to fulfil historical ecological roles and to meet human needs. Thus, traditional hunting cultures can be maintained. The work of Pro-fauna, which has helped restock national parks and indigenous reserves in Brazil, provides a useful example.

There is a case to be made for extensive rearing of ruminant livestock. Cattle can be surrogates for wild ruminant communities (faeces provide organic material to build soil, and increase water retention). Cattle have been used to successfully rehabilitate soils around mines that would not grow plants. Use of vaccines, buffers and guard dogs to fend off predators could be used more effectively to manage the interface between wildlife and domestic animals. Also, disease resistance genes can be developed in livestock through both natural selection and genetic manipulation. Extensive management of livestock, done well, can increase biodiversity while providing for traditional ranching cultures as well as pastoralist societies.

There is a case to be made for confinement rearing of livestock. Confinement can limit weather-associated stress on the animals. It can also prevent disease transmission between domestic animals and wildlife. It is important to appreciate the ongoing risks of influenza strains that cycle among waterfowl, poultry, swine, and humans. Poultry and swine can readily be produced with very limited risk in highly biosecure facilities. Such an approach can protect human health from a range of pathogens, maintain the profitability of animal production, and promote evolutionary gains and genetic diversity of nearby wildlife. Confinement systems can be humane if they are engineered and operated so that the animals are happy, healthy and entertained. Appropriate animal densities, with astute feeding and manure handling can also ensure that wastes are desirable fertilizers instead of pollutants.

Summary and Conclusion

Improved wildlife and ecosystem health will require vast increases in collective knowledge and wisdom in society to the point that private stakeholders, NGOs, governments and international bodies choose to work together to develop and achieve shared goals for regional ecosystems and human populations. As informed citizens demand appropriate action, there are many reasons to assume that more political, business and religious leaders will contribute to meaningful progress and justly take credit for the benefits accrued.

As stated by Janet Larsen, “While this may be the first time in history that a single species can precipitate a mass extinction event, it is also the first time in history that a single species can act to prevent it.” (Larsen, 2004).

Part B

Stewardship of Biodiversity

Authors: Jeffrey D. Brawn, Scott Citino, Kelly Estes, and Philip S. Miller

Coordinating Author: Jeffrey Levengood

Facilitating Recovery of Threatened and Endangered Species

2

Philip S. Miller
International Union for Conservation of Nature, Apple Valley, MN, USA
Scott Citino
White Oak Conservation Center, Yulee, FL, US

Introduction

The global biodiversity crisis has been discussed at length in the natural resource conservation community. Despite this familiarity, however, the grim statistics remain alarming. Of those species groups that have been almost completely evaluated using the IUCN's Red Listing process – mammals, birds, amphibians, and gymnosperms – nearly 20% of all described species are classified as threatened with extinction (http://www.iucnredlist.org/info/stats, accessed May 2008). Moreover, the number of threatened species in each of these groups (with the exception of mammals) has increased since 1996. Most notable in this regard are the amphibians, which have seen a meteoric rise in threatened status in just the past few years. This revelation has emerged from an effort to catalogue the world's amphibian biodiversity and to assess the true threat posed by the fairly recent introduction of disease by chytrid fungus infection around the world (http://www.globalamphibians.org, accessed May 2008). We can draw only one conclusion from even a brief review of data such as these: An abundance of the world's species are threatened by mankind's activities on our planet, and the situation is getting worse.

We now focus our attention on the United States for a more in-depth look at the issue of endangered species identification and management. The Unites States Fish and Wildlife Service (USFWS) is the federal agency responsible for conserving, protecting, and enhancing fish, wildlife and plants and their habitats for the continued benefit of the American people. The Service's Endangered Species Program oversees the identification of threatened and endangered species within the country, as well as the process of developing strategies to prevent extinction of the species within its range. The primary tool to achieve this is the Unites States Endangered Species Act (ESA), which was originally drafted in 1973. The primary goal of the ESA is to protect threatened and endangered species to the extent that they are no longer endangered or threatened (NRC, 1995).

As of May 2008, the ESA has been used to list a total of 1,353 species (609 animals, 744 plants) as either threatened or endangered in the US. Effective conservation management of these taxa requires detailed guidelines for identifying the threats to species survival, and for specific actions that can be taken to minimize the impact of these threats. Within the context of the ESA, these guidelines are collectively knows as *recovery plans*. In 1978, the United States Congress amended the ESA to require that the USFWS (or its marine analog, the National Marine Fisheries Service) develop and implement recovery plans for all listed species. This amendment was received with great enthusiasm within the broad conservation community, as effective recovery of threatened and endangered species was now seen as a more tangible process.

Since the species listing process began in the late 1960's upon passage of a precursor law to the ESA, only nine listed species have been declared extinct. Despite this one measure of success, realization of the larger goal of broad species recovery has been problematic at best. As of May 2008, 19 species within the US have been reclassified from endangered to threatened (downlisted) and 15 species have been removed from the list altogether through active recovery (delisted). In contrast, as of the same date of analysis more than 250 species are candidates for addition to the endangered species list, with a few additional species currently being proposed for entering the listing process. Statistics like these have prompted many critics of the ESA to claim that the law facilitates adding species to the list, but does a poor job of creating conditions that are suitable to taking them off the list. More to the point, they doubt whether this piece of legislation – called by many the world's strongest conservation law – is effective enough to justify its continued use.

The goal of this chapter is to review the concept of endangered species recovery as defined within the context of the Unites States recovery planning process; to evaluate the strengths and weaknesses of this process; and to discuss ways in which endangered species recovery methodologies can be strengthened with existing tools used both within and, sometimes, outside the traditional professional conservation biology community.

What is a Recovery Plan?

The US Fish and Wildlife Service defines *recovery* as a process by which threats to a species' continued persistence are reduced or removed, thereby ensuring the long-term survival of the species in its wild habitat through population growth to more stable numbers. To recover a species, the Service will create a Recovery Team that is mandated to create a recovery plan, ideally within 2-3 years of the date of listing. Recovery plans provide a sort of guidebook that outlines the threats facing the species of concern, and specific management recommendations targeted at the federal, state, and/or local governmental level that will lead to mitigation of the identified threats. More importantly, additional recommendations are created that engage private citizens and organizations in the larger conservation effort. This is very important to the successful recovery of many endangered and threatened species in the United States, as significant portions of a species' habitat is often found on private land. Through this emphasis on public and private opportunities for action, USFWS strives to ensure a truly collaborative environment for fostering creative solutions to today's biodiversity conservation needs.

Ideally, a species recovery plan should serve the following purposes (USFWS, 2004):

- Delineate those aspects of the species' biology, life history, and threats that are pertinent to its endangerment and recovery.
- Outline and justify a strategy to achieve recovery.
- Identify the actions necessary to achieve recovery of the species.
- Identify goals and criteria by which to measure the species' achievement of recovery.

Recovery plans can also serve the following secondary functions:

- Serve as outreach tools by articulating the reasons for a species' endangerment, as well as why the particular suite of recovery actions described is the most effective and efficient approach to achieving recovery for the species.
- Help potential cooperators and partners understand the rationale behind the recovery actions identified, and assist them in identifying how they can facilitate the species' recovery.
- Serve as a tool for monitoring recovery activities.

- Be used to obtain funding for USFWS and its partners by identifying necessary recovery actions and their relative priority in the recovery process.

Interestingly, no agency or other entity is required to actually implement the recovery strategy or any of the specific actions defined in the recovery plan. In other words, *a recovery plan is a guidance document, and not a regulatory document*. However, the ESA is written so that the recovery plan is recognized as the central organizing tool for identifying and guiding the management activities required to facilitate endangered species recovery.

As will be discussed in more detail below, the rigor of a specific recovery plan is dependent in part on the breadth and depth of scientific and other information used in the analysis. Those responsible for writing the ESA recognized this, and included a section in the Act that authorizes revision of a recovery plan when new information surfaces or when a change in species status is warranted. This represents a practical application of the familiar adaptive management approach to conservation – an approach explicitly designed to promote flexibility in the identification of important species management guidelines when new information frequently arises.

Important Elements of a Recovery Plan

The following list highlights a subset of components that make up an effective recovery plan, listed in the intended order of their presentation as outlined in the USFWS Recovery Planning Guidance Document (USFWS, 2004):

- <u>Background Information on Species and its Habitat</u>
  This section provides the fundamental information on the species' description and taxonomy; conservation status; population trends and distribution; life history/ecology; and habitat characteristics. Taken together, this review of available literature should provide a single source of biological information on the species.

- <u>Designation of Critical Habitat</u>
  Section 3(5)(A) of the ESA mandates that critical habitat be designated for each species at the time of formal listing. Critical habitat is defined as "*…the specific areas…on which are found those physical or biological features essential to the conservation of the species and which may require special management considerations or protection…*" This section is to lay out the type and extent of critical habitat, and the date when it was designated.

- <u>Reasons for Listing / Threat Assessment</u>
  This critical plan element should summarize the causes of species decline, and should also provide detailed information of the sources of those threats causing the decline. For example, habitat destruction could be a major threat for a given species, with the destruction occurring through agroconversion, commercial logging, etc. Threats included here should be discussed in the context of the five listing factors identified by the ESA:
  - The present or threatened destruction, modification, or curtailment of a species' habitat or range.
  - Overutilization for commercial, recreational, scientific or educational purposes.
  - Disease or predation.
  - The inadequacy of existing regulatory mechanisms.
  - Other natural or manmade factors affecting its continued existence.

  This section should also include a threat assessment: A formal approach to identifying threats to the focal species, the sources of the threats, and their relative contribution to compromising the species' status in the wild. This assessment usually results in a table that summarizes information on each threat, and can be invaluable in providing a single consistent source of threat data to all stakeholders in a species recovery planning process.

- <u>Recovery Strategy</u>
  The Recovery Strategy presents a summary of the Recovery Program for the focal species, based on the Background information presented earlier in the plan. This summary should be limited to just a few short paragraphs and should identify the key facts and assumptions underlying the proposed Program, the primary focus of the recovery effort (i.e., invasive species control or disease management), the overarching objectives of the program, and the identification of and rationale for specific recovery units (subpopulations) if appropriate. While this section may be com-

paratively short, it is extremely valuable for linking the species biological and human social information to the details of the upcoming recovery program.

- Recovery Goals
  The long-term goals of the Recovery Program are specified here. Nearly all endangered species recovery programs have recovery (delisting) of the species as the ultimate goal. For those species formally listed as endangered, downlisting to threatened status can be listed as an intermediate goal to recovery.

- Recovery Objectives
  Each Recovery Goal can be subdivided into specific objectives that describe the condition necessary to achieving the Recovery Goal. For example, a Recovery Objective could be the maintenance of adequate breeding habitat to ensure high levels of reproductive success and, by extension, minimize inbreeding depression.

- Recovery Criteria
  In order to determine if a specific Recovery Objective has been completed, specific parameters must be fulfilled. These parameters are defined as Recovery Criteria. These criteria provide the targets by which progress towards achieving recovery can be measured. The criteria should be objective and measurable and, where appropriate, should be written in reference to the five listing factors described above.
  An example of Recovery Objectives and associated Criteria is given in Table 2.1.

- Recovery Program
  The Recovery Program section details the individual actions that are necessary to achieve the Recovery Objectives, and the monitoring programs required to track the efficacy of these actions. The actions described in this section should be action-oriented, and targeted to a level of detail that facilitates proper funding from the appropriate authorities. In addition, the narrative that describes the Program should include actions for conservation in the short-term – in other words, those that prevent extinction of the species within the next five to ten years – and those that lead to long-term recovery through downlisting or delisting. Recovery actions must also include specific actions to control each of the threats to the species identified in the Threat Assessment, as categorized under the five listing factors of the ESA.
  An example of actions included in a Recovery Program narrative is given in Table 2.2.

- Implementation Schedule and Cost Estimates
  Section 4 of the ESA explicitly states that recovery plans must include "…estimates of the time required and the cost to carry out those measures needed to achieve the plan's goal and to achieve intermediate steps toward that goal". The Implementation Schedule is designed to fulfill this need. Importantly, the Schedule must identify responsible parties to complete the individual actions, ideally so that management authorities can track the completion of the actions within the stated timeline. Each action is given a priority score from 1 (prevent extinction or irreversible decline) to 3 (achieve full long-term recovery) within the Schedule, and the actions are then listed in this order to clearly outline which actions must be taken more urgently.

An Evaluation of Endangered Species Recovery Plans: How Good Are They?

The Endangered Species Act has come under considerable scrutiny and outright criticism for many years. Finally, in the mid-1990's a number of proposals were introduced that would have dramatically restructured the Act – resulting in, among other things, an increased role for the recovery planning process and their associated recovery plans. In response to these proposals, many institutions were very interested in an evaluation of the rigor of recovery plans and their ultimate value to the conservation of biodiversity within the United States.

Unfortunately, such an independent assessment of recovery plans had not yet been conducted. In response to this identified need, the Society for Conservation Biology (SCB) proposed a comprehensive evaluation of endangered species recovery plans. This review would

Table 2.1. Example subset of Recovery Objectives and Criteria, adapted from the Recovery Plan for the Sierra Nevada Bighorn Sheep (approved September, 2007). Source: United States Fish and Wildlife Service, 2007.

Recovery Objective
Attain population sizes and geographical distribution of bighorn sheep in the Sierra Nevada that assure long-term viability of the overall population and thereby allow its delisting as an endangered species.
Recovery Criteria
Downlisting
A minimum of 50 yearling and adult females exist in the Kern Recovery Unit (Great Western Divide), 155 in the Southern Recovery Unit (Olancha Peak to Coyote Ridge), 50 in the Central Recovery Unit (Mount Tom to Laurel Mountain), and 50 in the Northern Recovery Unit (Mount Gibbs and Mount Warren), for a minimum total of 305 females. The number of females is the limiting factor in reproductive output because one male can produce offspring with several females.
Delisting
The minimum number of females required for downlisting per recovery unit has been maintained as an average for one bighorn sheep generation (7 years) with no intervention (i.e. population management, buffering populations through translocations, captive breeding, etc.). Herd status for delisting must entail at least three censuses, one at the beginning of the period (qualifying for downlisting), one at the end of the period, and one intermediate count for each herd unit. Maintaining this number of females over a generation should be sufficient to indicate that predation is managed and that the number of individuals within the population is large enough to promote regular use of winter range.

Table 2.2. Example text of actions comprising a Recovery Program narrative, adapted from the Recovery Plan for the Sierra Nevada Bighorn Sheep (approved September, 2007). Source: United States Fish and Wildlife Service, 2007.

1. Protect bighorn sheep habitat.
1.1 *Identify and acquire important habitat not in public ownership from willing landowners.*
1.2 *Maintain and/or enhance integrity of bighorn sheep habitat.*
2. Increase population growth by enhancing survivorship and reproductive output of bighorn sheep.
2.1 *Prepare and implement a management plan to temporarily protect Sierra Nevada bighorn sheep herds from predation losses, where needed, until viable herd sizes are reached.*
2.2 *Increase use of low elevation winter ranges.*
2.3 *Minimise probability of bighorn sheep contracting diseases causing mortality and morbidity.*
3. Increase the number of herds, and thereby the number of bighorn sheep.
3.1 *Develop and implement a strategy for translocations.*
3.2 *Develop sources of translocation stock.*
4. Monitor status and trends of bighorn sheep herds, their habitat, and threats to them.
4.1 *Develop and implement a monitoring plan for population abundance and distribution of bighorn sheep herds in the Sierra Nevada.*
4.2 *Monitor key predators in the vicinity of winter ranges.*
4.3 *Monitor vegetation structure and composition changes likely to affect bighorn sheep population parameters.*

be conducted by more than 300 researchers from 19 universities. The design of a detailed "data collection instrument", within which information from different recovery plans could be coded consistently for rigorous statistical analysis where appropriate, was a key component of the overall study.

A total of 181 recovery plans, spanning the full taxonomic and chronological history of the recovery planning process since its inception, were included in the final analysis (Hoekstra et al., 2002a). As many different people were involved in collecting and interpreting the recovery plan information, the data collection instrument was carefully calibrated from the outset to maximize the degree of data consistency across plans. Through this calibration, the researchers were able to confidently analyze the full set of recovery plan data across a wide breadth of questions, collectively designed to provide unique insight into the reliability and value of the plans in particular and of the recovery process in general.

The following is a summary of the important results to emerge from this analysis.

Influence of the Academic Conservation Biology Literature

Since the mid-1980's, the depth and scale of literature to come out of the academic conservation biology community has grown tremendously. But despite this growth of knowledge and understanding within the field, the stereotype of academics as intellectually and functionally separate from their action-oriented counterparts "in the field" grew stronger with each passing year. Consequently, there was real concern that this volume of work was having little if any impact on the practical recovery of endangered species. To evaluate the impact of this perceived schism, Stinchcombe et al. (2002) reviewed 136 recovery plans for any evidence of their evolution in response to the growing body of conservation biology literature.

Overall, as the amount of literature increased over time in four keys areas of conservation biology – population viability analysis (PVA), conservation genetics, metapopulation dynamics, and conservation corridors – recovery plans appeared to be incorporating this information more frequently. Specifically, more recovery actions were targeted toward collecting more detailed information in these areas, with greater priority given to the current or future application of methods in population viability analysis and conservation genetics. Actions related to metapopulation dynamics and conservation corridors was conspicuously absent from many plans. Therefore, while there appeared to some positive response to the growing academic literature, there is still considerable room for improvement. This may be difficult, since at the time of the publication only 34% of recovery plans making up the SCB database have academic scientists as members of the recovery team that wrote the plan. In order to utilize the academic community more effectively, the USFWS must actively encourage more direct participation in the recovery planning process by conservation scientists.

The Role of Population Viability Analysis (PVA) in Recovery Planning

Population viability analysis uses simulation models of species demography and ecology to predict the most likely response of a population to changes in its environment through harmful activities by humans or though active conservation management of threats that may impact it (Beissinger and McCullough 2002; Reed et al. 2002; Miller and Lacy 2003). The tool can therefore be extremely valuable for prioritizing future research or management efforts, although serious concerns have been expressed about its irresponsible use in the face of considerable gaps in our understanding of species biology. Despite limitations such as this, PVA can be a valuable tool for detailed threat assessment and the establishment of quantitative recovery criteria.

Morris et al. (2002) reviewed the SCB recovery plan database to determine the use of PVA throughout the history of the recovery planning process. Although there was a significant increase over time in the proportion of recovery plans that presented results of a PVA or called for the collection of data that would be required for a future PVA, less than 50% of all recovery plans approved in the time period 1991 – 2002 called for such an analysis as part of the general recovery process. Moreover, the types of data collection recommended in the majority of plans would facilitate only the most basic PVA to be conducted: the so-called count-based PVA that relies solely on census data over a period of years, instead of more detailed population demographic data. More complex PVAs would be possible for <25% of recovery plan species. In response to these findings, Morris et al. (2002) urged the USFWS to enhance appreciation of PVA among recovery team members and associated Service personnel; to encourage the direct use of PVA in more recovery plans; and to link population monitoring protocols more directly to the data required to conduct a detailed demographic analysis.

The Use of Recovery Criteria to Guide Endangered Species Recovery

A final amendment to the Endangered Species Act was added in 1988, requiring all recovery plans to include objective and measurable criteria for delisting endangered or threatened species. Gerber and Hatch (2002) studied recovery plans in the SCB database to assess the degree to which recovery criteria were employed over time. They found that more than 80% of studied plans include at least one quantitative recovery criterion, and that the number of such criteria increased significantly over time since 1990. Moreover, there appears to be a positive correlation between the number of recovery criteria for a given recovery plan and the current status of a species. In other words, species that are characterized as improving in status tend to have a larger number of quantitative recovery criteria. This suggests that the specification of clear quantitative recovery criteria facilitates the progress towards species recovery.

On the other hand, the authors also found that a relatively high percentage of recovery plans contained quantitative recovery criteria that had no clear relationship to information on the species' biology. Moreover, this observation appeared to increase in frequency for plans approved since 1990. On a more positive note, species that were seen as improving in status had criteria that were tied more closely to information on their biology, furthering strengthening the apparent relationship between well-defined recovery criteria and the species' chances for recovery. Taken together, the analysis points to the value of specifying detailed quantitative recovery criteria for all species that are the focus of a recovery plan.

The Treatment of Threats in Endangered Species Recovery Plans

Species recovery depends critically on the proper identification of threats and the specification of management actions designed to mitigate them. However, many threats are quite complicated to describe and address effectively. Lawler et al. (2002) assessed the identification of threats among recovery plans in the SCB database, and found that various types of basic information (magnitude, severity, frequency, timing) was lacking for 39% of all identified threats facing the species. However, threats that were better understood had more recovery actions identified for them compared to threats that were more poorly understood.

Perhaps the most alarming aspect of this assessment was the observation that 37% of all threats that were identified as impacting species were not directly addressed through associated recovery actions. This may be due to a lack of understanding the threats or, more problematically, may reflect a lack of internal consistency in the content of the plans (see below). Whatever the reason might be, the authors concluded that a lack of basic understanding of threats and their mode of impact on threatened and endangered species may be hindering our attempts at recovery.

Internal Consistency in Endangered Species Recovery Plans

In order to have a major impact on species conservation, a recovery plan must possess a clear logic that lays out the major threats to the species, the justification for specific recovery objectives, and the articulation of clear recovery actions that directly address the primary threats endangering the species. In other words, the recovery plan must be internally consistent. Brigham et al. (2002) studied entries in the SCB recovery plan database for evidence of internal consistency. Overall, the authors found relatively high levels of consistency across key areas within recovery plans: management actions often address primary threats and/or target important data gaps. However, a major failure in consistency revolves around population monitoring, where many plans recommend monitoring schemes that do not directly link to species threats or to the identified species recovery criteria.

Overall, this analysis indicates a comfortable level of consistency within the bulk of endangered species recovery plans. To address concerns that came to light in the analysis, the authors recommend that more careful thought be directed to the derivation of monitoring schemes so that critical analysis of threat mitigation can emerge. In addition, monitoring protocols must be more carefully tied to the recovery criteria so that progress towards recovery can be observed more efficiently.

Monitoring as a Component of Endangered Species Recovery Plans

Because of gaps in our knowledge of endangered species biology and ecology, our decisions around conservation management decisions are filled with uncertainty. Consequently, our ability to monitor progress towards recovery is a vital aspect of the full recovery process; we must give ourselves the opportunity to adjust our practices if necessary as new information comes to light about the species and the threats it may face. Therefore, it is very important to evaluate the degree to which we include monitoring as an explicit component of species recovery plans.

Campbell et al. (2002) reviewed the role monitoring plays among recovery plans making up the SCB database, focusing on the extent to which monitoring was proposed in the plans as well as the degree to which the proposed monitoring plans were implemented. They found that actions focusing on monitoring population trends were the most commonly proposed and implemented; actions involving monitoring species demography, habitat quality, and the impacts of predators or competitors or exotics were recommended must less frequently. There appeared to be a marked taxonomic bias in the rigor of proposed monitoring schemes, with mammals and birds receiving more detailed monitoring attention compared to invertebrates. As stated above, proposed monitoring schemes in many plans did not appear to be consistently associated with the primary threats identified for the species. Finally, based on the common emphasis on population-level monitoring within recovery plans, the authors recommended against an emphasis on focal species monitoring as this likely reduces the attention that could be directed to other valuable forms of monitoring, e.g. density and impact of exotics.

Critical Habitat Designations in Endangered Species Recovery Plans

The designation of critical habitat is now a requirement of the ESA, but as of 2002 less than 10% of all listed species had such a designation. At the time of this SCB-mediated review, the USFWS saw addressing the backlog of critical habitat designations the top funding priority for their endangered species program. Hoekstra et al. (2002b) reviewed the role that critical habitat designation plays in the recovery plan process. A review of the SCB recovery plan database revealed that the designation of critical habitat did not increase the availability of information on the focal species' habitat needs, and it did not increase the frequency with which habitat management was prescribed in species recovery actions. Additionally, recovery plans that included critical habitat designations were not more likely to have habitat-based recovery criteria, although they recommended a greater number of habitat monitoring programs.

In total, the authors concluded that critical habitat designations had negligible positive impact on individual species recovery planning processes. To address this important shortcoming, they recommended a more standards-based system for designating critical habitat that would more effectively account for the biological and ecological needs of the focal species. This would lead to critical habitat designation constituting parameters *x*, *y*, and/or *z* instead of simply confining critical habitat within specific spatial boundaries, such as elevational gradients or bounded segments of a river.

The Value of Multi-species Recovery Plans

As the biodiversity conservation community moves to a more ecosystem-centric management philosophy, recovery plans for multiple species within a given habitat system become more attractive. By the beginning of this century, more than half of all ESA-listed species were covered by multi-species recovery plans. Clark and Harvey (2002) evaluated these plans for their rigor and breadth of coverage of important topics related to biologically sound recovery. Multi-species plans typically have a poorer understanding of the biology of included species, tend to lack a consistent adaptive management approach, and have a lower probability of being revised over time. Perhaps most importantly, these plans do not appear to be successful at choosing groups of species based on similarity of threats – a characteristic explicitly encouraged in USFWS recovery planning guidelines.

Based on these findings, the authors conclude that multi-species recovery planning is not as effective as single-species planning processes, and recommend that the multi-species planning process be critically evaluated before employing it further. Specifically, development and implementation of a more robust threat similarity index is warranted.

Revising Endangered Species Recovery Plans

The USFWS encourages the revision of recovery plans in response to new knowledge or to new events on the ground, but there is no consistent criteria to decide when and how such a revision should take place. Harvey et al. (2002) evaluated the SCB recovery plan database and found that recovery plans for vertebrates are four times more likely to be revised than invertebrates or plants, assuming no critical habitat designations. Knowledge of species biology and status appears to improve after revision, as does information on the presence and magnitude of threats impacting the species. However, this new knowledge does not lead to improved recovery criteria or monitoring programs. Consequently, the authors recommend a more stringent protocol for recovery plan revision triggers, as well as improved methods for applying new information to the specification of better recovery criteria and monitoring efforts.

Factors Affecting Implementation of Endangered Species Recovery Plans

Of course, recovery plans will be meaningless unless implemented on the ground. In their analysis of the SCB recovery plan database, Lundquist et al. (2002) found an average of about 70% of all recovery actions had been either partially or fully implemented within a given plan, although this value varied from 0% to 100% implementation across plans. As expected given other information from this review, recovery plans for animals had a higher degree of implementation than those for plants. On the whole, terrestrial species had a lower degree of recovery plan implementation than their aquatic counterparts, and multi-species plans are implemented more slowly. Recovery plans from teams with a dedicated coordinator had a higher level of implementation, as did plans that explicitly identified conflict among stakeholders as an element of the larger species recovery landscape. This effect of "social process" suggests that inclusion of diverse stakeholder interests within a structured, hierarchical team approach will lead to a more positive recovery process.

This observation was strengthened by Hatch et al. (2002), who evaluated the jurisdictional landscape and its influence on recovery development and implementation. Coordinating landowner and management agency domains is one of the biggest challenges to recovery planning. The authors found that species residing on federal land are more likely to be improving in status than those found at least partly on private land, indicating the difficulties in implementing recovery strategies in situations where public and private interests may be in direct conflict. Specifically, they found that increasing federal jurisdiction over recovering species leads to greater recovery plan implementation through a larger and more diverse body of stakeholder comprising the associated recovery team. However, a point of diminishing returns exists where an excessive number of implementing agencies on the team can inhibit an efficient path to recovery.

In conclusion, it appears that the variation in recovery success can be explained more completely by differences in how the recovery plans are created and how they're implemented – not simply by differences in the types and magnitude of threats the species encounter in the wild. This implies that careful consideration must also be given to the human sociological component of endangered species recovery planning – something that a traditional academic training in conservation biology does not typically address.

The collective results from this important analysis can be distilled down to the essential elements given below (USFWS, 2004):

What is working with the recovery planning process?
- Species with recovery plans in place for longer time periods show more improvement in status
- Most recovery plans have a fairly high degree of implementation
- High priority recovery actions are more likely to be implemented

What is improving?
- Emphasis on monitoring species is increasing
- Recovery criteria are increasing in specificity
- Scientific tools, such as population viability analysis, adaptive management, and metapopulation analysis, are being used more frequently

What needs more improvement?
- Explicit addressing and monitoring of threats
- Diversity of contributors, while keeping diverse teams manageable
- Monitoring of species trends, threats, implementation, effectiveness of implementation, and recovery criteria
- Internal consistency of plans, i.e., connecting biological information to recovery criteria/actions
- Inclusion of new science and theories
- Elimination of taxonomic biases
- Prioritization of species' plans for implementation and revision
- In multi-species plans, addressing of individual species needs, revisions, and implementation
- Addressing of needs for critical habitat management, where designated

Some Thoughts on Improving the Species Recovery Process

The biological science of endangered species recovery, while complex in its own right, is often comparatively easy when pitted against the sociological environment surrounding recovery. Those people involved in the threatening activities are nearly always in conflict with those attempting to manage the impact of those activities. Moreover, it's not uncommon to see different Federal agencies in conflict over the same endangered species management program – even if both agencies are devoted to improving the status of the species. These institutional conflicts often combine with our individual tendencies to see the world only through our own eyes and not through those of others. This combination makes endangered species conservation in a diverse, multi-stakeholder domain extremely complex (Westley and Byers, 2003).

Species recovery planning cannot be done in a sociological vacuum. Some organizations, such as the IUCN's Conservation Breeding Specialist Group (CBSG), have combined quantitative conservation biology with the complexities of human social dynamics to create a highly effective decision-making process that explicitly incorporates the perspectives and needs of relevant stakeholders in a meaningful way (see Westley and Miller, 2003 for additional information). This process has been applied to a small number of species within the United States, and the US Fish and Wildlife Service has expressed a high level of satisfaction with the process and the product that results from it – essentially forming the core of a new species recovery plan or the revision of an existing plan. The broader adoption of stakeholder-inclusive processes such as this, when combined with rigorous scientific analysis of the available data including thoughtful application of PVA methodologies, can only serve to improve our ability to direct conservation of endangered species with greater efficiency and efficacy.

Maintaining and Restoring Avian Habitat in Agricultural Landscapes

3

CASE STUDY
USA

Jeffrey D. Brawn
University of Illinois at Urbana-Champaign, Urbana, IL, USA

Introduction

Use of land for agriculture had led to the loss of over one third of the globe's natural habitat. And in the next 100 years, habitat loss owing to conversion for food, fiber and fuel production may reach 60% (Wade et al., 2007). Agricultural conversion coupled with increasing urbanization/suburbanization will place unprecedented strains on natural ecosystems and their constituent biota (Robertson and Swinton, 2005).

One inevitable result of conversion to agriculture is habitat loss and fragmentation. The direct effects of each on ecological integrity and biodiversity have been documented thoroughly for nearly all biogeographic regions (for example review papers, see Andren, 1994; Aguilar et al., 2006; Cushman, 2006). The magnitude of change varies with the ecological context and history of land-use, but there is little uncertainty that agricultural conversion can cause profound loss of biodiversity and ecosystem function (Best et al., 2001; Wade et al., 2007). Typically, only remnants of natural habitat are retained and the species that flourish are commensal with human dominated landscapes.

Human needs for land and natural resources will only increase; therefore, the challenge to conservation biologists and land managers is identify not only the problems associated with agricultural conversion, but also the solutions that will minimize local and regional losses of biodiversity (Kleijn and Sutherland, 2003). Increasingly, these solutions are to be found in the disciplines of ecological restoration and landscape ecology. The objective of this chapter is to briefly review the effects of agricultural conversion on biodiversity and to review the effectiveness of selected restoration practices with a focus on avian communities and populations.

Habitat Fragmentation in Agricultural Landscapes

Habitat fragmentation has two elements; habitat loss and isolation of the remaining tracts. The effects of fragmentation on biodiversity and the integrity of ecosystems are largely determined by the size of habitat tracts, the spatial arrangement and connectedness of these tracts, and the landscape matrix in which these tracts reside. The mobility and size of an organism determine the relevant spatial and temporal scales at which these factors exert influence. Birds, for example, are comparatively mobile and vagile; therefore, their population processes can operate at the local to regional spatial scale (Robinson et al., 1995). At more local scales, the habitat-matrix interface is especially important for avian productivity since "edge effects" can have profound influence on the nesting success of resident populations (Brawn and Robinson, 1996). Larger tracts of habitat in a landscape matrix that is not inhospitable generally support more diverse and viable communities.

In North America, much research on the habitat fragmentation has been conducted in the Midwest where the land's great value for agricultural production with intensive row cropping has led to chronic fragmentation and habitat loss (Donovan et al., 1997). For forest birds, the source-sink dynamic among spatially structured subpopulations appears to be especially applicable. Small habitat patches in a matrix of agriculture function as population sinks and, plausibly, ecological traps. The productivity of bird populations in small forest fragments is so low, owing to nest predation and brood parasitism, that many species likely persist only because of emigration from less fragmented and more productive source regions (Robinson et al., 1995; Brawn and Robinson, 1996). Similar findings have been reported in European landscapes as well (Baillie et al., 2000). The effects of agricultural conversion in North America have been most serious for species that inhabit grassland or prairie ecosystems. As a result, grassland bird communities are among the most threatened (Herkert, 1994).

Solutions for Habitat Loss and Fragmentation

Natural habitat in agricultural landscapes comes about by preservation or restoration. Much research has been devoted to assessing how to maximize the value of remnant or restored natural habitat. In the U.S., government sponsored programs to set aside agricultural land (e.g., the Conservation Reserve Programs) for soil, water and wildlife conservation have motivated research on the value of filter strips or conservation buffers for wildlife. These buffers are often created between rivers or streams and production fields. A study of 33 filter strips in Iowa found that many were too narrow to afford protection from nest predators characteristic of edge habitats. Notwithstanding, the authors concluded that the strips were of value to certain species of management concern (Heningson and Best, 2005).

Research on avian conservation and agriculture in the United Kingdom, where the government has committed to reverse the declines of birds by 2020, also indicate that linear habitats such as hedgerows and field margins can enhance the local diversity of birds, but the primary problem is management practices in fields themselves. In a more general study of the effects of environmentally sensitive farming practices on biodiversity in five European countries, Kleijn et al. (2006) concluded that the primary benefit was to species that are relatively common.

The function of linear habitats as corridors for enhancing habitat connectivity has been assessed for a diverse suite of habitats and species (Chetkiewicz et al., 2006). Generalizations about the value of these corridors for movements are controversial. The use and value of corridors depends largely on the resource requirements, behavior, and perceptual abilities of the organism under consideration (Baguette and Van Dyck, 2007). For highly mobile organisms such as birds, the value of movement corridors may be different than for species than cannot easily bridge unfavorable habitats. Notwithstanding, the value of corridors appears positive for terrestrial organisms such as butterflies and large mammals (Davros et al., 2006).

Restoration efforts in agricultural landscapes have proven successful in attracting and retaining unique communities and populations of birds – especially in disturbance mediated ecosystems such as grasslands and savannas (Fletcher and Koford, 2002; Brawn et al., 2001; Brawn, 2006). With managed application of ecological disturbance, such as the use of prescribed fire, otherwise poor habitats can be restored relatively quickly. The size of restored tracts can be an important determinant of the diversity of constituent animal communities in grassland and forest species. In contrast, communities of birds associated within disturbance-mediated habitat and ecosys-

Table 3.1. Selected research questions and themes for the management and conservation of birds in Agricultural Landscapes.

What are the implications of increased emphasis on cellulosic or food crop biofuel production? Do certain options provide better habitat for birds than traditional row crops?
What is the role of birds in integrated pest management? What landscape configurations enhance the effectiveness of birds as predators of crop pests?
Do conservation buffers in agricultural landscapes support viable populations of birds?
What size should restoration units be to attract a viable grassland bird community? What effect does the surrounding landscape matrix have on the community. Are certain birds of conservation concern area-insensitive?
What can be done to minimise the probability of nest predation on birds in agricultural landscapes?
Are patch connectivity and corridors necessary for highly mobile species such as birds?
What is the most cost-effective procedure for monitoring the success of restoration efforts in agricultural landscapes?

tems may be relatively area insensitive. For example, in North America, the size of successional habitats such as scrublands and oak savannas account for little variation in the diversity of birds (Brawn, 2006). Such "area insensitivity" enables land managers to consider the restoration and management of certain habitats on a relatively small scale in landscapes dominated by production agriculture and where the restoration of large tracts is simply infeasible.

Research Needs and Future Directions

In a comprehensive review, Wade et al. (2007) recommend a complex research agenda for establishing effective and sustainable management practices that will maintain some semblance of biodiversity where agricultural production is (or will be) demanding of land and other resources. A list of sample research questions and themes is provided in Table 3.1. Nearly all management options will require region-specific biological monitoring and significant support from governments to offer incentives to landowners. Fundamental questions for new issues such as biofuel production include the effect of alternative crops and feedstocks on biodiversity and the potential of native grasslands for biofuels production. Though the issue is not new, the role of controlling farm pests with biological agents such as insectivorous birds is receiving renewed interest (e.g. Van Bael et al., 2008) and merits additional attention. Growth in the resources needed to meet human demands will require innovative incentives and solutions for maintaining local and regional biodiversity. Close and mutual communication among researchers, managers and policy makers is essential.

Terrestrial Invasive Species of the Great Lakes Region

4

Kelly Estes
Illinois Natural History Survey
University of Illinois at Urbana-Champaign, Champaign, IL., USA

Introduction

Invasive species have threatened the Great Lakes region since the area was first settled by the English in the 1800's. As stated by the Great Lakes Commission (1999), more than 140 exotic, aquatic invaders have become established in this area. Nevertheless, terrestrial invasive species have had a tremendous impact on the Great Lakes region, especially in the last decade.

According to the National Invasive Species Council (2001) an "invasive species" can be defined as a species that is 1) non-native (or alien) to the ecosystem under consideration and 2) whose introduction causes or is likely to cause economic or environmental harm or harm to human health. Immigration was the only mode of arrival by organisms into new areas until several thousand years ago. Today, while immigration (or range extension) still occurs, many introductions of non-native species are facilitated by human activities (Hallman and Schwalbe, 2002). The means to which an invasive species is introduced to an environment is called a "pathway". There are several different types of pathways in which invasive species can be introduced into new environments (Hallman and Schwalbe, 2002; Mack et al, 2002).

Some organisms are imported for various reasons, but end up causing environmental problems rather than correcting them. For instance, kudzu (*Pueraria lobata*) was imported to help combat soil erosion. Now this fast growing vine is taking over large areas of the southeastern United States (Stein and Flack, 1996). Increased globalization that has accompanied the trade of goods and services has provided another outstanding means of transportation for these unwanted pests. Travel and trade between the United States and foreign countries has increased considerably. Direct airline routes have increased, thus increasing the speed at which cargo can be moved around the globe. Invasive plant pests can be transported into the United States from almost any point in the world within twenty-four to thirty-six hours (Mack et al., 2002). This timeframe is well within the survival time of many species. Hitchhikers can accompany commodities such as produce, nursery stock, or livestock and stowaways hide in solid wood packing materials or a ship's ballast water. The use of seaborne shipping containers continues to rise as well. Unfortunately, it is impossible to inspect every shipping container that enters a port. With the increased use of shipping containers, concerns lie with the possibility of new infestations in not only port areas, but inland where containers may be opened for the first time when reaching their final destinations (Hallman and Schwalbe, 2002).

Not only will a potential invasive pest affect the economy – Pimentel et al. (2005) estimates the economic damages associated with alien invasive species effects and their control amount to approximately $120 billion/year. Invasive species may also affect the beauty of our landscape, the diversity of our environment, and lead to the destruction of natural habitats. The Great Lakes area has suffered the consequences of many invasive species that have become established there (Great Lakes Council, 1999). This area continues to be on the look out for the introduction of new invasive species and the movement of others currently found here (Table 4.1).

The Asian longhorn beetle was discovered in Chicago in 1998. Following intense survey, public awareness and regulatory activities, the Asian longhorn beetle was declared eradicated from the previously infested neighborhoods in Chicago in 2008. Later that year, a single Asian longhorned beetle adult was found in the Deerfield mall parking lot. At the time this chapter was printed, delimiting surveys and inspections were still underway to find out more about this occurrence. Ironically, many of the trees infested in the neighborhoods that were under attack by the Asian longhorn beetle were planted to replace ones killed by Dutch elm disease nearly 50 years before. Dutch elm disease is one of the most familiar tree diseases in North America. It has continued to affect native and urban elm populations since 1930. While both of the Asian longhorn beetle and Dutch elm disease have had a serious impact in the Great Lakes area, let's look at two more invasive pests in greater detail.

Table 4.1. A selected listing of terrestrial invasive specie threats to the Great Lakes Region. Source: Author.

Insects	Asian Longhorn beetle (*Anoplophora glabripennis*) Emerald Ash Borer (*Agrilus planipennis*) European woodwasp (*Sirex noctilio*) Hemlock woolly adelgid (*Adelges tsugae*) Gypsy moth (*Lymantria dispar*) Oak splendour beetle (*Agrilus biguttatus*) Pine shoot beetle (*Tomicus piniperda*) Soybean aphid (*Aphis glycines*) Swede midge (*Contarinia nasturtii*) Viburnum leaf beetle *(Pyrrhalta viburni)*
Pathogens	Bacterial Leaf Scorch (*Xylella fastidiosa*) Beech bark disease (*Nectria coccinea* var. *faginate*) Chestnut blight (*Cryphonectria parasitica*) Chrysanthemum white rust (*Puccinia horianna*) Dutch elm disease (*Ophiostoma ulmi*) Karnal Bunt (*Tiletia indica*) Oak Wilt (*Ceratocystis fagacearum*) Plum pox virus (*Plu pox potyvirus*) Sudden oak death (*Phytophthora ramorum*) Potato golden cyst nemoatode (*Glodobera rostochiensis*)
Plants	Garlic mustard (*Alliaria petiolata*) Giant hogweed (*Heracleum mantegazzianum*) Japanese hops (*Humulus japonicas*) Japanese knotweed (*Polygonum cuspidatum*) Leafy spurge (*Euphorbia esula*) Multiflora rose (*Rosa multiflora*) Oriental bittersweet (*Celastrus orbiculatus*) Purple loosestrife (*Lythrum salicaria*) Spotted knapweed (*Centaurea biebersteinii*) Teasel (*Dipsacus syvestris*)

Emerald Ash Borer

Historical Background

The emerald ash borer (*Agrilus planipennis*) was first discovered in North America in southeastern Michigan near Detroit during the summer of 2002. It is thought to have been accidentally introduced into the United States in packing material made from ash wood in the 1990's. The native range of the emerald ash borer is eastern Russia, northern China, Japan, and Korea where it occurs on several species of ash. Since its introduction, the emerald ash borer has been found in nine other states and Canada (Figure 4.1). This beetle is responsible for killing millions of ash trees and threatens to kill millions more throughout North America.

Biology

The emerald ash borer generally has one generation per year. However, studies suggest that development may sometimes take longer in newly infested, healthy trees. In late May through August, beetles begin to emerge from infested ash trees; peak emergence occurs in mid- to late June. Trees display a small (0.3 cm) D-shaped hole in trunks and branches from where adults have emerged. For three to six weeks, adults with feed on ash leaves, mate, and lay eggs. Females will lay about 50-60 eggs individually on the bark surface or within bark cracks and crevices. Larvae hatch and tunnel into the tree. They feed on the phloem, creating S-shaped galleries just under the bark. Over the course of several years, the formation of these serpentine galleries disrupts the flow of nutrients and water, causing thinning of the canopy, branch dieback, and ultimately, tree death. Larvae feed

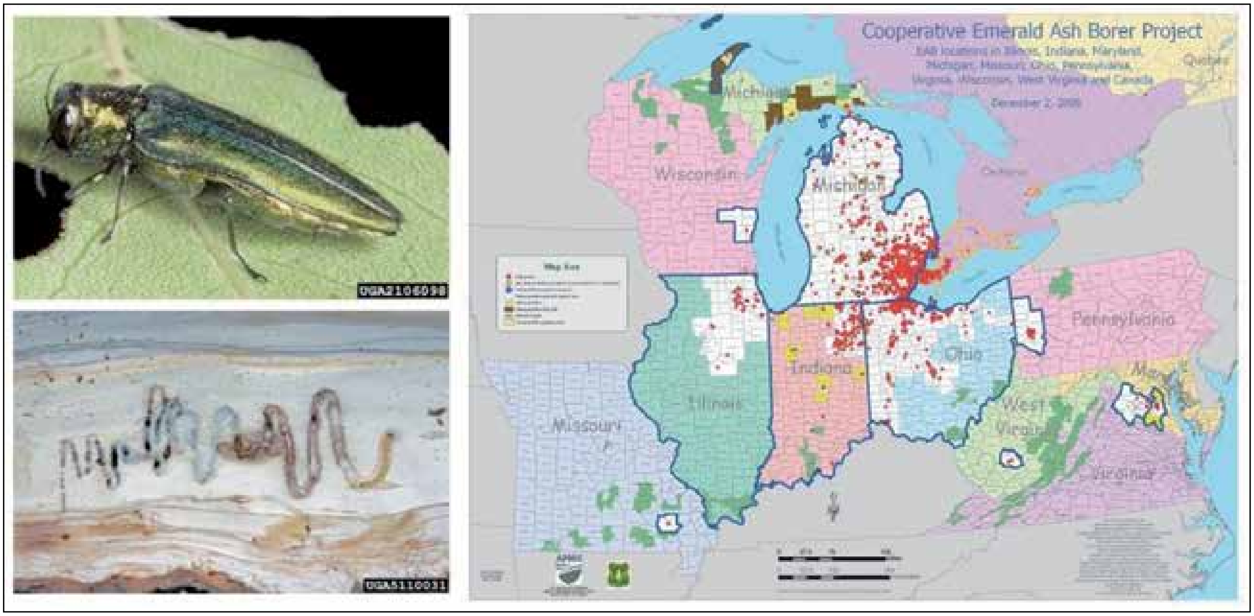


Figure 4.1. Emerald Ash Borer and distribution map. Photos: Courtesy of David Cappaert, Michigan State University. Distribution map: Courtesy of USDA and USFS.

throughout the summer and early fall before overwintering in the outer bark. In mid- to late spring, pupation occurs and adult emerge soon thereafter (McCullogh and Katovich, 2008).

Environmental and Economical Impacts

There is potential for the emerald ash borer to destroy all native ashes in infested areas because these native ashes lack an evolutionary history with the beetle. In an Ohio study, all native ashes (*Fraxinus americana,F. nigra, F. pennsylvanica, and F. quadrangulata*) have been shown to be sensitive to emerald ash borer. Ash species are dominant in eastern North American forests and well as being important nursery and landscape trees. Some losses are very straight forward and easy to put a financial value on, including the cost of tree removal (including stumps), and the cost of replacement trees. On the other hand, other losses associated with emerald ash borer infestations can be very difficult to assess. The loss in landscape value can include a multitude of aspects – reduced aesthetic value, increased heating and cooling costs, reduced property values, increased storm water runoff, and reduced wildlife habitat.

Control

Management options for emerald ash borer are still being widely researched. First and foremost, attempts should be made to limit the artificial movement of emerald ash borer, such as the movement of infested logs and firewood to prevent the spread of this insect to non-infested areas. In areas where emerald ash borer is present, large scale efforts including quarantines, infestation surveys, tree removal and outreach education to manage the emerald ash borer are primarily conducted by federal and state agencies. Ash trees can be preventatively treated for emerald ash borer, but this is not recommended unless infestations are within 24 km of tree and these treatments can be very costly as they required annual applications. Treatments are more effective when overall tree health is maintained. Imidacloprid and emamectin benozoate can be applied as trunk injections, and imidacloprid can also be used as a soil drench (Nixon, 2008). The efficacy of these treatments is currently being researched. It is also important to know that these insecticides may not be recommended or labeled yet in certain areas; please follow all label instructions. Insecticidal management options are continuing to be researched as well as biological

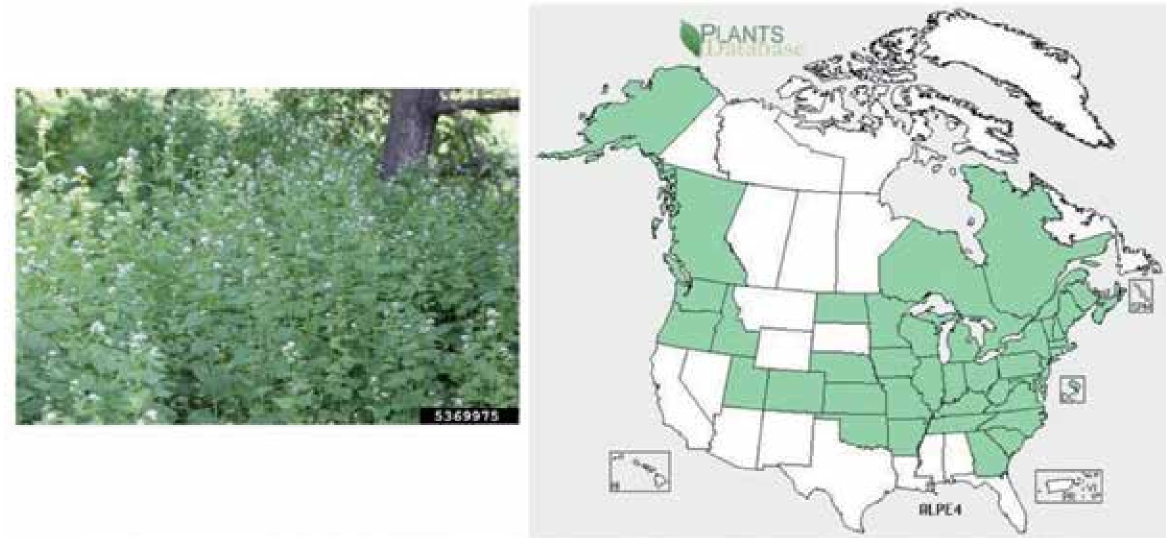


Figure 4.2. Garlic Mustard and distribution map. Photo: Courtesy of Steven Katovich, USDA Forest Service, Bugwood Org. Distribution map courtesy of Plant Database, USDA.

control options. In 2007, three parasitoids, *Oobius agrili, Tetrastichus planipennisi, and Spathius agrili*, were released in Michigan and will be evaluated over the next five years.

Garlic Mustard

Historical Background

Native to western Eurasia, garlic mustard (*Alliaria petiolata*) is thought to have been introduced to North America is the early 1800's by European settlers who valued it as a food and medicinal plant. In 1968, garlic mustard was first reported as growing in native communities in Long Island, New York (Kaufman and Kaufman, 2007; Nuzzo, 1993).

Biology

This biennial plant can be found growing along forest edges, riverbanks, and roadsides as well as deep within forest ecosystems. Widely spread throughout the Great Lakes region, garlic mustard can be found from southern Ontario and Quebec south to Virginia and west to Wisconsin (Nuzzo, 1993). However, it occurs as far south as Georgia, west to Oregon and Washington, and even in southern Alaska (Figure 4.2). During its first year, garlic mustard forms a rosette of leaves and in mid-spring the second year, sends up weak single stems, 30-40 cm high with small clusters of white flowers. Known for its high seed production, this invasive plant can take over an area in a matter of years. (Kaufman and Kaufman, 2007)

Environmental and Ecological Impacts

Garlic mustard is considered by some as the most serious plant invader of forested area. Its ability to overwhelm natural habitats and dominate the understory of forested areas is a cause for great concern. Traits such as shade tolerance and the fact it does not require a disturbance to become established or to proliferate, allow it spread at an alarming rate. Additionally, it may inhibit the growth of micorrhizal fungi that is needed by many native plants that use the fungi to obtain nutrients from the soil. Garlic mustard populations threaten spring-blooming wildflowers and forest ecosystems, affect tree growth and ultimately animals that depend on natural growth.

Little direct economic damage has been described or documented, nor has its ability to cause changes in forest productivity (Kaufman and Kaufman, 2007; Nuzzo, 1993).

Control

The key to garlic mustard control is to attack it early, before it becomes established. Once garlic mustard becomes widespread, eradication is very difficult. Hand pulling second year plants is effective when dealing with small infestations. The optimum time to pull plants is just after they have started flowering; the entire plant should be removed, including the roots. Larger infestations are best controlled with herbicides. Glyphosate, a safe and widely used herbicide, can be used early in the spring when garlic mustard plant growth begins (and before native flowering plants begin). First year plants can also be sprayed in late fall, after native plants have been hit with frost. Garlic mustard control is a multiple year process; seed in the seed bank can continue to emerge for several years. Researchers are currently investigating potential biological control agents to aid in the management of garlic mustard (Kaufman and Kaufman, 2007).

Conclusion

The Great Lakes region has dealt with the arrival of many unwanted, invasive plant pests and will continue to do so well into the future. Each has the potential to significantly impact the urban and natural landscapes of the region. Preventing the arrival of these pests is the best and most economical form of protection against invasive species. This is easier said than done. Thus we must rely on early detection followed by eradication or if the introduced pests are already established, contain their spread and attempt to minimize their affects on the environment.

Part C

The Baltic Sea

Authors: Magnus Appelberg, Anders Bignert, Britt-Marie Bäcklin, Natalia Chukalova, Sverker Evans, Elisabet Ekman, Lennart Gladh, Björn Helander, Anders Hellström, Leif Norrgren, Galina Rodjuk and Anna Roos

Coordinating Author: Leif Norrgren

History, Geological, Hydrological and Anthropogenic Features

5

Leif Norrgren
Swedish University of Agriculture, Uppsala, Sweden

The Baltic Sea, one of the largest brackish seas on the earth, is under severe ecological pressure as a result of industrialisation, agriculture, forestry, hydrologic and climatic conditions. The drainage area, which is four times the area of the Baltic Sea, is inhabited by approximately 100 million people (Figure 5.1). The Baltic Sea is divided into a few major basins (Figure 5.2) and is connected to the North Sea by the Kattegat and the Danish straits, both of which have constituting shelves restricting the inflow of salt water from the North Sea. Salinity decreases from 10-20‰ in the Kattegat Strait to 3-5‰ in the northern Gulf of Bothnia, forming a salinity gradient that affects the distribution of the biota. Well-defined haloclines and thermoclines stratifies the water mass. The biological diversity is low, with few and relatively simple food webs. The capacity of the Baltic Sea ecosystem to withstand environmental disturbances is limited and for many decades there are numerous alarming signs concerning the health of the ecosystem.

The history of the Baltic Sea basin started after a retreat of the last continental ice sheet (Weichselian glaciation) approximately 13,000 years ago. Since then the Baltic basin has periodically shifted between a fresh water lake and a marine sea (the Baltic Ice-Lake, the Yoldia Sea, the Ancylus Lake and the Litorina Sea). The evolution of the sea through these marine and lacustrine phases has resulted from climatic changes, which caused smelting

Figure 5.1. The drainage area of the Baltic Sea. Source: Hugo Ahlenius, UNEP/GRID-Arendal. http://www.grida.no/baltic/htmls/maps.htm

Figure 5.2 The Baltic Sea major basins. Map created by Norman Einstein. Source: http://en.wikipedia.org/wiki/File:Baltic_Sea_map.png

of ice, an increase in the level of the world oceans and subsequent rising of the land. The Baltic basin as it is delineated on a map today has developed to the brackish marine environment about 4,000 years ago. The present-day Baltic Sea covers 375,000 km^2 of water area, which is completely surrounded by land except for the narrow Danish sounds that connects it to the North Sea. It is a shallow sea with a mean depth approximately 50 m, and the deepest place, 459 m, in the Baltic Proper. More than 200 large rivers from the drainage area that hosts highly industrialized countries, discharge water into the Baltic Sea. There is a great seasonal variation in the river discharges. The maximum run-off occurs in spring and minimum during later summer or in midwinter. The land of catchment varies geographically with the high mountain and forest areas in the northwest part of the Scandinavian Peninsula and the areas of agriculture in the south. In addition to the large catchment area, a long-range transport of airborne pollutants contributes significantly to the contaminant burden of the Baltic Sea. The water turnover rate is slow and it takes about 25 years before the whole water body is replaced. Water temperature is fairly low which means, that deterioration of organic contaminants is slower compared to warmer water. Due to short history, many of the Baltic Sea species have not have time to adapt genetically to brackish water ecosystem and are living at the edge of their physiological tolerance, thus being subjected to adaptation stress. These features make the ecosystem even more fragile and living organisms more sensitive to anthropogenic contaminants (Havsutsikt, 2003; Håkanson, 2002).

The Baltic Sea was considered healthy as late as the 1950s. Since then the health of ecosystem has deteriorated due to contaminants from expanding industry and urban areas and fertilizers from agriculture. Today the Baltic Sea is one of the most contaminated seas and, although the situation has improved during the recent 20 years, it is still a place with unacceptably high levels of many contaminants. During the 1970s, grey seals and the white-tailed sea eagle where close to extinction and the otter had disappeared along the coasts of the Baltic Sea. The causal links between pesticides and other contaminants and catastrophic development of the many top predators, grew. At the same time started the monitoring of xenobiotics and possible effects in the marine Baltic fauna, initiated by the scientific community and the authorities. The first monitored contaminants were mercury, DDT and PCB. Within the framework of HELCOM, the council of ministers agreed on a reduction of 50% for several of the "classical" contaminants (DDT, PCBs). With the monitoring activities focused on temporal trend assessment it was possible to show the results also in biological samples from the environment. It is essential to see how various measures to protect the environment work in practice. In some cases the reduction of contaminant burden in the ecosystem was faster than one would expect considering their persistence to degradation. This is especially true for pesticides like DDT and Lindane where the bans in Sweden and Western Europe implied decreasing trends in fish that were estimated to between 10 to 20% a year. For industrial contaminants like PCB included in a number of products, the decrease was slower, about 5 to 10% a year. During the last decades the release of phosphorus, nitrogen and other nutrients into the Baltic has led to an increase in primary production. Decomposition of this organic material causes further

Alien Species

The Baltic Sea is affected by invasion of alien species both fish and invertebrates.

One example is the black mouthed goby (*Neogobius melanostomus*), which has invaded the Batic Sea from the Kaspian Sea and the Black Sea. Another example the bentic polycheat *Marenzelleria viridis*, which first recorded in the Baltic Sea during the 1980s. This polycheat has increased dramatically and in certain regions such as along the Estonia and Lithuania, but also along the Swedish coast. Ballast water is one of the major pathways for the introduction of non-indigenous marine species. Unintentional introductions may include a variety of adverse biological as well as economic effects such as changes in biodiversity and trophic relationships, dense algal blooms and kills of caged fish, shellfish toxicity, clogging of fishing nets and blocking of water intakes.

Shipping and aquaculture have contributed to the invasion of around 100 non-indigenous species in the Baltic Sea region, of which about 70 have established viable populations. The establishment of new species/populations may cause considerable changes in the structure and dynamics of the marine ecosystem. Examples from other parts of the world show ecological and economic disasters as a consequence of introductions and stress the needs to minimise introductions of non-indigenous species to the Baltic.

Leif Norrgren

Figure 5.3. Black mouthed goby. Source: Swedish Board of Fisheries.

deterioration of oxygen conditions in bottom waters and especially deep waters in the Baltic Sea. Together with chemical contaminants, eutrophication poses the most serious threat and reason of concern to the future of the Baltic.

Eutrophication

6

Lennart Gladh
World Wildlife Fund, Sweden

Introduction

Eutrophication is a process where bodies of water, such as seas, lakes, estuaries, or slow-moving streams, receive excess nutrients. The most important nutrients in this context are nitrogen (N) and phosphorus (P). Dissolved in the water, the nutrients act as fertilizers and thereby enhance plant growth.

Both nitrogen and phosphorus are 'natural' substances that normally exist in the Baltic Sea. They are necessary for normal life in the sea. What has happened over the last 100 years or so is that human activities have added excess amounts of these substances to the sea. The Baltic Sea now contains four times as much nitrogen and eight times as much phosphorus as it did in the early 1900s. Although a small decline in the nutrient loading has been observed in recent years, little change in eutrophic effects has been recorded.

The imbalance caused by the abundance of nutrients has led to numerous changes in the ecological composition and state of the Baltic Sea. Certain plants and animals thrive, enabling them to increase in number and geographic spread, frequently at the expense of other species.

Some of the negative effects of the nutrient overload of the past century are:

- Excessive growth of plants and algae. There has been an increase in primary production by 30-70% in different parts of the Baltic Sea.
- Annuals such as green and brown filamentous algae have grown at the expense of the perennial bladder wrack, which in turn has had severe impacts on the littoral ecosystem.
- So called "algal blooms", some of them toxic, are frequent phenomena in the Baltic Sea every summer.
- Water transparency has been decreased by 2.5-3 meters as a result of the increase in biomass.
- There are changes in fish species composition. Economically less valuable freshwater fish species are thriving while the cod is severely affected.
- There has been a decrease in numbers and spread of predatory fish, such as pike, in coastal waters.
- Due to the high plant production that later decompose, oxygen is consumed at the sea bottom, resulting in oxygen-low or oxygen-free areas with little or no life – so called "dead zones".

Cod – Not Only a Victim of Overfishing

Eutrophication, in combination with the poor water exchange in the Baltic Sea deep basins, is also a threat to the reproduction of Baltic cod. Fertilized cod eggs need water of rather high salinity (12-15‰) to remain floating. Such salinity concentrations are only found at great depths, for example in the Bornholm Basin. The problem is that at those depths, the supply of oxygen is insufficient due to eutrophication. Newly hatched cod larvae require more oxygen in the water (> 2 mg/liter) than hitherto believed. Optimal periods for successful cod reproduction might, thus, become shorter, or even non-existing, in the Baltic Sea. With a decrease in inflow of high-salinity water, and increased oxygen-consuming processes in the deep waters of the Baltic, two vital components are in jeopardy.

Figure 6.1. Satellite image of algal blooms in the Baltic Proper. Photo: ESA/Envisat.

Algal Blooms

Algal blooms are mass developments of microscopic algae, called phytoplankton. These blooms are actually a natural phenomenon in the sea. Due to the eutrophication process, however, phytoplankton blooms have become more frequent and intense. Today, harmful blooms occur annually in the Baltic Sea. During the summer when large quantities of these microscopic, free-floating algae perform photosynthesis, multiply and lump together, we see them as a greenish, yellowish, brownish or reddish layer on the sea surface, or as a thick 'soup' in the water – an algal bloom. In most of the Baltic Sea, there are two major annual blooms, the spring bloom and the cyanobacterial (also called blue-green algae) bloom in late summer (Figure 6.1). In the southern Baltic Sea, autumnal blooms are regular, too. Additionally, exceptional blooms formed by various species can occur locally. Exposure to an algal bloom might cause nausea, irritated skin and eyes, gastrointestinal problems and fever. The toxins can linger in the water for some time after the algae have disappeared.

A Vicious Circle

Tonnes of phosphorus eventually sink to the sea bottom and bind to the bottom sediments. When oxygen depleted sea bottoms become dead zones because of eutrophication, the lack of oxygen leads to chemical reactions in the sediments causing phosphorus to be released back to the water a consequence. This phenomenon is called "internal loading" and scientists do not yet know how much phosphorus is added to the Baltic Sea through this process.

Where do the Nutrients come from?

Both nitrogen and phosphorus exists naturally in the environment surrounding us. But while nitrogen has a natural cycle phosphorus the excess of P is an limited resource. The natural amount of P is caused by erosion while the excess is due to mining of P rich minerals.

About half of the nutrients that enter the Baltic Sea originate from agriculture in the nine countries around the sea. The rest originate from poorly treated sewage, industrial processes, traffic and other combustion processes.

In agriculture, nutrients are both added from the outside, as mineral fertilizers, and produced on the farm, as manure. When the manure and fertilizers are stored, transported, and used on the fields, some of the nutrients will leak to the environment. How much is depending on soil composition, soil management, crops and weather conditions. Nitrogen is easily dissolved in water and transported with groundwater and rivers, while phosphorus is mainly "washed away" as particles and transported on the surface as run-off from the fields. Sooner or later most of these nutrients will reach the sea.

The amount of nutrients that actually reach the sea depends on:

- How the manure is handled and stored on the farm. Open systems more easily lead to leakage both to air and to water, while closed systems keep the nutrients contained.

- How and when manure and mineral fertilizers are spread on the field. Fertilizers that are spread in the growing season with equipment that can spread exact amounts help to reduce leakage.
- Which crops are planted and when. By keeping the fields covered with growth all year round, the farmer can keep the nutrients in the ground.
- How and when the land is tilled. The more the soil is cultivated, the more nutrients it will leak.
- The composition and quality of the soil. Sandy soils leak more than clay soils.
- The physical landscape around the field where the manure or mineral fertilizers are spread. With protection zones around fields and along water courses, phosphorus run-off can be caught.
- The physical landscape in and around the rivers and streams that transport the nutrients towards the sea. Naturally flowing and meandering water course together with wet-lands catch a lot of nutrients and act as natural treatment plants.

Emission Terminology

Emissions of nutrients are described and categorized in many different ways:

Airborne or Waterborne Nutrients?
Nutrients can enter the Baltic Sea either as gases or particles transported by winds through the air, or as dissolved substances or as particles transported with groundwater, streams and rivers.

Land- or Sea-based Sources of Emissions and Discharges?
Sea-based means that the activity or source is located at sea, like shipping, offshore oil drilling or other activities to extract resources from the seabed.

Anthropogenic or Natural Sources?
Anthropogenic sources are human activities (as opposed to natural processes) that cause emissions or discharges of airborne or waterborne pollutants or nutrients.

Point Sources or Diffuse Sources?
Point sources can be either stationary or mobile. One could say that point sources have some kind of constructed outlet, like a discharge pipe or a chimney. They can be clearly singled out – a factory, a power plant, a sewage treatment plant, an industrial complex, a vehicle – and subjected to specific conditions and requirements for the operation. One can, at least theoretically, shut a point source down from one day to the next.

Diffuse sources are much more difficult to delimit and do something about. It can be a soil that due to natural processes continues leaching nutrients for decades although no new fertilizers are added. Examples of diffuse sources are land run-off and leaching of nutrients from arable land and forest land, leaching of various substances from landfills and mining wastes, the joint emissions from combustion (although each individual vehicle or power plant is a point source). Single households in rural areas and villages are besides agriculture a major source for diffuse pollution. Measures against diffuse sources must be directed towards practices and handling of various substances in agriculture, forestry, transportation, energy generation, waste handling etc. The most cost effective (per kg P) measure to reduce phosphorus is to ban P in detergents. Poorly treated sewage still remains a problem some cities but due to EU directives and investments also in Russia this is a decreasing source. Waste and wastewater also contributes and can locally be a major source. Nitrous Oxide (NO_2) is a gas that is released into the atmosphere in combustion processes, in industry and in vehicles, for example. Through rain the nitrogen gets deposited on land and in the sea, where it acts as a nutrient for algae.

Global Warming a Contributor

Global warming is also stimulating eutrophication as higher amounts of precipitation will increase leakage and runoff from land-based sources. Higher temperatures in the Baltic Sea region will also increase the decomposition rates of the algae, compounding the effects of the nutrients. The most widely predicted consequences of climate change in the Baltic Sea catchment area are rises in average temperatures and increases in rainfall. These changes are expected to be most pronounced during the winter months. Increasing rainfall will change the hydrological balance of the Baltic Sea. Coastal waters will receive increased fresh water inputs, lowering their salinity, and this trend may have an effect on the whole Baltic Sea basin. At the same time, inputs of dissolved materials including eutrophying nutrients from the catchment area will also tend to increase.

Conclusions

The imbalance caused by the abundandance of nutrients, primarily P and N, has led to numerous changes in the ecological composition and the state of the Baltic Sea. Certain plants and anmals thrive, enabling them to increase in number and geographic sprea, frequently at the expense of other species.

Fish Communities

7

Magnus Appelberg
Swedish University of Agricultural Sciences, Öregrund, Sweden

The Baltic Sea has a species-poor fish community in comparison to other comparable marine areas, in average ca 100 fish species (EEA, 2002), with fewest species in NE, and highest species richness in SW. The fish species composition is the result of a fairly recent colonisation process. During the last 10,000 years, the salinity have changed from being fully marine to freshwater, ending up being brackish during the last 4,000 years. Today, the fish fauna comprise a mixture of marine and freshwater species, with most marine species found in the southwestern and central part of the Baltic Proper, and the freshwater and migratory species dominating in the Gulf of Bothnia, Gulf of Finland and in the lagoons in the SE Baltic Proper.

Cod *(Gadus morhua*), sprat (*Sprattus sprattus*) and herring (*Clupea harengus*) constitute the most important species in the open sea. Other important marine species are e.g. flatfishes, gobides and sculpins. Along the coasts freshwater species such as Eurasian perch (*Perca fluviatilis*), ruffe *Gymnocephalus cernuus,* zander (*Stizostedion lucioperca*), northern pike (*Esox lucius*) and cyprinids species such as roach (*Rutilus rutilus*), bream (*Abramis brama*), white bream (*Blicca bjoerkna*), rudd (*Scardinius erythropthalmus*) are dominating (Ådjers et al., 2006; HELCOM, 2006). Among cold water adapted species whitefish (*Coregonus lavaretus*) and vendace (*Coregonus albula*), as well as migratory species Atlantic salmon (*Salmo salar*), sea trout (*Salmo trutta*), grayling (*Thymallus thymallus*). The European eel (*Anguilla anguilla*) has also been common in most coastal areas, however, it has become less abundant during recent years

Historical Development and Recent Regime Shifts

Among factors structuring the Baltic fish fauna eutrophication, including oxygen deficit in deeper areas, fishery, climate change, invasion of alien species and habitat degradation are considered to be of special importance. Decades of large-scale eutrophication and depletion of top-predators has resulted in dramatic changes in the Baltic Sea ecosystem (Österblom et al*.,* 2007). Primary production has more than doubled since the 1920-40s (Elmgren, 1989, 2001), and the production of phytoplankton and macroalgae has become a serious environmental problem for large parts of the Baltic Sea (Bonsdorff et al.,1997; Jansson and Dahlberg, 1999; SEPA, 2006). Intensive fishing of large fish predators has affected most parts of the food web (Carpenter et al., 1985, 1993; Pace et al., 1999; Österblom et al*.,* 2007; Casini et al., 2008).

The fish community in the open Baltic Proper has undergone severe changes the past 100 years, including drastic reduction of the cod population, and changes in both sprat and herring abundances. Österblom et al. (2007) showed that reduced top-down control from seals and increased bottom-up forcing due to eutrophication largely could explain the historical dynamics of the main fish stocks (cod, herring and sprat) in the Baltic Sea between 1900 and 1980. During the last three decades integrated ecosystem analyses show that there have been two relatively stable periods in central Baltic Proper, 1974-1987 and 1994-2006 (ICES, 2008b). The former period was characterized by high abundance and recruitment of cod and herring, and also high abundance of the zooplankton species *Pseudocalanus acuspes*. During the latter period sprat have been the main planktivorous fish, and the zooplankton species *Acartia* spp. and *Temora longicornis* have been abundant. The intermediate period between the two periods was characterized by variable climate with no large saltwater inflows. Similar ecosystem shifts was also observed in other parts of the Baltic Sea within the same time period. It is suggested that the primary drivers to these shifts is decreasing salinity and increased temperature, i.e. possibly the effects of a climate change.

The shift from a cod dominated to a clupeid dominated system has resulted in substantial trophic cascades in both the open sea and in coastal areas. The increase of sprat has negatively affected growth of zooplanktivores (sprat and herring; Casini et al., 2006) and the breeding success of the fish-eating common guillemot (Österblom et al*.,* 2006). It also is suggested to have had cascading effects down the food web on zooplankton and phytoplankton (Casini et al*.,* 2008). Analogous to the shift in the offshore ecosystem, shifts in the coastal ecosystem have been observed. Despite a temperature increase since the end of the 1980s (Alheit et al*.,* 2005), abundances of the major coastal predators (perch and pike) seem to have declined in open coastal areas of the Baltic Proper since the early 1990s (Ljunggren et al., 2005; Ask and Westerberg, 2008). Simultaneously, low or variable recruitment of both species has been documented in the same areas (Andersson et al., 2000; Nilsson et al., 2004; Ljunggren et al., 2005). The reason to this situation is presently unclear, although factors such as increased predation (Nilsson, 2006) or other interaction with stickleback may contribute.

The development of the fish communities in Gulf of Finland and Gulf of Bothnia differs to some extent from that of Baltic Proper. In Gulf of Finland most open sea species, except for sprat, show a negative trend since early 1990s. These negative trends of fish stocks have been attributed to salinity decrease and frequently occurrence of anoxic conditions. In Gulf of Bothnia the pelagic fish species comprise herring, sprat and vendace. Sticklebacks have also become common in the open sea ecosystem (Swedish Board of Fisheries, unpublished data). Since late 1980s salmon and sea trout are together with grey seals the main predators on fish. Until the late 1980s, cod was found in quite high numbers in the gulf, but has since then more or less disappeared, possibly due to a combination of high fishing pressure and climatic change. Due to the low salinity, there are no suitable spawning sites for cod in the Bothnian Sea.

Baltic Sea Fish Species

Baltic cod is usually divided in two different stocks, one western and one eastern population, although there is some migration of fish between areas. Recruitment is variable and is dependent upon the strength of incoming year classes. Successful spawning of Baltic cod is related to the volume of water with appropriate salinity and oxygen levels (Köster et al., 2001), which in turn depends on climate driven salt water inflow and river run-off (Hinrichsen et al., 2002). Also ecological processes, such as the abundance and competition for zooplankton with zooplanktivores (herring and sprat, Casini et al., 2006) affect the recruitment success. Spawning of the eastern population is limited to the deep basins where fertilized eggs are neutrally buoyant. The total and spawning-stock biomass increased by the end of the 1970s due to favourable reproduction conditions in the southern and central Baltic Sea, where after it fell to a historical low level the most recent years (ICES, 2008a). The decline of the cod stock in the Baltic Sea in mid-1980s possibly was the result of a combination of high fishing pressure negatively affecting spawning stock, and decreased salinity and oxygen levels restricting the recruitment volume in the Bornholm Basin (Figure 7.1).

Baltic Sea **herring** comprises a number of spawning components. This population complex experienced a high biomass level in the early 1970s, but has then declined until 2001 (Figure 7.1). The southern Baltic Proper herring have declined, and the more northerly and somewhat smaller herring now dominate the catches (ICES, 2008a). Since 1990, mean weight-at-age has decreased by 15-45% across all age groups to the early 2000's, where after mean weights have stabilized, and now remain at a low level. As in the Baltic Proper, herring in Gulf of Finland has decreased since late 1980s and sprat has increased since mid-1990s. The herring population in Bothnian Sea was assessed to be at a relatively low level of until the mid-1980s, after which the spawning stock biomass more than tripled by 1994 (ICES, 2008a). Favourable environmental conditions (warm summers and mild winters) have possibly contributed to good recruitment. The most northerly herring population in Bothnian Bay is the smallest herring stock in the Baltic Sea, and largely influenced by environmental factors. Like in Bothnian Sea, the herring stock of the Gulf of Riga increased in the late 1980s. The year-class strength of this population is significantly influenced by climatic variation and mild winters in the second half of the 1990s is suggested to have governed several rich year-classes and an increase of the biomass.

Spawning stock of **sprat**, which was low during the first half of the 1980s have been at a high level during the last decade in Baltic Proper. In the beginning of the 1990s, the stock increased rapidly, possibly due to both strong recruitment and a declining natural mortality (effects of low cod biomass), and in mid-1990s it reached the maximum observed spawning-stock biomass (ICES, 2008a). During the same period, sprat abundance has also increased in Gulf of Finland and possibly also in Gulf of Bothnia.

Flatfishes of the Baltic Sea consists of turbot (*Psetta maxima*), brill (*Scopthalmus rhombus*), European flounder (*Platichthys flesus*), plaice (*Pleuronectes platessa*), sole (*Solea solea*) and dab (*Pleuronectes limanda*). Among those, flounder, turbot and dab are the most widely distributed, whereas the other species are restricted to the south-western part of the sea.

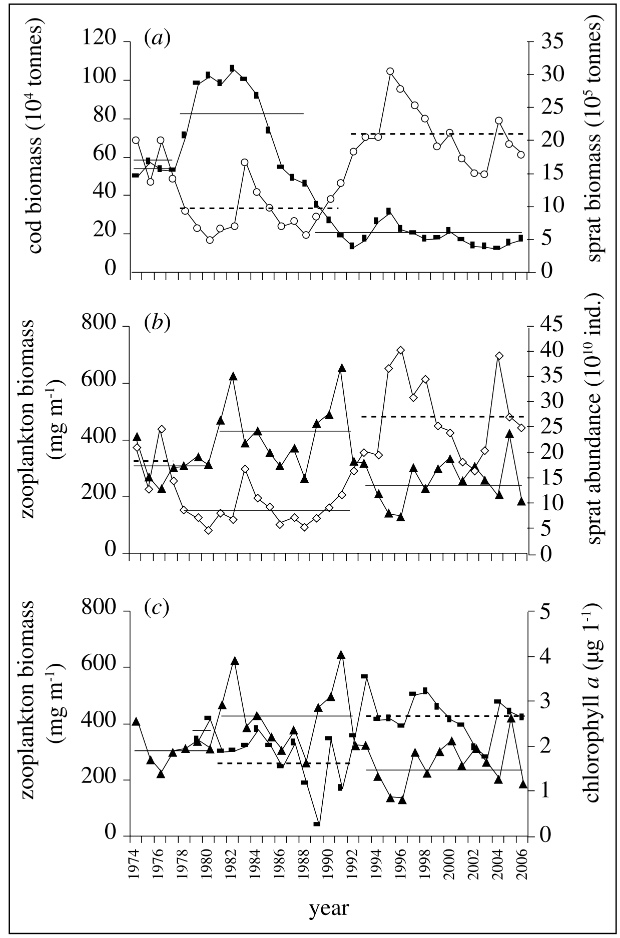


Figure 7.1. Trends of (a) cod biomass (squares) and sprat biomass (circles); (b) sprat abundance (diamonds) and zooplankton biomass (triangles) (ind., individuals); and (c) zooplankton biomass (triangles) and chlorophyll a (squares). The horizontal lines indicate periods of different average levels in the biological time series as detected by the cumulative z -scores. After Casini et al., 2008.

Flounder is distributed over the whole Baltic Sea, except the most part of Bothnian Bay. It appears to be separated in several distinct populations; earlier studies by Aro (1989) and Bagge and Steffensen (1989) suggested that there are eight rather northern distinct flounder populations. However, in a more recent study Florin et al. (2005) could separate three major flounder populations, one in Skagerrak/Kattegat, one in southwestern Baltic Sea, and one in eastern Baltic Sea. These three populations correspond with differences in spawning behaviour (Nissling *et al.*, 2002), with demersal eggs in the eastern Baltic, and with buoyant pelagic eggs in the south-western Baltic.

Turbot is a coastal species distributed over the Baltic Proper (Molander, 1964; Neuman and Píriz, 2000; Voigt, 2002). It displays a size and age dependent depth distribution where young fish are found in shallow water while old and large fish is found in deeper waters (Molander, 1964; Pihl, 1989; Støttrup et al*.,* 2002). Spawning takes place on sandy bottoms for feeding and reproduction, but migrates to deeper areas during winter. Despite a stationary behaviour, small genetic differences among populations indicate exchange between separate sub-populations. Females are generally growing faster than the males, which seldom reach lengths above 30 cm TL. An intense fishery in the mid 1990's resulted in a decrease in relative abundance, but the population has recovered since then. A continuous decrease of large individuals indicates that fishing pressure from both commercial and recreational fishery is high.

Plaice and **dab** are mainly occurring in the southern and western Baltic Proper. Both species are living close to the coast at sandy bottoms. The plaice population in the southern Baltic Sea is regarded to be weak, possibly dependent on migration from the Kattegat area. The population status of dab is uncertain, however, there are indications of an increasing trend (Ask and Westerberg, 2008). Dab from the Bornholm basin differs from those of the Belt Sea area (Temming, 1989), suggesting that the species comprise several separate stocks.

Atlantic Salmon populations in the main Basin and the Gulf of Bothnia was severely impaired by the damming of rivers along the coasts of the Baltic Sea. The development of hydro-electrical power resulted in extensive stocking programmes, strongly affecting the survival of wild salmon. A Baltic wide Salmon Action Plan was launched in 1997, and according to ICES (2008a) the total wild smolt production has increased about fourfold in the North-eastern Bothnian Bay and Bothnian Sea stocks since the plan was adopted. Wild smolt production is estimated to be about two thirds of the potential total smolt production, although smolt production is still low in rivers where salmon were extirpated and are now being reintroduced. In the Gulf of Finland salmon consists of only a few small wild populations together with a number of mixed/reared stocks. The wild salmon populations are genetically distinct from each other, indicating that these are still original salmon stocks. Surveys show that rivers where the stock mainly depends on enhancement releases still support fractions of the original wild salmon populations.

Survival through the post-smolt phase has decreased from about 20-30% during the 1990s to 10-15% for wild, and lower for reared, salmon during the years 2004-2006 (ICES, 2008a). The resasons to the decrease in post-smolt survival is unclear, however, an increased abundance of seals (both grey and ringed seals) during the same period is suggested to be one important factor. Despite a low survival at-sea the estimated smolt production has increased and rivers in the northern Baltic Sea is expected to reach 50% of their natural production capacity within some years. The status of less productive wild stocks, especially in southern Baltic Sea, is still poor. The proportion of wild salmon has increased relative to reared in catches, reflecting an increased wild smolt production. Commercial fishery has decreased recent years, whereas the recreational catches has increased and are likely to increase further.

High mortality rates in yolk-sac fry is caused by the reproduction disorder, the so called M74 syndrome. It was first detected in 1974 in Swedish Atlantic salmon hatcheries, where it caused increased mortality among alevins. Although the cause of the syndrome is still unknown, a linkage between the syndrome and a deficiency of thiamine has been established (Börjeson and Norrgren, 1997; Romakkaniemi et al., 2003). Norrgren et al. 1993 suggested that one factor involved in the etiology of M74 was organochlorine substances. This has later been supported by Michielsen et al. (2006) suggesting that oxidative stress may play a key role in the disease.

Sea trout in the Baltic consists of approximately 1,000 stocks, whereof about half are wild. Populations in Gulf of Bothnia are relatively stationary in the coastal areas, whereas most populations in Baltic Proper are migratory and spend there growth at sea. In the Bay of Bothnia, the populations are weak, possibly as the effects of fishery, migratory obstacles, and habitat degradation in spawning areas (Ask and Westerberg, 2008). In the Bothnian Sea, the situation is slightly better and the populations are showing a positive trend in abundance in some rivers. The development of sea trout in the Baltic Proper is negative, with the exception for the most southerly streams. Most of the stocks in the Baltic Sea migrate in the coastal area within about 150 km of the home river, but particularly those from Poland and some from southern Sweden and Denmark migrate further into offshore areas. The fish that migrate only short distances are mainly exploited in coastal and river fisheries. A recent ban of driftnets is expected to benefit the sea trout populations.

The immigration of **European eel** to the Baltic Sea has declined considerably; indices of young yellow eel recruitment show a reduction since about 1950 and is now just a few percent as compared to the mid 1900 (ICES, 2008c). The decline is more pronounced in the more northern and southern parts of the species distribution. The species is now regarded as one of the most endangered species in the Baltic Sea (HELCOM, 2006). Under natural conditions eel migrates to most freshwaters along the Baltic coasts, however, due to the extensive exploitation of rivers for hydro-electricity production, many of the migration routes are closed. In addition, eel is subjected to a high fishing pressure increasing the mortality of both growing yellow eels and mature silver eels returning to the Sargasso Sea. Recently, the European commission decided that national management plans have to be in force, with the goal to let at least 50% of the silver eel to return to there spawning habitats in the Sargasso Sea. If this management measure will be effective is not known.

Vendace is a pelagic freshwater species common in the coastal areas of Bothnian Bay. It is mainly fished for the roe, and the distribution is now mainly restricted to western part of the bay (Ask and Westerberg, 2008). As for herring, population size seems to be strongly related to climatic variation, although high fishing pressure also is a restricting factor. In 1990s the species was subjected to overfishing, but the situation has changed due to a number of strong year classes in the early 2000th

Whitefish display several different traits, and exists in two major forms in the Baltic Sea, one anadromous form spawning in streams and rivers, and one stationary form spawning in coastal areas on sandy bottoms. As for salmon and sea trout, the anadromous whitefish has over the years been negatively affected by the damming of rivers, especially in the northern part of the Baltic Sea. Due to stocking, the genetic variation among populations has become less pronounced, and river specific strains are uncommon (Florin and Aho, 2004). Fishery independent data suggest that the population abundance decreased in the late 1980s, and has been low since then (Ask and Westerberg, 2008). Additional reasons to this decline could be a climatic shift as well as a population increase of grey and ringed seals in the Bothnian Sea and Bothnian Bay. Seals are suggested to be one of the most important predators on whitefish presently. Like whitefish, **grayling** appear in one anadromous and one sea spawning form in the Gulf of Bothnia. The latter form is classified as endangered, but the reasons to the low population size are not clear. Attempts to restore the sea spawning form are ongoing.

Coastal Fish

The coastal fish communities comprise a variety of freshwater, marine and migratory species, and also glacial relicts. Species composition varies among Baltic Sea regions in relation to their different habitat characteristics, with salinity, temperature, and nutrient availability being the most important factors. Both the number of species and the abundance of marine species decline with decreasing salinity. Salinity also determines the distribution of most freshwater species.

On the west coast of the Baltic Proper, freshwater species such as perch, pikeperch and pike constitute the main predatory fishes. Frequently marine species such as young cod, flounder and turbot are present part of the year. Freshwater cyprinid species (e.g. roach, bream, white bream, silver bream, tench and rudd) are abundant, especially in more eutrophied areas. Among coldwater adapted species burbot and eelpout are common. In the eastern and southern coastal areas, salinity is higher and the marine species more common. The Curonian and Vistula lagoons in SE Baltic, being more or less freshwater habitats, are dominated by a variety of cyprinid and percid species, typical for eutrophied waters.

Eutrophication affects species composition and long-term development of the coastal fish communities, also resulting in increased production of fish biomass and changes in fish community structure and function. In general, eutrophication increases the abundance of

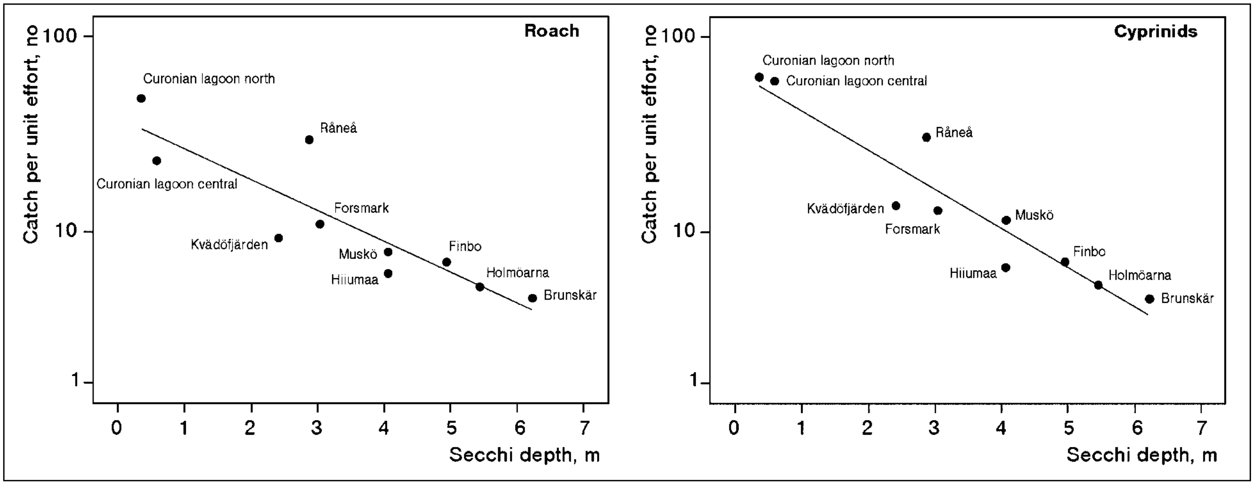


Figure 7.2 Relationship between Secchi depth and CPUE in terms of numbers of roach and other cyprinids. Significant relationships between CPUE and Secchi disc depth were found forboth roach (linear regression, p=0.032) and cyprinids (linear regression, p=0.006, After HELCOM, 2006).

cyprinids (Anttila, 1973; Hansson, 2007; Bonsdorff et al., 1997; Lappalainen, 2002). Water transparency, a proxy for high nutrient load, has decreased in the entire Baltic Sea during the past century (Laamanen et al., 2004). A relationship between trophic state, expressed as Secchi disc depth, and cyprinid fish is suggested by the negative relationship between water transparency and catch per unit effort of roach and other cyprinids in coastal areas (Figure 7.2). The smallest Secchi depth (i.e., least water transparency) and largest catches per unit effort of roach and other cyprinid species appeare in the Curonian lagoon, whereas the largest Secchi depth and small catches per unit effort of roach and other cyprinid species appeare in the Archipelago Sea, between Baltic Proper and Gulf of Bothnia.

Also climatic variation has had a substantial impact on the development of the coastal fish communities. Several coastal freshwater fish taxa living in the Baltic Sea (e.g., percids and cyprinids) prefer warm-water conditions. Temperature has been proved to be an important factor governing the recruitment success, growth, and year-class strength of, for example, perch in the Baltic Sea (Böhling et al., 1991; Karås and Thoresson, 1992; Karås, 1996, Appelberg et al., 2007). High water temperature gain population and invidual growth rate of species with high temperature optima (e.g. Figure 7.3), but restrain growth

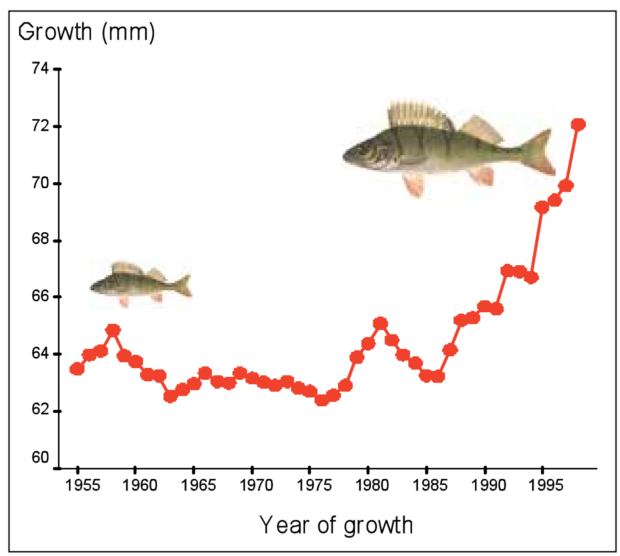


Figure 7.3. Growth of Y-O-Y perch in the coastal area of Baltic Proper. Change in growth rate reflects a temperature increase during the same period. Data from Institute of Coastal Research, Swedish Board of Fisheries

rate of species with low temperature optima. However, as climate change also affects e.g. primary production and land run-off, secondary effects may counteract the direct temperature effects.

The increase in sea surface temperature in the Baltic Sea the last decades is suggested to strongly affect the fish communities (MacKenzie, et al., 2007) and marine species will possibly be displaced by freshwater species due to increased precipitaiton and decreased salinity. In Gulf of Bothnia increased water temperature has resulted in a substantial increase in biomass of roach and bleak, and a reduction of whitefish (Appelberg et al., 2007). Significant increasing trends in relative abundance of European perch and roach have been observed in the Archipelago region between the southern Gulf of Bothnia and the northern Baltic Proper. Possible reason for these trends is an ongoing coastal eutrophication as well as increased temperatures during the past decade. In southern Gulf of Bothnia the two cold water adapted species, burbot and eelpout, display negative population development, despite an increase in relative biomass of coastal fish species the last 25 years. In some areas bream and pikeperch, both warm water adapted species have increased, possibly as an effect of a change in climate. In sheltered coastal areas in western Baltic Proper, number of species with high temperature optima increased during the same period. A strong indication of the effects of temperature change is an increased growth rate of perch starting in mid-1980s (Sandström et al., 2005)

Although the spread of non-indigenous species has been suggested to be among the most severe threats to global biodiversity (Leppäkoski et al., 2002), the significance of most non-indigenous fish species introduced into the Baltic Sea remain of less importance since they have failed to form self-sustaining populations. However, two species, the accidentally introduced round goby (*Neogobius melanostomus*) and the intentionally introduced Prussian carp (*Carassius gibelio*), are exceptions in this respect. In particular, the round goby is spreading from the Gulf of Gdansk region and new records of its occurrence are reported every year (Almqvist, 2008). The most recent report is from the southern coast of Sweden, where it was reported for the first time in mid 2008.

Vetema et al. (2005) reported that the invasive species gibel carp *Carassius gibelio* has increased significantly along parts of the east coast of Baltic Proper. The species, which was first observed in Estonian brackish water in 1985, has spread along the entire Estonian coastline, now being one of the most frequently occurring coastal species. They suggest that warm summers and low abundance of predatory fish are two major reasons to this change.

Conclusions

Baltic Sea fish communities are presently experiencing dramatic changes. Ongoing eutrophication, high fishing pressure, climate change, habitat degradation and spreading of invasive species, together with indirect effects from food web feed back loops, provides the basis for continous, and maybe irreservible, changes in the fish community structure and function. A drastic example is the recent reduction of piscivorous fish species (e.g. cod in the open sea area, which strongly have influenced lower trophic levels and possibly also affected primary production. Even if the nutrient load will be substantially reduced, it is probable that the retention from sediments and global warming will continue to affect the Baltic Sea, thereby governing community changes also affecting the Baltic Sea fish.

Aquaculture and Fish Health

8

Anders Hellström
National Veterinary Institute, Uppsala, Sweden
Natalia Chukalova and Galina Rodjuk
AtlantNiro, Kaliningrad, Russia
Elisabet Ekman and Leif Norrgren
Swedish University of Agricultural Sciences, Uppsala, Sweden

Introduction

Fish has an enormous value as the most frequent source of animal protein for the major part of the humans, especially in countries during rapid development in Asia and Africa. All fish species are a part of the local ecology and as such has a biological and environmental importance.

Fishing also gives a great pleasure to sport fishermen around the world and at the same time contributes to the local economy. In spite of this fish receive very little attention in consideration to diseases and health protection especially in regard to feral populations. These populations are often very vulnerable and to promote sustainability it is of outmost importance to take active actions and promote international regulations for restrictions of fisheries. Fish is a difficult group to handle regarding welfare, diseases and health protection. The reasons for this are several but the most important are:

- Fish lives in water – and water is an excellent medium to transfer pathogens from one area to another, which contributes to a fast and uncontrolled transmission of infectious diseases.
- It is often difficult to recognise clinical signs of disease in fish since they lives in water. Fish also lack mimic, voice or recognizable body language, which gives a late discovery of the disease and restrain a fast starting up of the treatment.
- Vertically transferred diseases have been described in all vertebrate classes, which mean that the offspring from infected females are born with the pathogen in their bodies. The fecundity in fish is high compared to other classes of vertebrates, leading to that a single infected female may deliver several thousands of infected of eggs.
- Feral fish lives in the vicinity to farmed fish. Consequently, it is unavoidable that the two groups share infectious agents with each other. Farmed fish are often kept during extreme conditions with high population densities, and a not completely optimized environment.This is a situation which promote options for pathogens to infect farmed fish which thereby can constitute a reservoir for further spreading of diseases.
- Fish are as other animals biological packages, which means that they share habitat with other organisms (Figure. 8.1).
- A feral fish population is in practice impossible to treat for diseases without very significant negative impact on the environment.

Infectious Diseases

Viral Diseases

Infectious Pancreatic Necrosis (IPN)

IPN is one of the most important virus diseases in aquaculture. IPN is caused by a birna virus. The clinical symptoms may vary depending on the species infected and the serotype of the virus. IPN is primarily of great concern in juvenile rainbow trout (*Oncorhynchus mykiss*). Infected individuals show abnormal behaviour and abnormalities such as a swollen abdomen, exopthalmia (protruding eyes) and dark pigmentation of the skin. Pathological findings include inflammation and necrosis of the pancreas and intestines with bleedings (Figure. 8.2).

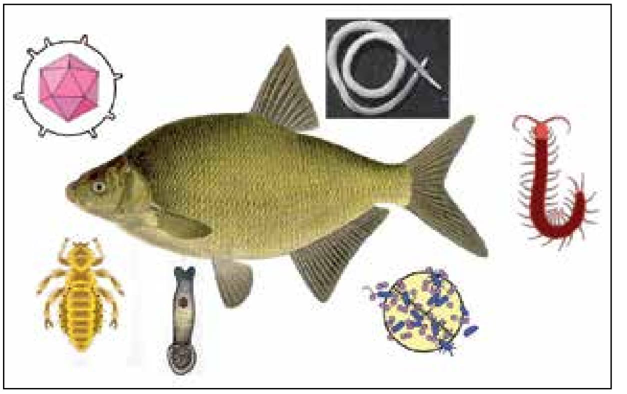


Figure 8.1. A fish is more than one organism. Source: Anders Hellström/SVA.

Viral Hemorrhagic Septicaemia (VHS)

VHS is caused by a rhabdovirus. The virus can be carried by both fresh water and marine fish species. VHS causes problems mainly in rainbow trout aquaculture but also affect feral fish and can cause mass mortality in herring (*Clupea herrangus*). The virus is considered a serious threat to the fish populations in the Great lakes, USA/Canada, due to the high mortalities and that it strikes so many different species. The clinical symptoms of VHS varies considerably. Pathological findings include haemorrhages in muscles (Figure 8.3).

Figure 8.2. IPN infected brown trout with hyperaemia of the liver and pylorus and heamorrhages in pylorus. Photo: Anders Alfjorden/SVA.

Infectious Haematopoietic Necrosis (IHN)

IHN is caused by a rhabdovirus and is a highly infectious disease affecting cultured salmonids Disease usually occurs at water temperatures between 4°C and 18 °C. Young fish up to 1 year are most susceptible to overt infection often with high mortalities. The virus occurs and cause disease in fish populations in several countries within EU. Transmission can be both vertical and horizontal.

Spring Viremia of Carp (SVC)

SVC is caused by a rhabdovirus. It originates from carp in Asia but has been spread (partly by goldfish) to Europe, and constitute a problem for farmers of carp in the central Europe but also for fresh water cyprinids in infected areas.

Figure 8.3. Rainbow trout (*Oncorhynchus mykiss*) with multiple haemorrhages (bleedings) in the dorsal muscles. Viral haemorrhagic septicemia/VHS. Photo: Anders Alfjorden/SVA.

Lymphocystis

Lymphocystis, caused by an iridovirus is a widespread disease revealed in more than 140 marine and freshwater

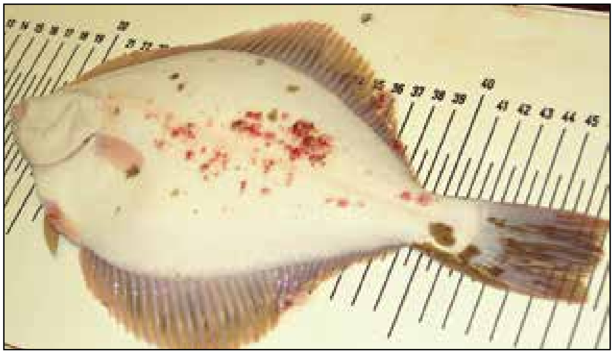


Figure 8.4. Lymphocystis in flounder. Photo: Bengt Ekberg/SVA.

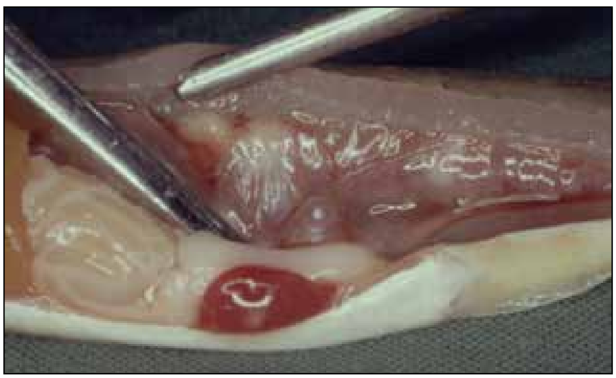


Figure 8.5. Bacterial kidney disease (BKD) in rainbow trout, note the enlarged kidney and spleen with white nodular changes. Photo: Bengt Ekberg/SVA.

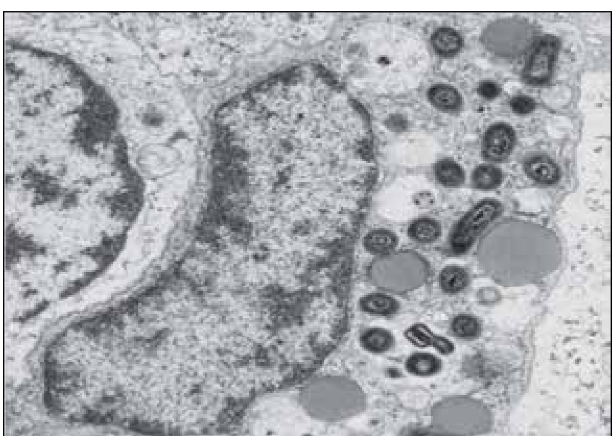


Figure 8.6. Transmission electron micrograph of a macrophage showing the intracellular bacteria *Renibacterium salmoninarum* causing BKD. Photo: Leif Norrgren/SLU.

fish species. The disease is characterized by occurrence of tumour-like clusters of relatively hard nodules up to 2 mm in diameter of white, cream, gray or brownish colour localized mainly on the body surface of affected fish including fins (Figure 8.4). Occasionally, lymphocystis nodules also can occur on gills and internal organs. In marine areas the various flatfish species has been reported as mainly affected by disease. In the Baltic Sea lymphocystis has been revealed in dab (*Limanda limanda*), flounder (*Platicthys flesus*) and plaice (*Pleuronectis platessa*).

Bacterial Diseases

Bacterial Kidney Disease (BKD)

The disease is caused by the intracellular bacterium, *Renibacterium salmoninarum*, a gram-positive, non sporulating, non-variable, coccoid rods. The bacterium is transferable to all salmonids, including whitefish (*Coregonidae*) and has also been demonstrated in non-salmonids such as stickleback (*Gasterosteiformes*). Arctic char (*Salvelinus alpinus*) and salmon (*Salmo salar*) seems to be more sensitive to the disease than other salmonids. Rainbow trout (*Oncorhynchus mykiss*) seems to be more resistent to the disease, often with no visible symptoms. The disease occurs in most European countries as well as in Canada and the USA. The disease is both horizontally and vertically transmissible. The horizontal transmission is probably the most important factor in the eradication program for the disease. As the disease can exist in a very low prevalence in adult fish and be transferred to a few offsprings it can be very difficult to detect the disease in the fish farms. BKD causes granulomatous changes with nodular changes in the kidney but also liver, spleen and heart can be affected (Figures. 8.5 and 8.6).

Furunculosis

The disease is caused by the gram-negative bacteria, *Aeromonas salmonicida* subsp. *Salmonicida* and occurs both in both fresh- and seawater. The disease has been widely distributed geographically and occurs throughout Europe. The bacterium causes disease in all salmonids, but has also been isolated from other species. The rainbow trout is less susceptible to the disease and can therefore act as sub-clinical carriers, which may also be the case with a number of non-salmonid fish species. It is only transmitted horizontally and a vaccine with good effect is available.

Acute outbreaks can be treated with antibiotics. Clinical manifestations depends on age and species affected.

Enteric Redmouth Disease (ERM)

ERM is caused by *Yersinia ruckeri*, a gram-negative bacteria, which exists in several serotypes and can cause disease in several fish species, including non-salmonids. The disease is only horizontally transmitted. The bacterium is a major health problem for rainbow trout in European aquaculture and gives rise to significant economic losses in terms of increased mortality and discarded products. A well-functioning vaccine is available and acute disease can be treated with antibiotics.

Flavobacteriosis

Flavobacterium spp are gram-negative, filamentous, yellow-pigmented bacteria. Some of the most important pathogens causing great economical losses in aquaculture all over the world are *Flavobacterium psychrophilum* and *Flavobacterium columnare.* Establishment of infections caused by *F. columnare* and *F. psychrophilum* are highly dependent on environmental factors such as water temperature, water quality, stocking densities and handling of the fish (Wakabayashi, 1991; Holt et al., 1993).

Flavobacterium psychrophilum causes rainbow trout syndrome (RTFS) and bacterial cold water disease (BCWD). The bacterium is psychrophilic and causes disease at water temperatures below 15°C. Problems are mainly seen in salmonids but probably all freshwater species in cold water are susceptible. The severity and signs of disease depends on the age or size of the fish. Young life stages are the most seriously affected with a systemic disease, sometimes with external lesions. At later life-stages ulcerations of the skin are the most common sign of disease. Ulcerations often occur on the peduncle but other locations of the body i.e. anterior to the dorsal fin, at the anus or on the lower jaw are also common. The bacterium is horizontally transmitted but the definite routes of infection are not yet fully elucidated. *F. psychrophilum* has also been isolated from both male and female sexual products indicating a vertical transmission from brood fish to the offspring (Ekman et al., 1999). However, if a true vertical transmissionactually is present is debated. Fish suffering from *F. psychrophilum* infection are often treated with antibiotics. Bath-treatments with anti-bacterial chemicals are sometimes used to treat mild outbreaks of disease with primarily external lesions. No commercial vaccine is available. (Figure 8.7)

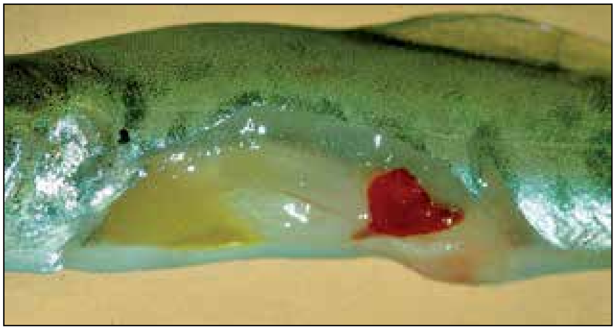


Figure 8.7. Rainbow trout infected with *Flavobacterium psychrophilum* with pale liver and enlarged spleen. Photo: Elisabet Ekman/SLU.

Figure 8.8. Rainbow trout with skin ulcers and necrotic tissue around the dorsal fin infected by *Flavobacterium columnare*. Photo: Anders Alfjorden/SVA.

Flavobacterium columnare causes columnaris disease characterized by skin lesions, fin erosions and gill necrosis (Figure 8.8). Both wild and farmed freshwater fish are susceptible to the bacterium. Columnaris disease is a problem in salmonid aquaculture worldwide and also causes great economics losses in the channel catfish industry in the US (Pulkkinen et al., 2010). Disease usually occurs at high water temperature, above 20°C. The bacterium is horizontally transmitted. Diseased fish can be treated with antibiotics or bath-treatments with anti-bacterial chemicals. A commercial modified live columnaris immersion vaccine is licensed for use in catfish in the US.

Vibrios

These gram-negative bacteria are curved rods motile by flagellae. Several species of Vibrio can cause disease in fish both in marine and brackish waters. *Vibrio anguillarum*, today renamed to *Listonella anguillarum* is the cause of vibriosis affecting both farmed and feral species including eel (*Anguilla anguilla*), salmonids, turbot (*Scophthalmus maximus*) and cod (*Gadus morhua*). Disease occurs especially when the fish are stressed due to high water temperatures, poor water quality or high stocking densities etc. *Alivibrio salmonicida (*formerly *Vibrio salmonicida)* is the aetiologic agent of cold water vibriosis (Hitra disease) and causes problems mainly in farmed Atlantic salmon in Norway. Vaccines are available to both vibriosis and cold water vibriosis.

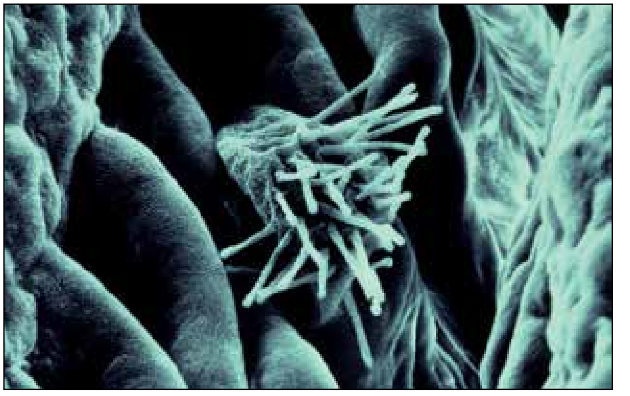


Figure 8.9. Scanning electron micrograph of Trichophyra on gill of a salmon. Photo: Leif Norrgren/SLU.

Parasites

Parasites are important components of any ecosystem, which not only play key roles in fish population dynamics and community structure, but also can provide important information on environmental stress, food web structure and function and biodiversity. Numerous studies demonstrated effects of anthropogenic-induced environmental perturbations on parasitic organisms at both the population and community levels. In general, responses of hosts and communities depend very much on the type and intensity of the stressor, the parasite life cycle and duration of exposure.

Examples of common protozoan parasites are *Trichophyra* spp. (Figure 8.9), *Ichtyobodo* spp. (Figure. 8.10) and *Trichodina* spp. (Figure 8.11). These ektoparatsites are often found in large number on gills and skin and may cause high mortality.

Anisakis simplex (Nematoda) is a zoonotic parasite which can infect humans. *A. simplex* infect a large number of fish species, i.e. Baltic herring and cod. The mean annual prevalence trend for infection of Baltic herring with larvae of *A. simplex* in the Russian EEZ of the Baltic Sea (26 ICES sub-division) are decreasing between 1997-2008 (Figure. 8.12)

Anquillicola spp (Nematoda) is found in the swimbladder of eel. The parasite has spread from eel farms in southern Europe to the Baltic sea where is first was found 1987.

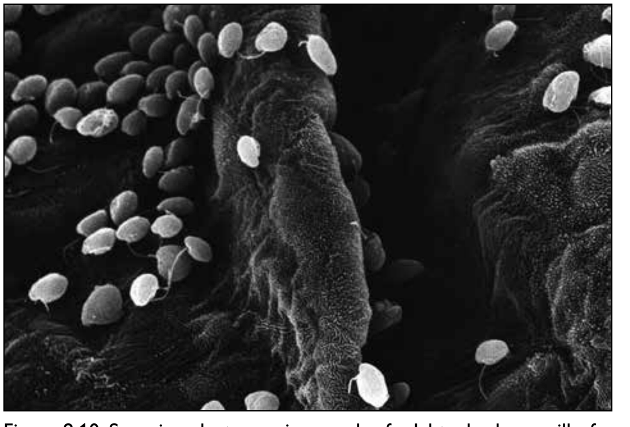


Figure 8.10. Scanning electron micrograph of a Ichtyobodo on gill of a salmon. Photo: Leif Norrgren/SLU.

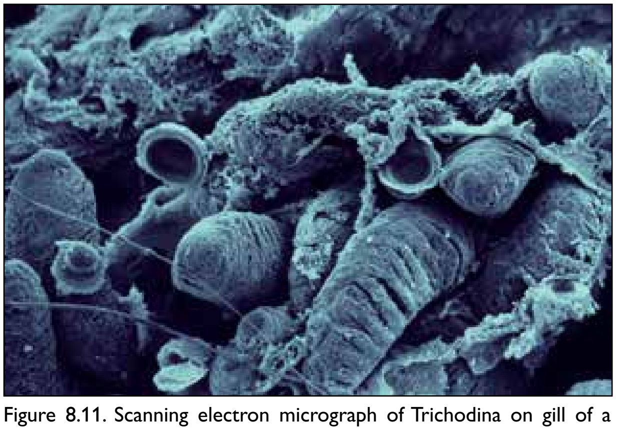


Figure 8.11. Scanning electron micrograph of Trichodina on gill of a salmon. Photo: Leif Norrgren/SLU.

Cryptocotyle spp. (Trematoda) infect species such as in dab, flounder and cod.

Lepeophtheirus pectoralis (Crustacea)– in dab and *Lernaeocera branchialis* (Crustacea)- in cod (Lang and Rodjuk, 2006).

In connection with aquaculture the role of parasites is fairly known, but when it comes to wild fish and new species for farming the knowledge is limited. It is known that ornamental fish, now frequently imported from Asia and EU for garden ponds, can carry exotic parasites, for example *Atractylotocestus huronensis*, *Bothriocephalus acheilognathi* and *Monobothrium wageneri*, which can cause disease problems in native species. Some parasites will be favoured due to the climatological change – for example the agent that causes whirling disease in salmonids *Myxobolus cerebralis*. Also a myxozoan parasite of salmonid fishes, *Tetracapsuloides bryosalmonae*, which causes proliferative kidney disease (PKD), one of the most serious parasitic diseases of salmonid populations in Europe and North America. *T. bryosalmonae* is favoured by high water temperature.

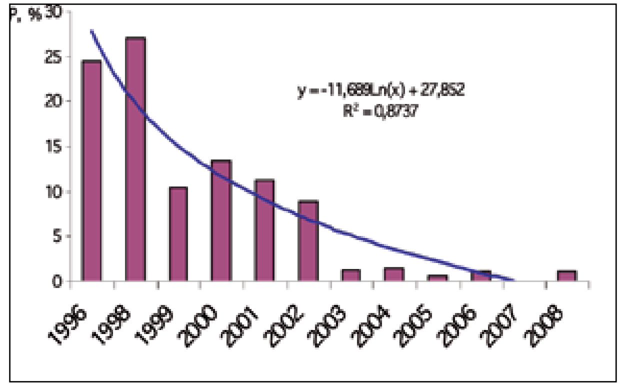


Figure 8.12. Long-term trend of herring infestation with *Anisakis simplex* l. in the Russian EEZ of the Baltic Sea (26 ICES sub-division) (Rodjuk 2007 with addition).

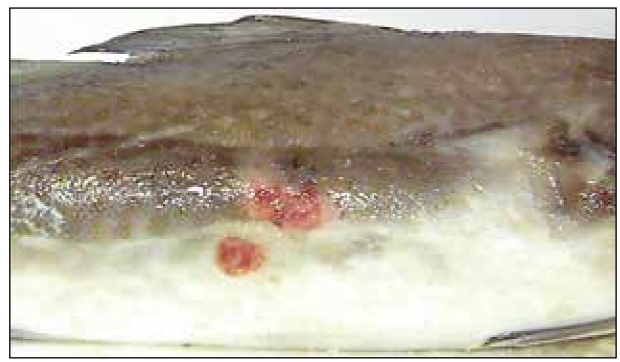


Figure 8.13. Skin ulcers in cod. Photo: Bengt Ekberg/SVA.

Miscellaneous Diseases

Skin Ulcer Disease

Skin ulceration is commonly seen in numerous marine and freshwater wild fishes as well as in farmed fish. The skin ulcers can be from small to large and of irregular shape, and usually appear as shallow lesions. They are often spherical with red necrotic centers surrounded by a thickened rim of epidermis and white peripheries. Sometimes lesions can penetrate deeply into the muscle and reach several cm in diameter (Bucke et al., 1983, Wiklund, 1994). The etiology of skin ulcer disease has not been thoroughly examined. In scientific literature the ulcer lesions are connected with various infectious agents (viruses, bacteria and parasites), dinoflagellate toxins, immunological, nutritional and metabolic perturbations in fish (Sinderman, 1979, 1996). This disease is associated with water salinity, temperature variations, skin injuries and pollution (Wiklund, 1994). Therefore the etiology of ulcer disease is suggested to be complex and multifactorial (Vethaak, 1992). In the Baltic Sea, skin ulcers (Figure 8.13) have been described in herring, sprat, cod, eelpout, four-bearded rockling, smelt, turbot, flounder and plaice.

Fin Rot Erosion

Fin rot is a disease, which is mostly observed in aquaculture, but can also occur in natural populations. This disease is characterized by a progressive necrosis of the anal and dorsal fins, and less frequently, caudal fins. The first signs of the disease are milky white areas appearing on the fish fins, particularly around the edges. The fins begin to fray and get ragged, becoming shorter in some time. Usually the edges look white, and may give rise to a fuzzy growth due to the secondary infection. Slowly the lesion becomes red and inflamed, bloody patches appear and fin is eaten away. Fin rot starts at the edge of the fins,

Figure 8.14. Cod (*Gadus morhua*) with a congenital defect involving the bones of the skull. Opening of the skull revealed a severe hydrocephaly. Photo: Bengt Ekberg/SVA.

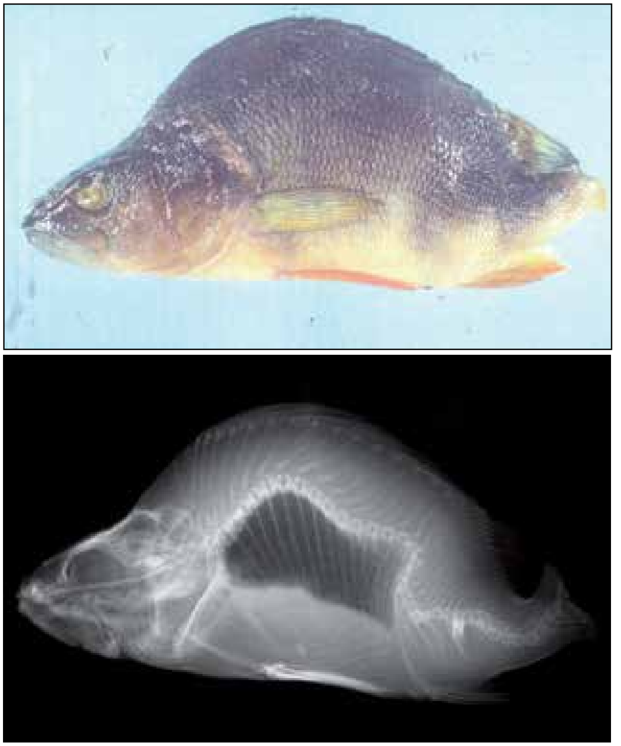


Figure 8.15. Perch severely disformed (above) and X-ray image of a perch (*Perca fluviatilis*) with extensive spinal defect. The cause of the injury has not been established, but one possible cause can be an early environmental impact. Photo: Bengt Ekberg/SVA.

and destroys more and more tissue until it reaches the fin base. If it reaches the fin base, the fish will never be able to regenerate the lost tissue. At this stage, the disease may attack the fish body. Secondary infections are common at the advanced stage of fin rot, bringing new symptoms to the afflicted fish such as white cotton wool-like tufts or streaked red patches. Fin rot can be the result of a bacterial or fungal infection. Sometimes, both types of infection are seen together. Fin rot disease is commonly associated with bad water conditions, injuries, poor diet, or as a secondary infection in a fish which is already stressed by other disease. In the Baltic Sea, fin rot erosions have been described in many species such as dab, flounder and cod.

Skeletal Deformities

Skeletal deformities have been found in a numerous fish species. The following types of deformities occasionally occur in fish populations: “pug-head” (Figure 8.14), scoliosis, lordosis, kyphosis, (Figure 8.15), dwarfism, deformations of jaw and gill-cover shortening etc. The background to these types of malformations is often unknown but potential causes are for example pollutants, nutritional. In general, skeletal deformities constitute a “grey area” of understanding and in those cases a genetic aetiology is often proposed. In the Baltic Sea the skeletal deformities have been reported in many species such as herring, pike, perch, sprat, fourhorn sculpin, cod and flounder.

Skin Tumours

Skin tumours of fish are easily recognizable lesions, some of which have been known for centuries (Figures 8.16 and 8.17) Tumours are widely reported in different fish species including the Baltic ones. The term “a tumour” is defined as an abnormal mass of tissue which can grow independently and uncoordinated from normal tissue. A benign tumor do not spread to other organs or infiltrate adjacent structures and is often within its own capsule. A malign tumor has the ability to spread to different organs and infiltrates surrounding tissues. Skin tumours are classified on the histological basis according to the origin of the proliferating cells and their degree of malignancy. Among the naturally occurring fish tumours, benign epidermal hyperplasias and papillomas are the most frequently observed. The causes of fish tumours formation vary. In more than half of all causes examined by electron microscopy and virological methods, virus or virus-like particles were found in tumour tissues. So far, oncogenicity has been clearly demonstrated only for herpes viruses isolated from benign tumours. Several studies suggest that toxic chemicals present in sediments and tumours observed in certain fish species may be related (Kinne, 1984; Möller and Anders, 1986; Maccubbin and Ersing, 1991; Anders and Yoshimizu, 1994). Nowadays the aetiology of fish tumours is proposed to be multifactorial. Skin tumors have been reported in dub, eel and smelt in the Baltic Sea (Lang, 2002).

Figure 8.16. Skin tumour in a pike. Photo: Bengt Ekberg/SVA.

Liver Tumors

The hepatic tumours are recognized to be in close connection to marine pollution. Therefore, they are common in bottom feeders, including 13 flounder species. An increased prevalence of liver tumours and preneoplastic lesions in marine flatfish species is considered as an indicator of biological effects of carcinogenic environmental contaminants (e.g. chlorinated biphenyls, polycyclic aromatic hydrocarbons). Nodular lesions are situated on the surface of liver and look lighter or darker in colour than the surrounding liver tissue with well-defined margins. The benign hepatomas occur more frequently in fish. Hepatic tumours in Baltic Sea have been reported in flounder and dab.

Figure 8.17. Pike (*Esox lucius*) with a tumour in the abdomen. Photo: Bengt Ekberg/SVA.

Kidney

Nephrocalcinos is a condition when high levels of carbon dioxide results in deposition of calcium carbonate within the kidney tubules. Nephrocalcinosis which results in extensive kidney damage is irreversible (Figure 8.18)

Reproductive Disorders

Disruption of the endocrine system has been shown to occur in wild fish populations across the globe and affect fish during the last 25 years (Jobling and Tyler, 2003). Effects range from subtle changes in the physiology and sexual behaviour of fish to permanently altered sexual differentiation, impairment of gonad development and altered fertility. The intersex condition is defined as simultaneous presence of female gametes within the testis tissue which contained predominantly normal male gametes at late stages of spermatogenesis. Specimens with

Figure 8.18. Renal calcification (nefrocalcinos) in the rear part of the kidney in rainbow trout (*Oncorhynchus mykiss*).The disorderis caused by an excess of carbon dioxide in the water. Photo: Bengt Ekberg/SVA.

a severe form of intersexuality exhibited large strands of oocytes in the testicular lobules (Matthiessen, 2003). These reproductive disorders may be induced by wide variety of adverse environmental conditions, including sub-optimal temperatures, restricted food supply, low pH, environmental pollutants, and parasites. Furthermore, it is possible that all these factors can act simultaneously. The gonad alterations like intersex and atresia seem to be suitable biomarkers to detect the effect of reprotoxic stressors in the water environment. In the Baltic Sea reproductive disorders are found in eelpout, stickleback and perch (Gercken and Sordyl, 2002).

Figure 8.19. Intersex gonads from a turbo. Photo: Leif Norrgren/SLU.

Intersex

Fish living in the vicinity of for instance sewage treatment works and industrial point sources may be exposed to substances resembling endogenous hormones, i.e., Endocrine Disrupting Chemicals (EDCs), such as alkyl phenols, phthalates and synthetic estrogens. In the 1980s anglers in the UK reported intersex roach in sewage effluent lagoons. Elevated incidence of intersex in roach and many other fish species has been shown to be widespread all over Europe. Several field studies throughout the world have confirmed that fish exposed to EDCs are displaying intersex (Figures 8.19, 8.20), altered spermatogenesis and decreased reproduction success. Even though field observations have contributed to disclose the risk of exposure for instance in the vicinity of industries and sewage treatment works, there are still very large gaps of information regarding the effects of EDCs on reproductive fitness and population dynamics.

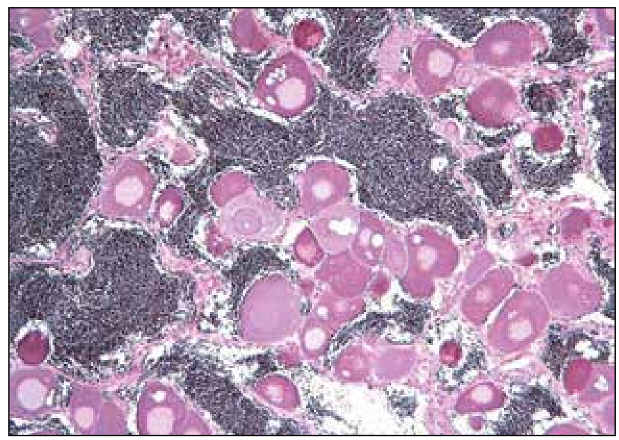


Figure 8.20. Light microscopic picture of intersex in a gonad from roach. The pink cells are immature eggcells and the dark blue areas spermatozoa. Photo: Leif Norrgren/SLU.

The M74 Syndrome

The most serious threat towards the Baltic salmon has been abnormal yolk-sac fry mortality. This diseases is designated M74 and was first observed in the early 1970s. The incidence of M74 have varied and the situation was extremely serious during the 1990s when extremely high mortality was recorded (Figure. 8.21). A number of different explanations to M74 have been considered, i.e., pollutants including persistent organic compounds, pathogens, abiotic conditions such as water temperature, changed feed, nutritional compounds such as astaxantin and other vitamins. During the middle 1990s, US scientist reported that a syndrome affecting salmonids in the Great

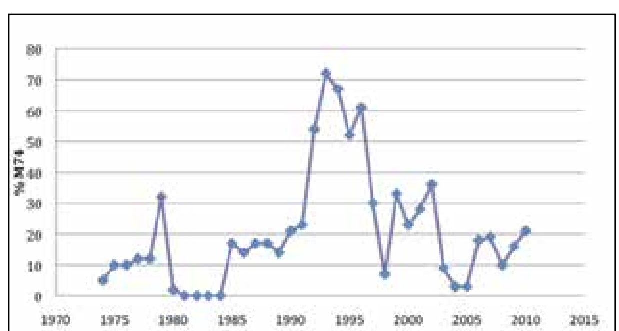


Figure 8.21. The incidence of M74 in Swedish Baltic salmon rearing stations between 1974-2010.

lakes with similar pathogenesis as M74 was caused by thiamine deficiency. This information initiated research to evaluate the role of thiamine in M74, and it was shown that thiamine deficiency was involved in both EMS and M74. The background to low thiamine levels in salmon suffering from M74 is still unknown.

Conclusions

Fish in the Baltic Sea are affected by a number of health problems including infectious diseases such as Flavobacteriosis, Vibriosis, Furunculosis, IPN, PKD and BKD. A number of species are affected by reproduction disorder and the most significant, M74, has been a serious threat towards Baltic salmon populations.

Contaminants and Health of Aquatic Wildlife

9

Anders Bignert, Britt-Marie Bäcklin,
Björn Helander and Anna Roos
Swedish Museum of Natural History, Stockholm, Sweden

Introduction

There are thousands of chemicals, both naturally occurring and antropogenic which are present in the aquatic environment. Some of the more recent introductions of chemical have disrupted the natural accommodation and caused ecosystem damage (polluted air and water resulting in retarded bird reproduction, fish kills and other disruptions of wildlife), eutrophication and hypoxia from excess nutrients, human respiratory ailments, systemic poisoning, cancer and many other impacts to life on earth. Most of the more recent contaminants result from activities that provide useful products and processes for humans and society. The benefits of automobiles, electrical equipment, functional metals and polymers, fuels, fluids, pest control, coatings, adhesives and economical food production come with some hazards. We recognize some of the hazards (toxicity during manufacture and use, fertilizer run-off to waters, pesticide food residues, air and water emissions, mechanical hazards, potential explosions, accidental releases), which are mitigated through design and various regulations for delivery and use. However, later we find unanticipated contamination of air, water, avian and aquatic life, soil from wastes, long-range transport from distant sources, the long term presence of persistent contaminants, unknown health effects of continuous exposure to multiple chemical contaminants, and loss of beneficial uses due to ecosystem damage.

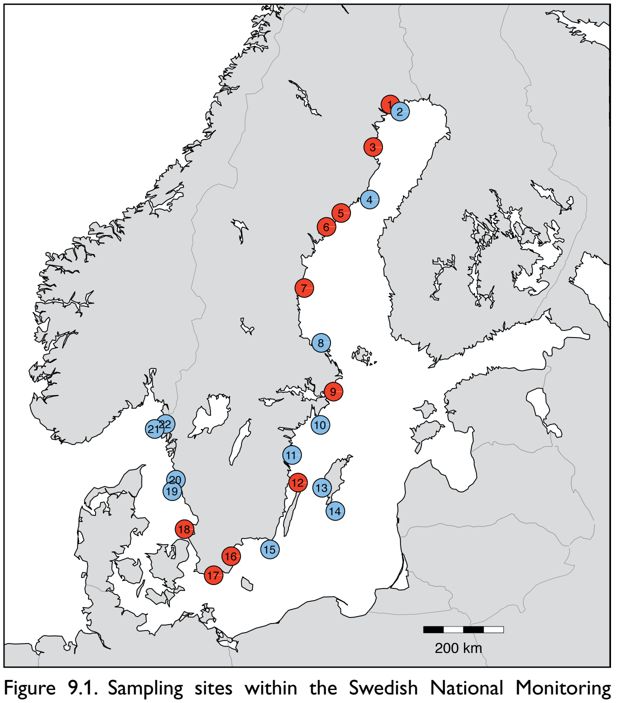


Figure 9.1. Sampling sites within the Swedish National Monitoring Programme in Marine Biota. Rånefjärden, 2) Harufjärden, 3) Kinnbäcksfjärden, 4) Holmöarna, 5) Örefjärden, 6) Gaviksfjärden, 7) Långvindsfjärden, 8) Ängskärsklubb, 9) Lagnö, 10) Landsort, 11) Kvädöfjärden, 12) Byxelkrok, 13) St.Karlsö, 14) SE Gotland, 15) Utlängan, 16) V. Hanöbukten, 17) Abbekås, 18) Kullen, 19) Fladen, 20) Nidingen, 21) Väderöarna, 22) Fjällbacka. Blue dots indicate stations where sampling has been carried out 30 years or longer, red dot are newly established stations with only a few years of sampling.

In the early 1960s, poor breeding success and declining populations in the white-tailed sea eagle was observed on the Swedish coast. Similarly, reproduction disorders were recorded in sea mammals such as seals and otter. These observations initiated the start of monitoring of contaminants and health in animals, thriving in the aquatic ecosystem, at different sites along the Swedish coast (Figure 9.1). Environmental quality criteria were defined, which are used to estimate the risks for animals based on the concentrations in different tissues (Figure 9.2).

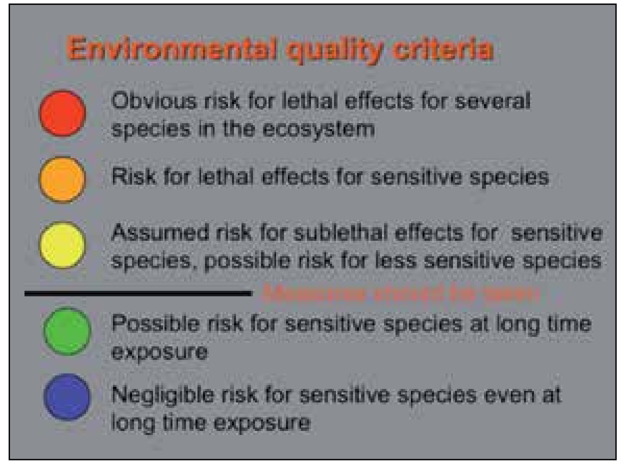


Figure 9.2. Environmental quality criteria used to estimate the risks for animals based on the concentrations in organs. In practice, a lot of work remains to relate actually measured concentrations in various matrices to these levels. The suggested colors in the figures below are only tentative.

Fish

Concern has been paid regarding elevated concentrations of heavy metals in biological samples, in particular mercury, lead and cadmium. After the removal of lead in gasoline and other restrictions, the lead concentration has decreased significantly in all monitoring time series of sufficient length (Figure 9.3, Lind et al., 2006). Despite the efforts made to reduce discharges of cadmium, the concentrations of cadmium measured in fish liver have

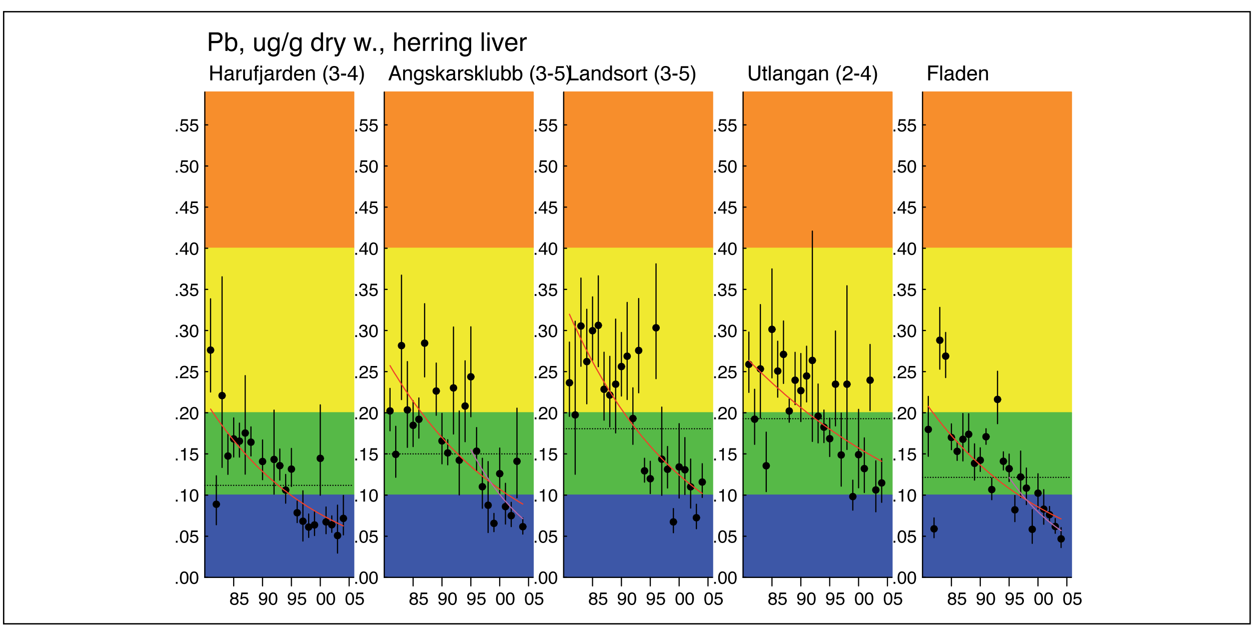


Figure 9.3. Lead concentration (ug/g d.w.) in herring liver along the Swedish coast (station 2, 8, 10, 15, 19, see map in Figure 9.1).

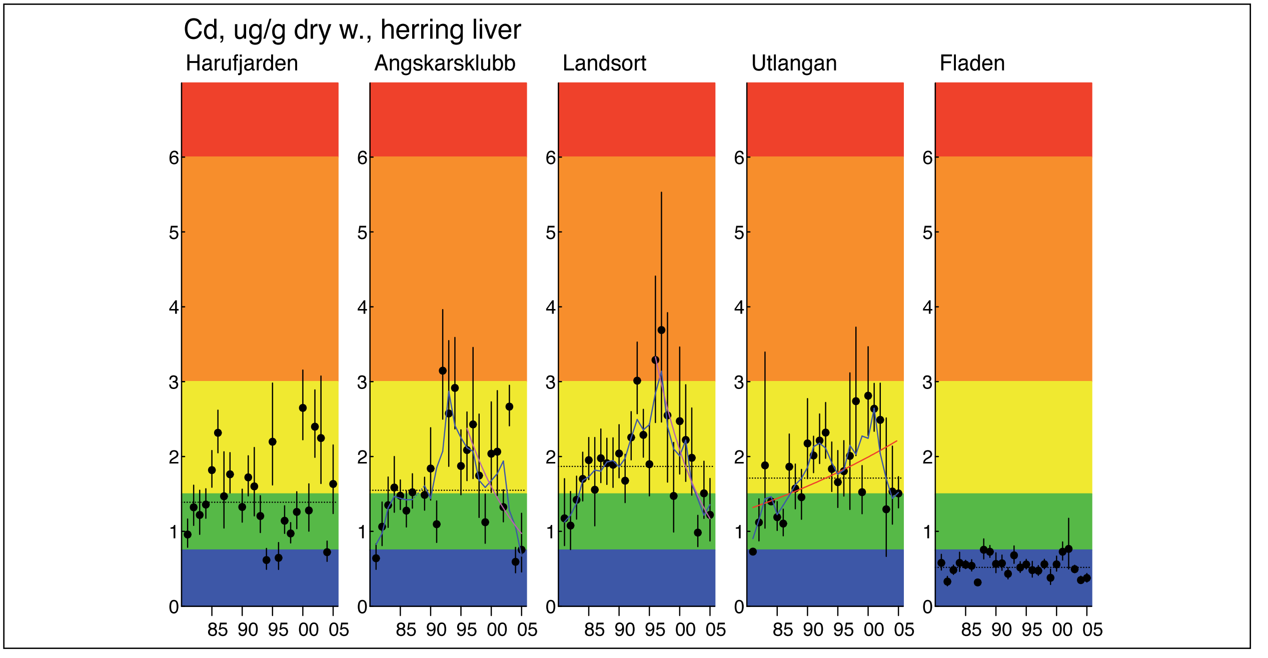


Figure 9.4. Cadmium concentration (ug/g d.w.) in herring liver along the Swedish coast (station 2, 8, 10, 15, 19, see map in Figure 9.1).

not showed the same encouraging development and the concentrations are fairly close to the levels where we may expect effects (Figure 9.4). The longest monitoring series of mercury starting in the beginning of the 70s show significant decreases of 20-40%, whereas some time-series starting in the 80s show even increasing trends. In general the concentrations in fish from the Baltic are low, below 50 ng/g wet weight (Bignert et al., 2010) in contrast to the high concentrations found in freshwater fish from many lakes at the Scandinavian Peninsula.

Within the framework of HELCOM, the council of ministers agreed on a reduction of discharges of 50% within 10 years with 1987 as the start year for several of the legacy contaminants like DDT and PCB and heavy metals like mercury, lead and cadmium. With monitoring activities focused on temporal trend assessment it was possible to show the results also in biological samples from the environment (Bignert et al., 1997). It is essential to see how various measures to protect the environment work in practice. In some cases the reduction of contaminant burden in the ecosystem was faster than one would expect considering their persistence to degradation. This is especially true for pesticides like DDT and Lindane where the bans in Sweden and Western Europe implied decreasing trends in fish that were estimated to between 10 to 20% a year. For industrial contaminants like PCB included in a number of products, the decrease was slower, about 5 to 10% a year (Figure 9.6).

Birds

Observations of poor breeding success and declining populations in the early 1960s initiated the start of monitoring of the white-tailed sea eagle on the Swedish Baltic coast in 1964. A retrospective study shows a significant drop already in the early 1950ies in the number of sea eagle chicks per successful breeding (Figure 9.7). Productivity decreased to a bottom level during the 1970s, with concentrations in the eggs of DDTs averaging 825 and PCBs 1,100 ug/g [ppm] lipid weight (corresponding to 34 and 46 ppm on a wet weight basis, respectively) (Helander et al. 1982). Following the bans of DDT and

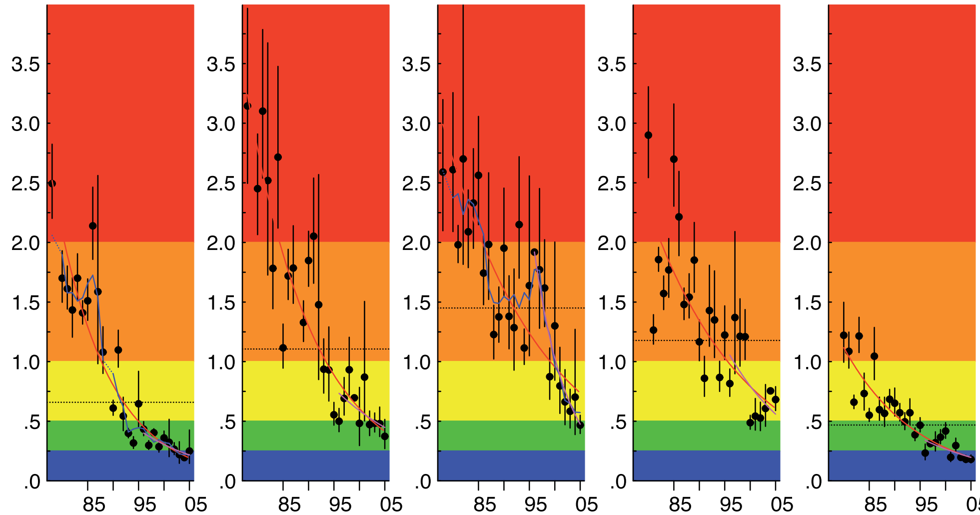


Figure 9.5. Changes of sPCB concentration (ug/g l.w.) in herring muscle along the Swedish coast (station 2, 8, 10, 15, 19, see map in Figure 9.1).

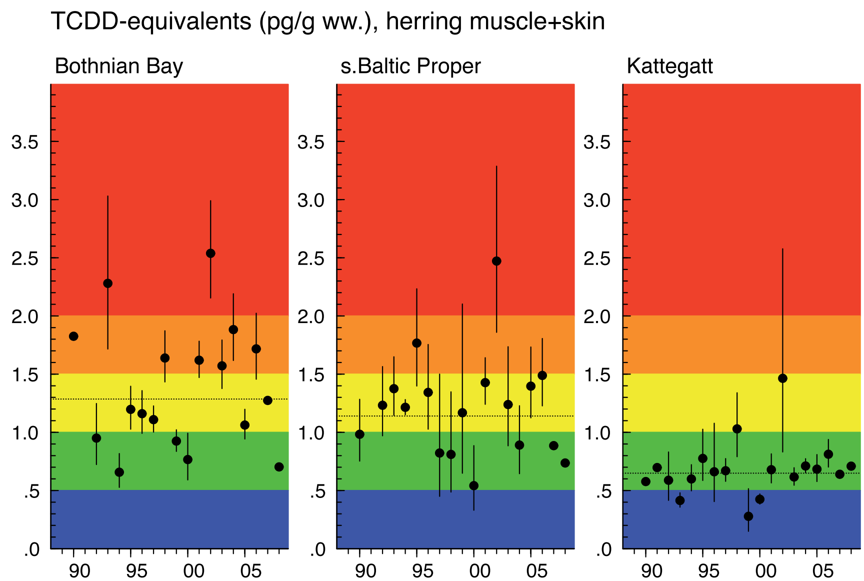


Figure 9.6. Dioxin concentration (TCDD-equivalents, pg/g wet weight) in herring liver along the Swedish coast (station 2, 8, 10, 15, 19, see map in Figure 9.1).

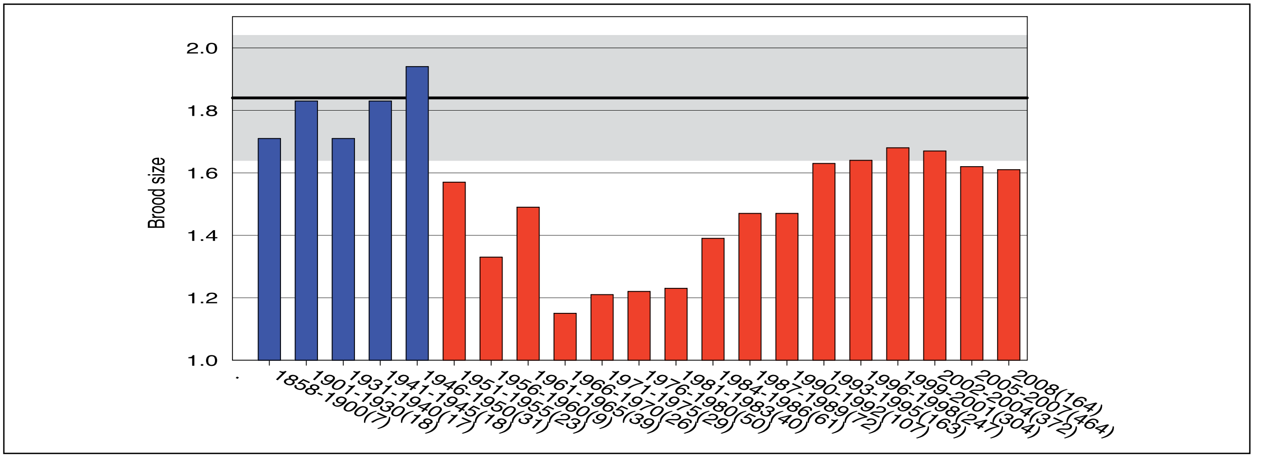


Figure 9.7. Mean nestling brood size in productive white-tailed sea eagle nests on the Baltic coast, 1858-2008. The grey zone indicates the 95 % confidence limits for mean brood size before 1951 (Helander, 2003). Brood sample size given in brackets.

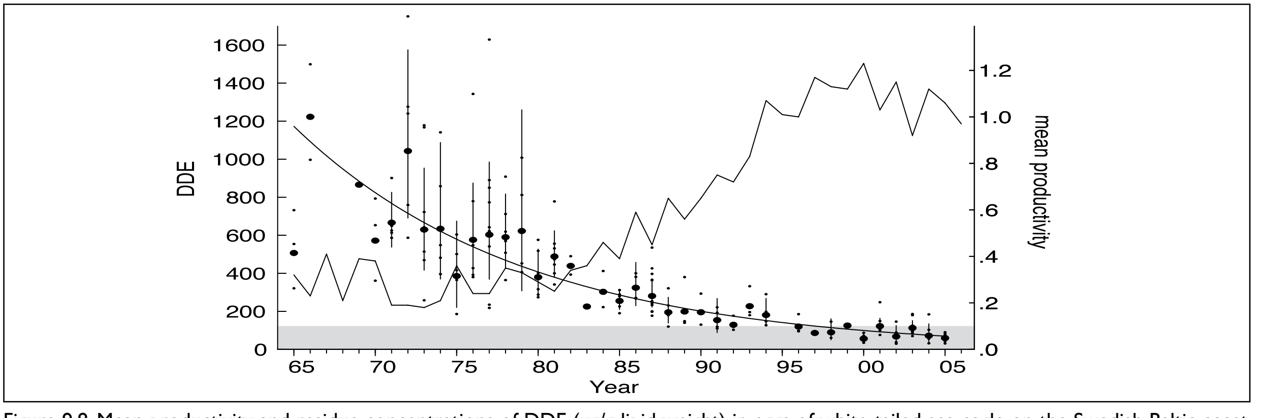


Figure 9.8. Mean productivity and residue concentrations of DDE (ug/g lipid weight) in eggs of white-tailed sea eagle on the Swedish Baltic coast, 1965-2005. The grey zone indicates a concentration interval below an estimated threshold level for effects from DDE on reproduction in this species (Helander et al., 2002).

PCB in the 1970s, concentrations in eagle eggs decreased on average 8.6 % per year for DDE and 5.5 % per year for PCB, 1977-1997. Although concentrations in the eggs decreased, productivity in this population remained unaltered into the 1980s and began to improve when the concentrations of DDE and PCB in the eggs averaged below 300 and 800 ppm, respectively. Productivity versus DDE in eagle eggs is outlined in Figure 9.8. The increase in productivity leveled off by the late 1990s near a reference level calculated from data up to the early 1950s (Helander 1985, 1994).

An interesting observation is that old females did not improve their reproduction although concentrations of DDE and PCBs decreased in their eggs, indicating persisting effects from previous exposure to much higher concentrations (Helander et al., 2002). During the 1960s to 1980s, a common feature was that eagle eggs from the Baltic coast were heavily dessiccated, a result from alterations in the structure of the eggshell. This was a feature appearing in parallel with eggshell thinning, but was not correlated with thinning (Helander et al., 2002). Productivity was significantly correlated with dessication but not so with eggshell thinning. Along with the replacement of old females over time, the occurrence of dessiccated eggs died away by 1990, and also eggshell thickness has increased back to normal. In one region on the Swedish Baltic coast, though, the occurrence of dead eggs is significantly higher and the number of young per productive nest is significantly lower than in neighbouring coastal regions (1994-2009). There is no difference in DDE or PCB concentrations in eagle eggs from these regions, and no difference in concentrations of flame retardants (Nordlöf et al., 2010).

During the 70ies bans were introduced stepwise for both DDT and PCB in Sweden and similar measures were taken also in the other countries around the Baltic. These measures to stop discharges had a significant positive effect and the monitoring programs could detect decreasing trends of contaminant concentrations in fish and guillemot eggs (Figure 9.10) and the eggs became thicker and are today almost as thick as those recorded in museum collections from times before the production of DDT started (Figure 9.10, Bignert et al., 1995). Perfluorinated substances have been seen increasing in biota worldwide, including remote areas as the Arctic. Temporal concen-

Fig 9.9. Nestling and a dead egg in a white-tailed sea eagle nest on the Swedish Baltic coast. Photo: Björn Helander/NRM.

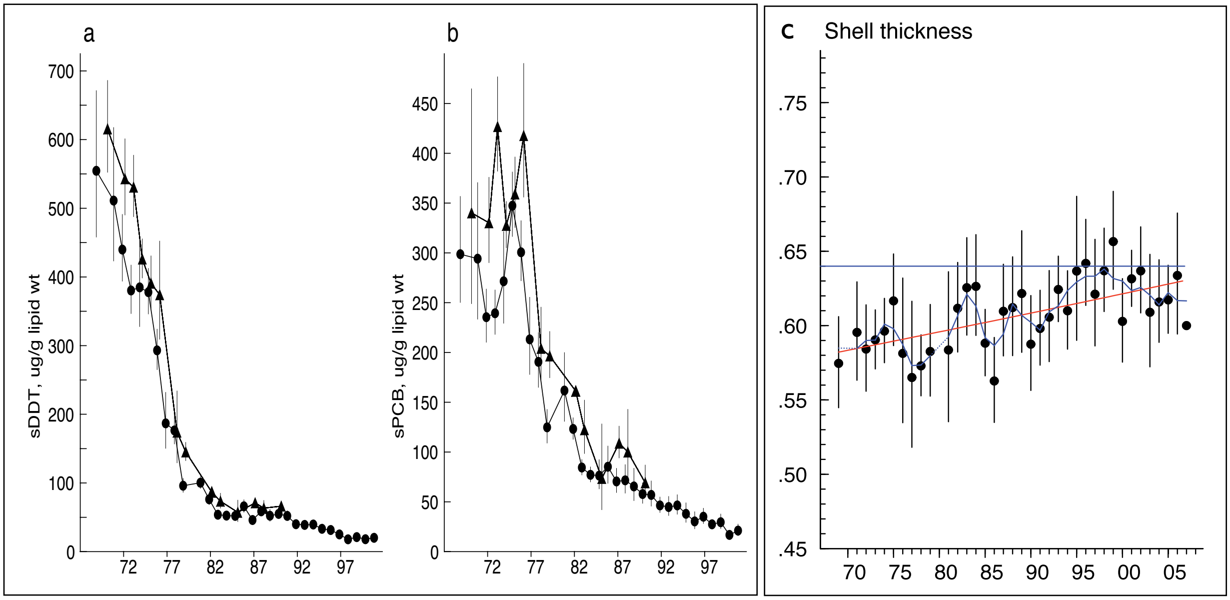


Figure 9.10. Concentrations of DDTs (a) and PCBs (b) decreased rapidly in guillemot eggs after the ban and the shell thickness (c) increased significantly during the same period.

Figure 9.11. White-tailed sea eagle embryo that died as a result of high concentrations of organochlorine substances in the egg. Photo: Björn Helander/NRM.

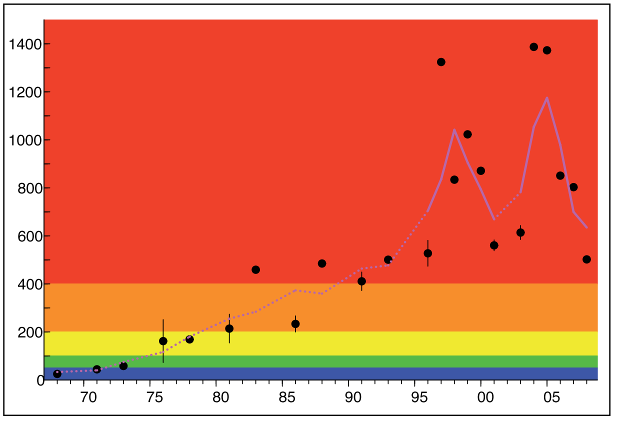


Figure 9.12 Temporal concentrations of PFOS (ng/g wet weight) in guillemot eggs.

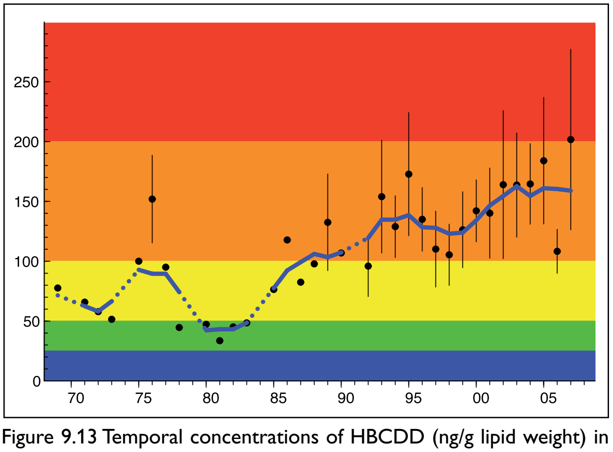


Figure 9.13 Temporal concentrations of HBCDD (ng/g lipid weight) in guillemot eggs.

trations of PFOS in guillemot eggs are analyzed for the presence of perfluorooctanesulfone (PFOS) and similar perfluorinated alkylated substances (Figure 9.12), with an average annual increase of 9% in guillemot eggs. An other substance which show a similar time trend as PFOS is the Hexabromocyclododecan (Figure 9.13), with an average annual increase of 3% in guillemot eggs.

Other Health Problems in Wild Birds

Since the early 1980s a syndrome causing occasional high mortality has been sporadically observed in wild birds living in the Baltic Sea region. The disease is characterized by a variety of symptom including difficulty in keeping the wings folded along the side of the body, inability to fly, inability to walk, tremor and seizures (Figure 9.14, Balk et al., 2009). The progression of the disease from early clinical signs to death varies between species and for instance in herring gull (*Larus argentatus*) this period is 10-20 days. Thiamine deficiency has been proposed to play a significant role in the ethiology of the disease (Balk et al., 2009). Another factor that has been considered is botulism (*Clostridium botulinum*). Poisoning by botulism is characterised by neurological symptoms resembling the ones described in connection with thiamine deficiency.

Figure 9.14. Photo: Lennart Balk/SU.

Aquatic Mammals

Otter

The otter, (*Lutra lutra*) is a medium sized mammal feeding mainly from the aquatic food web, with different fish species as its main food items (approx 80% of its diet) but it also eats frog, crayfish and some birds and small mammals. It decreased dramatically in numbers and range in Sweden during the 1950s-1980s, despite the fact that hunting was banned in 1968. In the 1980s, otters were only found in small scattered areas in Sweden and they were absent from the Baltic coast (Olsson and Sandegren, 1983; Olsson et al., 1984, 1988). A similar decrease was noted in many other European countries.

Figure 9.15. An adult female otter (*Lutra lutra*) from south central Sweden. Photo: Kenneth Johansson.

Many reasons for the decrease of the European otter populations have been discussed, but one in particular was the role of environmental contaminants. It was polychlorinated biphenyls (PCB), DDT, dieldrin and mercury (Hg) that most often were associated with the decline. There is no consensus on which toxin caused the decrease in otter but most studies point out PCB as the major threat to otters (Sandegren et al., 1980; Olsson and Sandegren, 1991; Mason and McDonald, 1986). This conclusion was based partly on result from experimental studies on another mustelid, the American mink (*Mustela vison*), which is very sensitive to PCB (Aulerich and Ringer, 1977; Jensen et al., 1977). In a long term study on the mink given environmental relevant concentrations of PCB, reproductive impairment was seen already at 12 mg/kg PCB l. w. in mink muscle tissue (Brunström et al., 2001), indicating a higher sensitivity than earlier studies have claimed. Wild Swedish otters had higher or much higher concentrations of PCB than the minks in the experiments and it is believed that it is PCB that caused the general population decrease. Another reason for pointing out PCB as the underlying cause was the fact that otters decreased or disappeared mostly from areas with high contaminant load.

PCB was banned in Sweden in the middle of the 1970s and since then the concentrations in Swedish biota, including otters, have decreased. Otters from Sweden have been analyzed for PCB and a body burden of up to 970 mg/kg PCB lipid weight in muscle tissue have been found (Figure 9.16. Roos et al*.,* 2001).

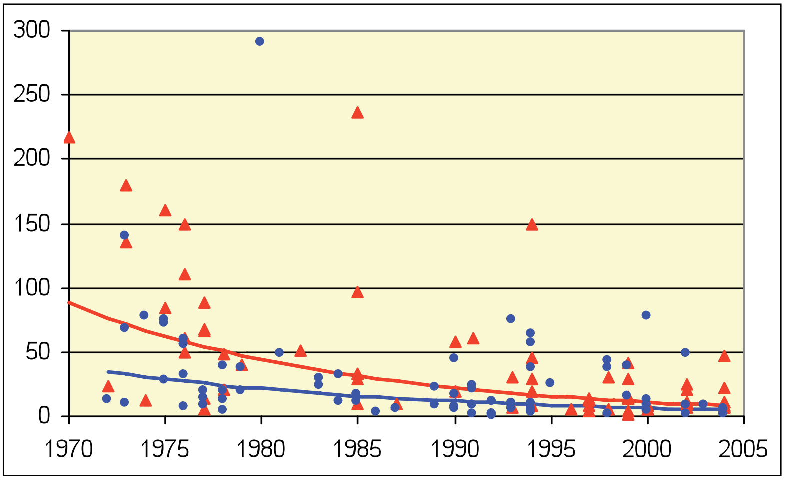


Figure 9.16. Concentrations of PCB in otter muscle tissue (mg/kg l.w.).The otters included in this diagram were either killed in traffic accidents or drowned in fishing gear. Red indicates otters from southern Sweden, and blue those from northern Sweden.

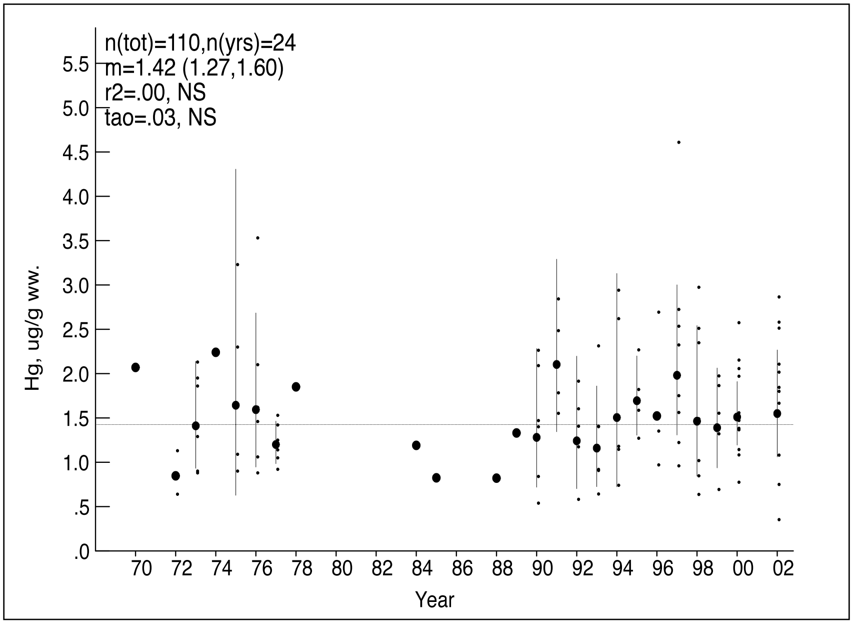


Figure 9.17 Mercury in otter muscle tissue (ug/g wet weight).

The proportion of analysed individuals with a PCB conc. under 12 ppm has increased over time: In the 1970s it was 14%, in the 1980s it was 23%, in 1990s 50% and in 2000s the proportion has increased to 68% (Roos unpublished data). This indicates that PCB still, more than thirty years after the ban, is a problem for otters in Sweden, but the situation has definitely improved.

Also mercury has been pointed out to be a serious threat to the otter. However, in a time trend study of mercury in muscle tissue from Swedish otters the authors concluded that the concentrations in otters are quite low, and has not changed in otter tissue from 1970-2002 indicating that mercury was not the major threat to the otters in Sweden (Figure 9.17. Idman and Roos, 2004).

But of course mercury is a threat to the otters in some areas, also in Sweden. It has been demonstrated that river otters in laboratory experiments develop signs of intoxication when mean concentrations reach 33 (muscle) and 20 µg/g (liver) (Gutleb et al., 1998).

Perfluorinated substances have been seen increasing in biota worldwide, including remote areas as the Arctic. Liver from fiftyfour Swedish otters were analysed for the presence of perfluorooctanesulfone (PFOS) and similar perfluorinated alkylated substances, PFAS (Roos et al., 2007). Two substudies were performed, one geographical study (37 otters collected between 2000 and 2006) and one time trend study (39 otters from only southern Sweden collected between 1972 and 2006). PFOS was by far the most dominant compound of all analysed. It was found in high concentrations in all samples, ranging from 300-8,300 ng/g w.w. The concentrations were up to 10 times higher than in grey seals (*Halichoerus grypus*) and ringed seals (*Phoca hispida*) from the Baltic (Kallenborn et al., 2004). The otters had similar or much higher concentrations of PFOS than river otters (*Lontra canadensis*) in North America (25-994 ng/g liver w.w.) and American mink (*Mustela vison*, up to 5,140 ng/g w.w.) (Kannan et al., 2002). The highest concentrations were found in otters from the south-central more urban parts of Sweden. A significant increase in concentrations since 1972 was seen in all PFAS but in particular for the long chain (9-14 carbons) perfluoroalkylated carboxylic acids, with a yearly increase ranging from 7-32% for the individual congeners. It is not known to what extent these substances are toxic to otters but the concentrations are considered very high.

There are a few other studies on otter health in connection with contaminants; one in particular is a study on

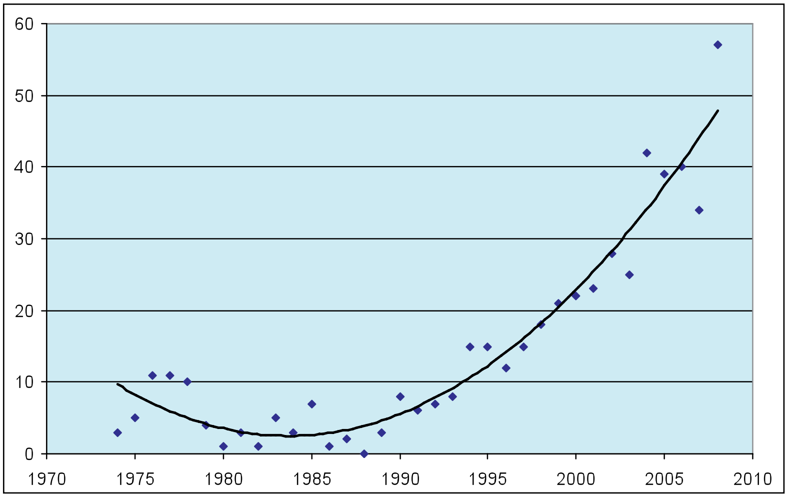


Figure 9.18. Number of dead otters found dead 1974-2008 and sent to the authorities (n=506). The increasing number indicate an increasing population.

PCBs in 77 Danish otters in relation to health condition. Here a tendency of increasing infectious diseases was seen with increasing PCB concentrations in liver of otters (Leonards et al., 1996) and a decrease of Vitamin A with increased concentrations of PCB. In Swedish otters a significant correlation between altered bone mineral density and high concentrations of PCB has been seen (Roos et al., 2010). However, the severe disease complex with connections to environmental contaminants that were seen in Baltic seals has not been observed in otters.

Trends in Swedish Otter Population

The otter is listed in a Swedish game law since 1972. If found dead it should be reported to the authorities, who send the carcass to the Swedish Museum of Natural History. The number of dead specimen can reflect the population status. In addition, many otter surveys have been carried over the years. They give a fair view of the population status and distribution and most of them report of an increasing otter population since the 1990s. During the 1990s the population start recovering. During this decade a total of 125 otters were reported dead, a remarkable increase compared with the 26 reported in the 1980s. Finally, between 2000 and 2009 more than three times as many dead otters were reported (n=383), in comparison to the decade before indicating a strong comeback (Figure 9.18).

Figure 9.19. A grey seal from the south coast of Sweden. Photo: Jan-Åke Hillarp.

Seals

There are three seal species living in the Baltic Sea, grey (*Halichoerus grypus*), ringed (*Phoca hispida*) and harbour or common seals *(Phoca vitulina*). The grey seals (Figure 9.19) and ringed seals are most abundant and the majority of them live in the central and northern parts of the Baltic Sea. Besides the grey seal population in the Baltic Sea there are further two populations of grey seals, in the western Atlantic in Canada and northern USA and in UK and Ireland. The ringed seals are mainly distributed in the circumpolar Arctic coasts. In the Baltic Sea they

Figure 9.20. Grey seal uterus, occlusions are seen in both horns. Photo: Bengt Ekberg/SVA.

Figure 9.21. Grey seal uterus showing three cut open leiomyomas. Photo: Charlotta Moraeus.

inhabit mostly the Bothnian Bay, the Gulf of Finland and the Gulf of Riga. Their distribution is mainly dependent on the ice condition since they breed and nurse their pubs in snow-covered lairs. The Baltic ringed seal is categorized as its own subspecies (Härkönen et al., 1998).

Harbor seals are found in temperate, subarctic, and arctic coastal areas on both sides of the North Atlantic and North Pacific oceans. In the Baltic Sea they are distributed on the southern coasts of Sweden and Danish Straits. On the Swedish southeastern coast there is a genetically separated small population of harbour seals (Härkönen et al., 2005).

Seals give birth to a single pup, once a year. In the Baltic, the period of birth peaks for grey seals in the beginning of March, for ringed seals in the beginning of April and for harbour seals in the end of June. All three species have a period of about 100 days of delayed implantation of the blastocyst in the uterus (King, 1983).

The numbers of Baltic seals decreased dramatically during the 1900s due to hunt and thereafter to pollutants (Hook and Johnels, 1972; Olsson et al., 1975; Helle and Olsson, 1976). During the most critical period in the 1970s, the Baltic grey seal population was below 4,000 individuals. In the early 1900s the population was estimated to 100,000 grey seals (Hårding and Härkönen, 1999). Since the mid 1980s the annual increase of the population along the Swedish Baltic coast was estimated to be 8% (Karlsson et al., 2008). The total number of grey seals in the Baltic Sea 2010 was estimated to 22,000 individuals; however after 2006 no further increase of the population has been observed. The Baltic ringed seal were estimated to have been about 200,000 in the early 1900s and about 5,000 in the 1980s. Thereafter the population have increased with about 4.3 % per year (Hårding and Härkönen, 1999). The ringed seal is vulnerable to climate changes since they are dependent on ice for reproduction.

During the early 1970s a high prevalence of lesions in female reproductive organs of Baltic ringed and grey seals was reported, and also other pathological changes in these species were described (Helle and Olsson, 1976; Bergman and Olsson, 1985; Bergman, 1999; Bergman et al., 1992; Bergman et al., 2001; Bäcklin et al.,2003). Female reproductive lesions consisted of occlusions of the uterine horns, which will prevent pregnancy, and stenosis, narrowing of the lumen of the uterine horns (Figure 9.20). These lesions were observed in both grey and ringed seals. In grey seals also a high prevalence of uterine tumours (leiomyomas) was observed (Figure 9.21). These uterine lesions correlate mostly to the earlier high levels of polychlorinated biphenyls (PCB) in the Baltic biota (Helle and Olsson, 1976; Bredhult et al., 2008). Occluded uterine horns have not been observed since 1993 in grey seals and the prevalence has also decreased in ringed seals, although

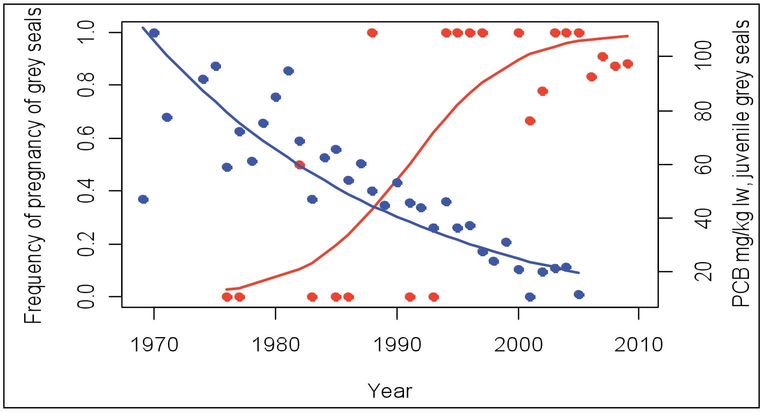


Figure 9.22. The pregnancy rate (red in the diagram) of grey seal females (n=92, 5-25 years old) collected between August and March 1976-2009. Also, 242 juvenile grey seals, collected 1968-2005 were analyzed for PCB (blue in the diagram) in blubber (mg/kg l.w.).

occlusions are still found in ringed seals (Bergman, 1999; Helle et al., 2005). As the prevalence of uterine occlusions decreased and the prevalence of pregnancy increased in investigated Baltic grey seals the population started to increase and the concentrations of PCB in young grey seals decreased (Figure 9.22. Bergman, 1999; Roos et al., 2010). Besides reproductive lesions there were also lesions recorded in integument, skull bones (Figure 9.23), intestine (mostly colonic ulcers), arteries, adrenals (Figure 9.24) and kidneys (Bergman and Olsson, 1985; Bergman, 1999). Also a decrease in the prevalence of most of these lesions has been observed (Bergman, 2007). In the 1970s decreases of DDT and PCB concentrations were recorded in Baltic biota including grey seal juveniles (Figure 9.22. Olsson and Reutergård, 1986; Bignert et al., 1995; Roos et al., 2007).

In the period 1987-1996 the prevalence of intestinal ulcers increased significantly in 1-3 years old grey seals compared with the period 1977-1986 (Bergman, 1999). Ten years later (1997-2006) the prevalence had increased significantly in 4-20 years old grey seals and it is still high (Karlsson and Bäcklin, 2009; Bäcklin et al., 2010a). The high prevalence of colonic ulcers seems unique for the Baltic population of grey seals since only one case has been reported outside the Baltic Sea (Baker, 1980). The cause of the increase is still unknown but a defect immune response to lesions made by intestinal acanthocephalan parasites has been suggested (Bergman, 2007). Since the beginning of 2000s a significant decrease in blubber thickness has been recorded in investigated grey seals (Figures

Figure 9.23. Grey seal skull (upper jaw) with severe loss of bone around the teeth and loss of teeth. Photo: Charlotta Moraeus.

Figure 9.24. Transverse sections of grey seal adrenals. A thickened cortex is seen to the right. Photo: Bengt Ekberg.

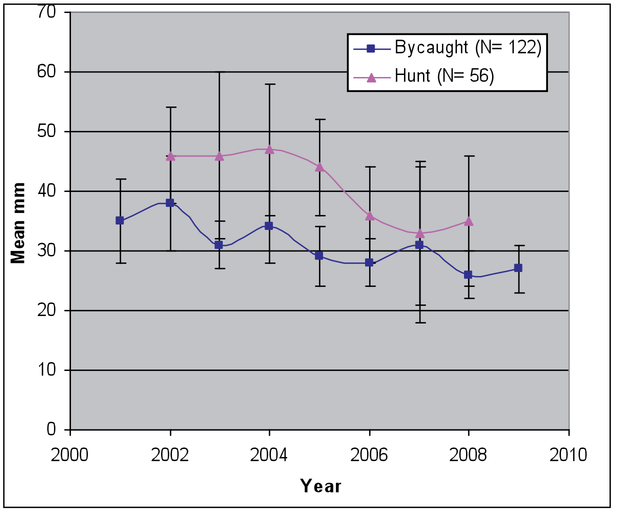


Figure 9.25. Mean annual blubber thickness in 1-4 years old non pregnant Baltic grey seals sampled from hunt and bycatch. The decrease is significant ($p < 0.002$).

Figure 9.26. A resting slim grey seal female from the east coast of Gotland in the Baltic Proper. Photo: Erik Isaksson.

Figure 9.27. Harbor porpoise, at Fjord & Bælt center, Denmark. Photo: Anna Roos.

9.25-26. Karlsson and Bäcklin, 2009). In 2008, there was also a significant increase in prevalence of grey seals showing liver lesions in relation to liver flukes (Bäcklin et al., 2010b). These changes could be signs of ecosystem changes in the Baltic with changes in the seals prey species and quality. During the last 30 years, sudden regime shifts in Baltic sub-ecosystems have been detected, the last one in the sound, Gulf of Finland and the Baltic proper coastal area in 2000-2003 (Bergström et al., 2010).

Whales

The harbor porpoise (*Phocoena phocoena*) is the only whale living permanently in the Baltic Sea and until the 1950s it was common especially in Baltic Proper (Figure 9.27). A survey study performed 1995 revealed that the population included approximately 600 individuals. Later estimations performed 2002 shows a dramatic decline and the number was estimated to 100-150 individuals in the Baltic Sea. The explanation to the dramatic decline of porpoise during recent years is not fully known, but major threats to the population in the Baltic include accidental trapping fishing gear and environmental contaminants.

Conclusions

In parallel with the decreasing contaminant trends, populations of several threatened species e.g. grey seal, white-tailed sea eagle and otter, have turned and are now increasing in numbers and distribution. On the other hand, an increased prevalence of intestinal ulcers and decreasing blubber thickness in grey seals are worrying indications that potentially could develop into a severe population decline. White-tailed sea eagles in the south Bothnian Sea still have a higher frequency of dead eggs and produce fewer offspring than other eagles on the Baltic coast. Through their position at a high trophic level these species are important indicators of ecosystem health. However, when the old, “classic” contaminants such as DDT and PCB were banned, other chemicals came into use, and several of them are found in increasing concentrations in biota, for example HBCDD and perflourinated compounds, including PFOS which are found in extremely high concentrations in Swedish fauna.

Shipping and Oil Production

10

Sverker Evans
The Swedish Agency for Marine and Water Management, Göteborg, Sweden

Introduction

Ocean transportation is steadily growing and shipping has the potential to become a sustainable way of transportation. However, commercial shipping has a variety of impacts on the marine environment, such as operational discharges, accidental and occasionally illegal releases of oil and hazardous substances, emissions of air pollutants such as sulphuric and nitrous oxides, introduction of non-indigenous species through the vector of ballast water, and loss of vessels and/or cargo. In addition, navigational requirements in coastal areas include dredging and disposal of sediments and large-scale development for port facilities.

The Baltic has some of the densest maritime traffic in the world. At any moment 2,000 ships are on route on the average. A combination of heavy shipping traffic, shipping lanes that cross each other, narrow straits, shallow waters and long periods with ice cover makes the Baltic a difficult area for navigation, with an apparent risk of shipping accidents.

Actions to mitigate environmental problems from ships have been part of the work by the Baltic Marine Environment Commission (HELCOM) since its start in 1974, and numerous recommendations related to shipping have been incorporated in the countries´ legislations. In addition, the Baltic has also been designated a

Figure 10.1. Fu Shan 2003. Source: Swedish Coast Guard.

Special Area under several MARPOL Annexes as well as a Particularly Sensitive Area. However, despite many years of international co-operation, several environmental problems remain.

Hazardous Substances

Operational losses of hazardous substances from shipping include losses by leaching biocides, tributyltin (TBT) used in antifouling coatings, and of zinc, copper and aluminium applied as anodes to ships hull, ballast tanks and cooling system as protection from corrosion. Data on losses by leaching of hazardous substances from antifouling coatings and anodes are hardly available. However, a first estimate suggests that the magnitude of most of the metals to the Greater North Sea equalled their direct discharges or atmospheric deposition; the magnitude of total losses of TBT in that sea region has been estimated to be around 130 tonnes in 2002 (OSPAR, 2006h).

The International Convention on the Control of Harmful Antifouling Systems, adopted by the IMO in 2001, provides a global framework for action to limit adverse effect on the marine environment and human health caused by antifouling systems. Adverse effects of TBT have been linked primarily to shell deformations and effects on reproduction of molluscs. A world wide ban on the use of TBT as antifouling agent has been agreed in the IMO framework from 2008.

Emissions of Air Pollutants

The combustion of marine fuels results in emissions of air pollutants, such as sulphur dioxide, nitrogen oxides (NO_x), particulate matter, and volatile organic compounds. These air pollutants can damage human health and contribute to acidification and eutrophication, damaging sensitive ecosystems. Emissions of air pollution by ships in or near ports are generally quite large, particularly compared to the amounts emitted by vehicles on shore. In contrast to the expected progress in reducing emissions from land-based sources, air emissions from shipping are expected to increase, unless further actions are taken.

A large part of the atmospheric input of nitrogen to the Baltic Sea could be attributed to NO_x emissions from shipping. However, technical solutions which cut NO_x emissions more than 95% are at hand, and economic instruments such as differentiated shipping lane fees could be parts of the toolbox to reduce airborne emissions. Given the contribution of shipping to local and regional environmental and health problems and in order to meet mandatory air quality standards, the need for additional measures to reduce marine air emissions, such as on-shore power supply where appropriate is highlighted.

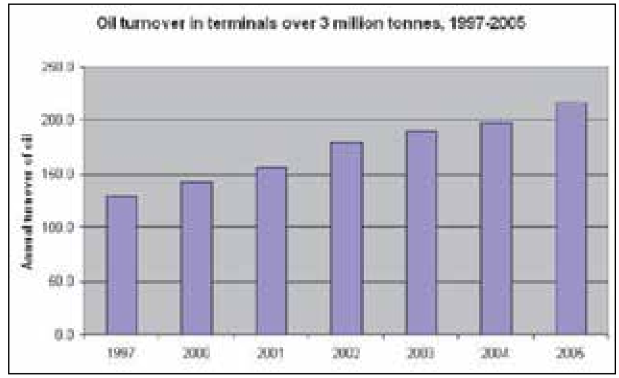


Figure 10.2. Oil turnover in the largest terminals of the Baltic Sea 1997-2005. Source: http://www.helcom.fi/groups/response/en_GB/main/.

Important steps have been taken to reduce air pollution from ships, such as the entry into force of Annex VI to MARPOL 73/78, and the designation of the Baltic Sea as a sulphur oxide emission control area. However, there are concerns that if no further measures are introduced, emissions of SO_x and NO_x from international shipping around Europe may have surpassed the total emissions from all land-based sources in the 25 EU Member States combined by 2020. This will have an impact on environmental and health problems. Furthermore, further work is needed on measures to reduce the climate change impact of international shipping.

Ship-generated Waste

Despite the wide range of measures taken in recent years, floating debris in the marine environment still remains a significant source of pollution causing environmental, safety and economic problems to marine and coastal environments, as well as to coastal communities. Accidental and intentionally discharges from ships pose a problem, which it is urgent to address through inter alia operational measures and by investigating the scope for the development of incentives for ships with a record of good environmental performance.

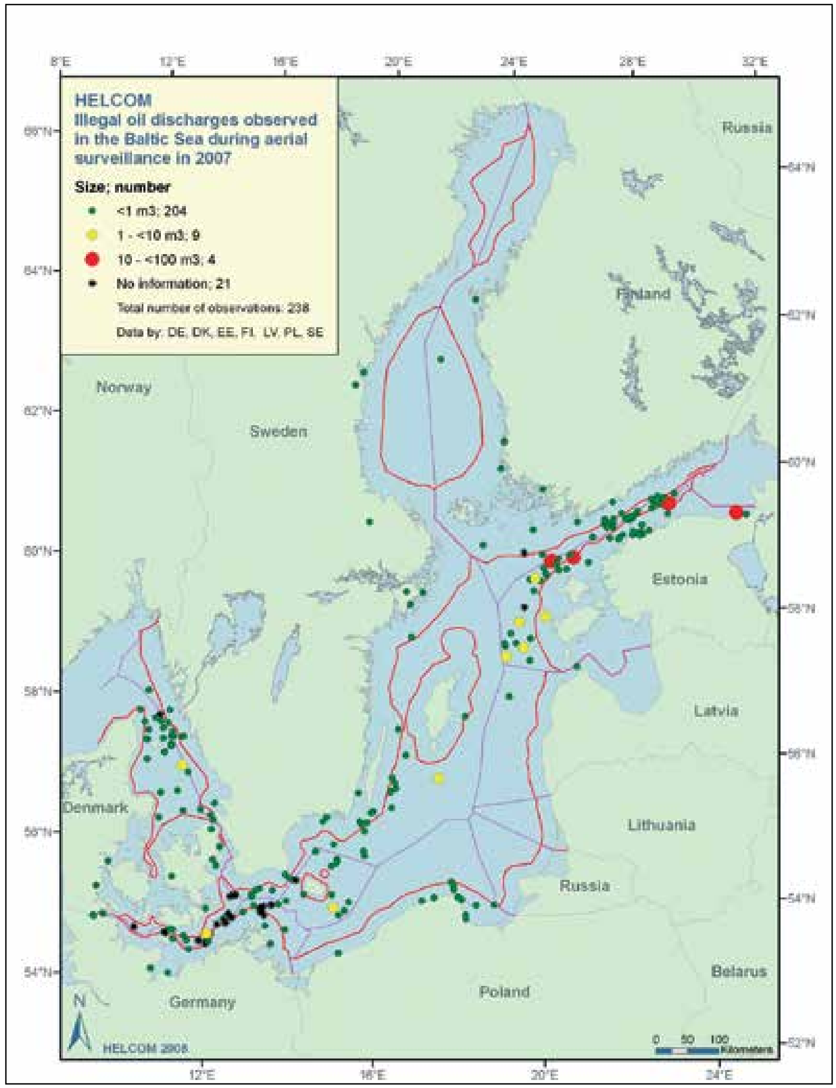
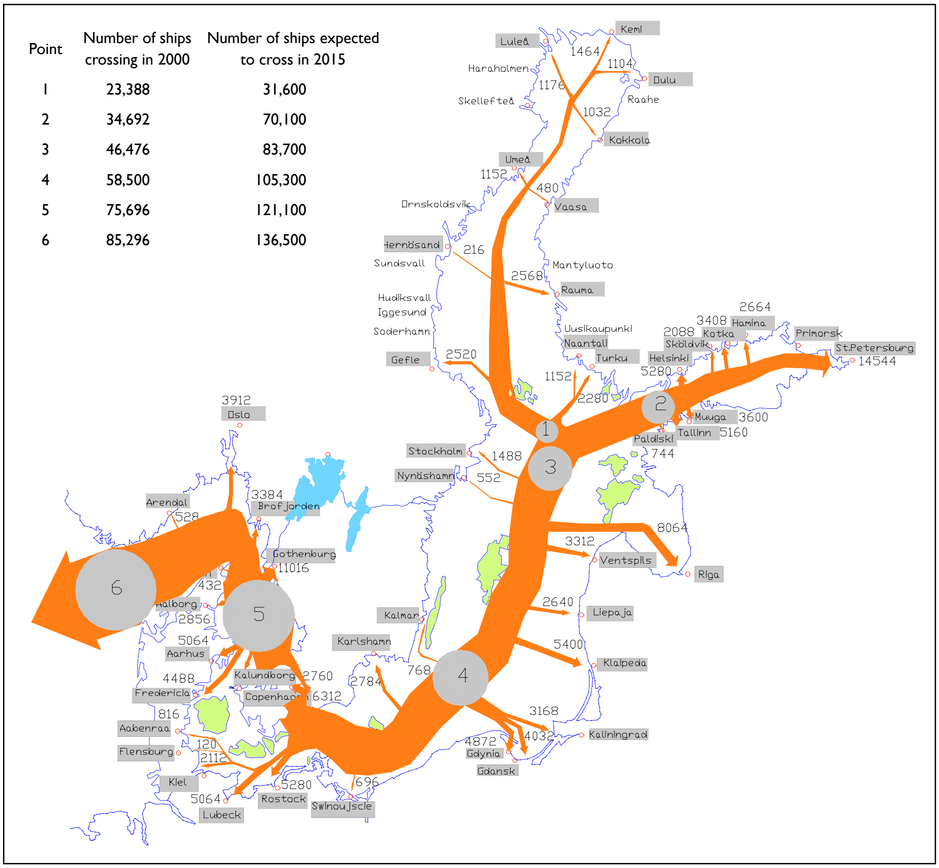


Figure 10.3. Illegal oil spills in 2007. Source: Helcom. http://www.helcom.fi/stc/files/Maps/oilspills/oilspills2007.pdf.

The provision of port waste reception facilities for mandatory use is one of the major tools for managing the disposal of garbage and other ship-generated wastes plus cargo residues, and for preventing illegal discharge into the Baltic Sea. The delivery of waste from ships and the provision of port waste reception facilities are both requirements of MARPOL 73/78 Annex I, IV and V and the EC Directive 2000/59/EC on Port Reception Facilities for Ship-generated Waste and Cargo Residues. Consistent with this Directive, a number of charging systems are in place in the North Sea States, and HELCOM Member States have a “No special fee” system for the Baltic Sea.

Special Areas

In Annexes I *Prevention of pollution by oil*, II *Control of pollution by noxious liquid substances* and V *Prevention of pollution by garbage from ships*, MARPOL 73/78 de-



Point	Number of ships crossing in 2000	Number of ships expected to cross in 2015
1	23,388	31,600
2	34,692	70,100
3	46,476	83,700
4	58,500	105,300
5	75,696	121,100
6	85,296	136,500

Figure 10.4. Total transport figures of the Baltic Sea in 2000 and forecast for 2015. Based on VTT, 2002.

fines certain sea areas as "special areas" in which, for technical reasons relating to their oceanographically and ecological condition and to their sea traffic, the adoption of special mandatory methods for the prevention of sea pollution is required. Under the Convention, these special areas are provided with a higher level of protection than other areas of the sea.

The International Maritime Organisation (IMO) meeting in London (2005) decided to designate the Baltic Sea as a "Particularly Sensitive Sea Area."(PSSA).

An Integrated Approach

The environmental impacts from the shipping sector need to be considered in an integratedway and be addressed at the national, regional and international levels through the *Clean Ship Approach* for sustainable shipping.

The concept addresses the design, construction, operation and recycling of vessels and thus includes a cradle to grave perspective on shipping. Each phase contains several elements on which the Clean Ship Approach can be built and which may lead to actions towards cleaner shipping. The long term goal is to eliminate any negative environmental impacts of shipping. The Clean Ship Approach will provide an increased opportunity for transport managers to choose environmentally sound sea transport options.

Ensuring Environmentally Friendly Maritime Activities

Due to the international nature of shipping, the measures adopted at the national or regional level can only have limited impact on environmental impacts from shipping in a specific region. The IMO is the global regulator for

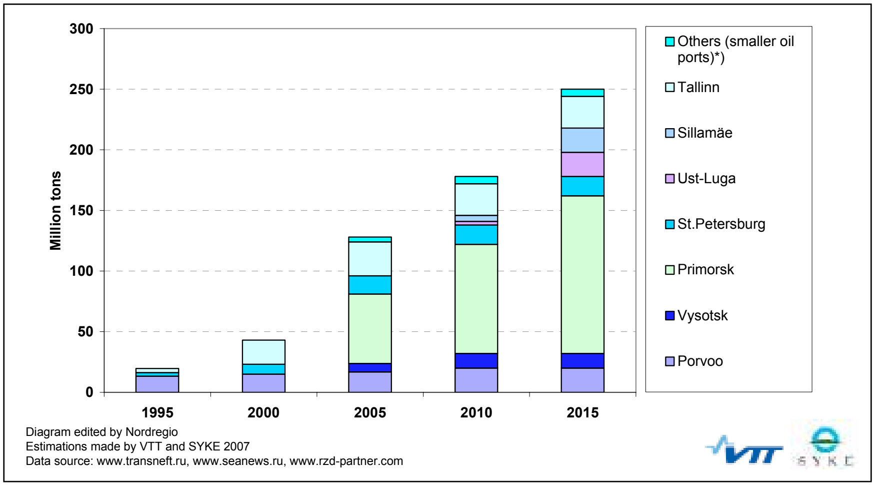


Figure 10.5. Oil transports from terminals in the Finnish Gulf 1995-2005 with calculated expansion to year 2015. Source: SYKE, Finnish Environment Institute and VTT Technical Research Centre of Finland.

the shipping industry and the international measures constitute the foundation on which regions and nations can build on by introducing non-discriminatory economic incentives to further reduce pollution from ships within their jurisdiction. All riparian states must therefore take active part in global actions initiated within the IMO to reduce environmental impacts from ships.

The HELCOM *Baltic Sea Action Plan* is an ambitious strategy to restore the good ecological status of the Baltic marine environment. This new strategy will be a crucial stepping stone for wider and more efficient actions to combat the continued deterioration of the marine environment resulting from human activities. As one of the first schemes to implement the ecosystem approach to the management of human activities, the action plan may lead to new innovative changes in the ways the environment in the Baltic Sea region is managed.

The share of the total pollution loads in the Baltic Sea originating from maritime activities is growing, partly due to the stricter controls now applied to limit pollution from land-based sources.

In order to reach to goal to carry out maritime activities in the Baltic Sea in an environmentally friendly way, further actions are needed with regard to six issues of major importance for all the Baltic Sea coastal countries. Six corresponding management objectives have been defined:

- No illegal pollution
- Safe maritime traffic without accidental pollution
- Efficient response capability
- No introductions of alien species from ships
- Minimum air pollution from ships
- Oil production

No Illegal Pollution

The annual numbers of illegal discharges of oil in the Baltic Sea have decreased. However, the member countries' ability to detect oil discharges must still be reinforced, also at night or during periods of poor visibility when discharges on purpose are more likely to occur.

The problem of intentional discharges does not only concern oil. Plastics and synthetic materials, which are durable and degrade slowly, have become the most abundant form of marine waste. The international shipping community should continue to develop quality management systems on board ships that address, and set down procedures for, the handling, storage and disposal of all wastes and encourage waste minimization and recycling, recognising the importance of adequate port waste reception facilities.

HELCOM states should act to speed up substitution of harmful antifouling systems with less harmful alternatives and to undertake to give full effect to the International Convention on the Control of Harmful Antifouling Systems on Ships (AFS Convention) in order to reduce

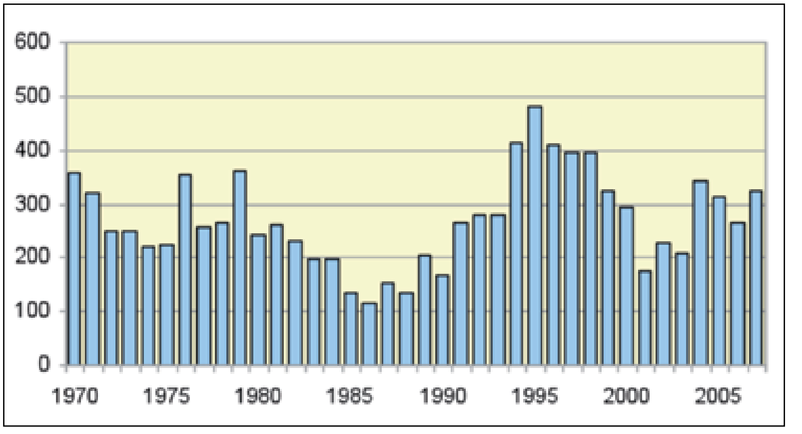


Figure 10.6. Confirmed oilspillages in swedish coastal waters 1970-2005. Source: Swedish Coast Guard.

or eliminate adverse effects on the marine environment and human health caused by antifouling systems.

Safe Maritime Traffic without Accidental Pollution
The statistics on shipping accidents in the Baltic shows an increasing number of groundings and collisions. This is mainly due to the growing intensity of ship movements, which requires the Contracting Parties to put even more emphasis to ensure the safety of navigation. One way to do this is to make full use of the new tools available to control shipping traffic, notably the Automatic Identification System. Considering the increase in the transportation of oil products and the difficulties to respond to oil spills in icy conditions, further measures should especially be taken to increase safety during winter time.

Efficient Response Capability
The risk of shipping accidents cannot be totally eliminated, and there is a need to ensure that the sufficient emergency and response resources are in place in the Baltic Sea states. Much has been done to build up an adequate emergency capacity and response capability. Around 30 emergency tugs with bollard pull of 50 tonnes or more, and around 40 sea-going response vessels are located at different areas in the Baltic. To build up an sufficient capacity is a costly and timely process, so a step-wise approach may be applied, starting with assessments of the risk of accidents in the various sub-regions. Such assessments have been started in most areas of the Baltic, but none have yet reached a stage where conceivable missing capacities have been quantified to allow decisions to fill in such gaps in the most efficient way possible.

No Introductions of Alien Species from Ships
Increasing numbers of non-native species are being observed in seas all around the world, and the Baltic is no exception. Shipping is the most important vector of unintentional species introductions into aquatic environments, due to releases of ballast water and the fouling of hulls. The entry into force of the IMO International Convention for Control and Management of Ships' Ballast Water and Sediments, 2004, would be the most important step forward to tackle this problem. The ratification of the Convention by the HELCOM countries is a challenging goal, but would provide an effective legislative tool to reduce the risk of introductions of non-native species into the Baltic. At the same time the need for measures along inland waterways connecting the Baltic Sea and the Ponto-Caspian regions should also be addressed.

A regional implementation strategy within OSPAR, HELCOM and other relevant regional organisations should be taken forward in order to implement a Ballast Water Management Strategy for the North Sea/North West Europe and the Baltic Sea to control the risks of non-indigenous species invasion through the control of ballast water exchange in line with the IMO Convention. This will establish adequate mitigation and control measures for the Baltic Sea based on ballast water exchange practices, prior to the IMO Convention's water quality standards entering into force.

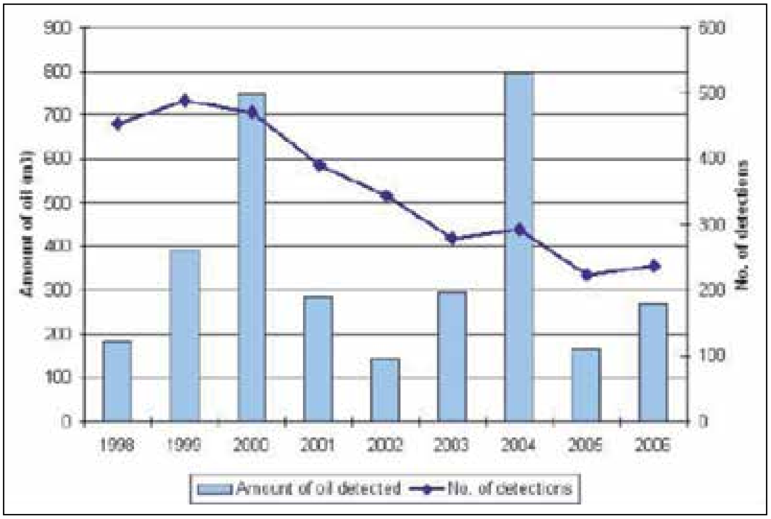


Figure 10.7. Number of detected oil spills 1998-2006. Source: VTT Technical Research Centre of Finland.

Minimum Air Pollution from Ships

Air emissions from shipping are significant and are expected to grow in future. However, there are already feasible and cost-effective methods to substantially reduce air pollution from ships. The HELCOM Contracting Parties should continue to elaborate common positions and provide joint inputs to the ongoing global legislative processes to ensure that the best solutions are promoted and up-to-date technology is applied.

Improvements in ships' environmental performance should also be promoted by introducing non-discriminatory economic incentives to further reduce pollution.

The possibility of integrating shipping into emission trading regime(s) to give incremental reductions in ships emissions should be considered. To strengthen the credibility of trading schemes, emission reductions should be monitored, calculated and verified in practice.

Installation of NO_x catalysers in those ships flying the flag of Baltic States which are on fixed routes in the Baltic Sea should be economically supported, and the introduction of differentiated shipping lane fees should be promoted.

Oil Production

The volume of maritime oil transportation in the Baltic Sea has increased significantly during the last decades. Ships within the Baltic area annually transport around 160 million tonnes of oil and the volume is anticipated to increase to 250 million tonnes in 2015. The technological standard of shipping safety and regulations which cover various kinds of environmental hazards has been improved during the last decades, not the least as a more or less direct consequence of major oil accidents. Despite the fact that total quantities of oil lost to the seas have been decreasing during the last three decades globally, shipping safety improvements, especially the phasing out of single hull vessels, might not offset the steep increase in transports in the Baltic Sea. The major driving force behind the contemporary increase in oil transports in this area is the high oil price and Russian export ambitions as well as an increased economic activity in general in the former Soviet Union region.

The environmental hazards of oil spills are of two different kinds. Firstly, accidents may produce large and dramatic effects that require substantial and swift responses by several authorities in cooperation. Although most of the previous oil spill accidents in the Baltic Sea have been comparably small in size, the environmental effects may still be substantial, because of the environmental sensitivity of this brackish water area. Secondly, it is a fact that oily sludge or various forms of oil-contaminated residues is discharged in open seas on purpose. The Baltic Sea as well as the North Sea has received the status of Special Area under Annex I of the IMO MARPOL 73/78

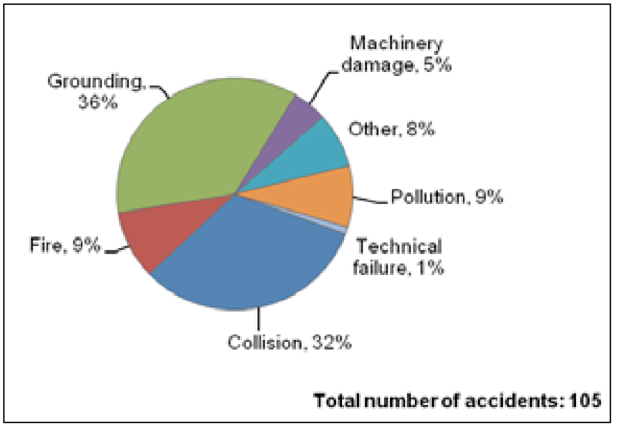


Figure 10.8. Types of ship accidents in the Baltic Sea 2009. Source: Helcom. http://www.helcom.fi/groups/response/en_GB/main/

Convention. This means that all discharges of oil or oil mixtures from shipping are prohibited.

The IMO decided in 2003 to accelerate the phasing-out of single hull tankers by 2010. The phase-out of single hull tankers will contribute to an increased level of maritime safety and environmental protection. However, this development has to be combined with a high level of control and maintenance of double hull tankers in order to avoid these vessels turning into high-risk ships.

Part D

The North American Great Lakes/St. Lawrence River and Estuary

Authors: David B. Bunnell, Steven Bursian, Guy Fitzgerald, John D. Fitzsimons, David J. Jude, Jeffrey M. Levengood, David P. Macarus, Charles P. Madenjian, Daniel Martineau, Matthew Zwiernik, Frouke Vermeulen, Chip Weseloh and Martha Wolgamood

Coordinating Author: Jeffrey M. Levengood and Daniel Martineau

Geological, Hydrological and Anthropogenic Features

11

Jeffrey M. Levengood
University of Illinois at Urbana-Champaign, Urbana, IL, USA
Daniel Martineau
University of Montreal, Saint-Hyacinthe, QC, Canada

The foundation for the North American Great Lakes-St Lawrence River (GL-SL) basin was laid millions of years ago during the Precambrian period, with the formation of a large expanse of bedrock of volcanic origin. This ancient bedrock, now called the Canadian Shield, extends from the Arctic Ocean southward to the northern portion of the GL-SL basin. After the formation of the Shield, during the Paleozoic Era, marine seas were formed and left deep sedimentary deposits that afterward compacted to form shale and limestone. The GL-SL basin as we know it now was shaped by the advance and retreat of the Wisconsin Glaciation, which ended approximately 10,000-15,000 years ago when the Laurentide Ice Sheet pulled back, after covering most of North America with a ice layer more than 1-mile thick in some places. On their way to the north, glaciers scraped clean the bedrock of the Canadian Shield, depositing till that would later become the rich soils of the southern GL-SL basin.

The lake basins, formed by the pressure and scouring action of the ice sheets, were filled with water during the warming and the ice retreat. Nowadays, these "inland seas", all above sea level, are arranged in a succession of decreasing elevation from west to east, which explains their slowly emptying into the Atlantic through the St Lawrence River (SLR) and Estuary (SLE) (Figures 11.1 and 11.2). The Niagara Falls, between Lake Ontario and Lake Erie, 53 m high, represent the largest difference in elevation between two Great Lakes. The Great Lakes have long water-retention times, ranging from 2-3 years for Lake Erie, the shallowest, to 191 years for Lake

Figure 11.1. North American Great Lakes-St Lawrence River basin. Map provided by François Boudreault and Nathalie Gratton, Environment Canada.

Superior. The 5 Great Lakes hold 95% of North America's surface water and 21% of world's fresh water supply. The Great Lakes basin alone drains 522,000 km^2, whereas the entire GL-SL watershed encompasses approximately 1.6 million km^2 (Figure 11.1). This system includes the largest freshwater lake in the world by surface area, Lake Superior, with 82,000 km^2, which is also the deepest of the Great Lakes - more than 406 meters deep.

High above sea levels are the many rivers and lakes associated with the Great Lakes, also excavated by the retreating ice. All thus represent formidable amounts of potential or exploited hydroelectricity. Early in the 20th century, the metallurgy industry – particularly the aluminum industry – which requires important electrical power, along with the power authorities of Quebec, Ontario and New York State were all quick to grasp the economic potential of these waterways.

Historically, the Great Lakes only have had a one-way, limited hydrologic connection with the sea, i.e., the Atlantic Ocean, via the St. Lawrence River and Estuary. Early Europeans who arrived from the Atlantic coast and through the St Lawrence were faced with falls and rapids that were obstacles to the transportation of people and merchandise westward. The potential for hydropower and the absence of environmental concern led to extensive damming and dredging over the first three quarters of the 20th century (Morin and Leclerc, 1998). With the advent of a series of canals and diversions to facilitate shipping, the Great Lakes watershed became hydrologically connected with the Mississippi River, Hudson River, Ohio River, and Hudson Bay watersheds. In addition, a series of intra-basin projects, most notably the channelization of the Detroit, St Claire and St Mary's Rivers, have changed the connections between the upper and lower Great Lakes. The construction of the Welland Canal, completed in 1932 to allow ships to circumvent the Niagara Falls, has had perhaps the greatest environmental impact on the system by allowing access not only to oceangoing vessels but, also to the non-native plants, animals and even pathogens such as viruses, bacteria and protozoa contained in the ballast water of these ships. At the end of the 90s, around 2,000 ships navigated the Saint Lawrence Seaway (SLS) yearly. Together these ships released around 13 Mt of ballast water yearly.

To the north of the GL basin, the climate is cold, with coniferous forests and shallow, acidic soils overlying the Canadian Shield. The extensive coniferous forests provided lumber and wood pulp for a growing human population. The active geologic history of the region explains the presence of various valuable minerals, ranging from coal to zinc, located near the surface and thus easily accessible. In particular, the presence of rich iron ore deposits in the Lake Superior region and on the north shore of the SLE, along with the capability of shipping that ore through the SLE , all explain the establishment and development of large steel mills on the shoreline of the

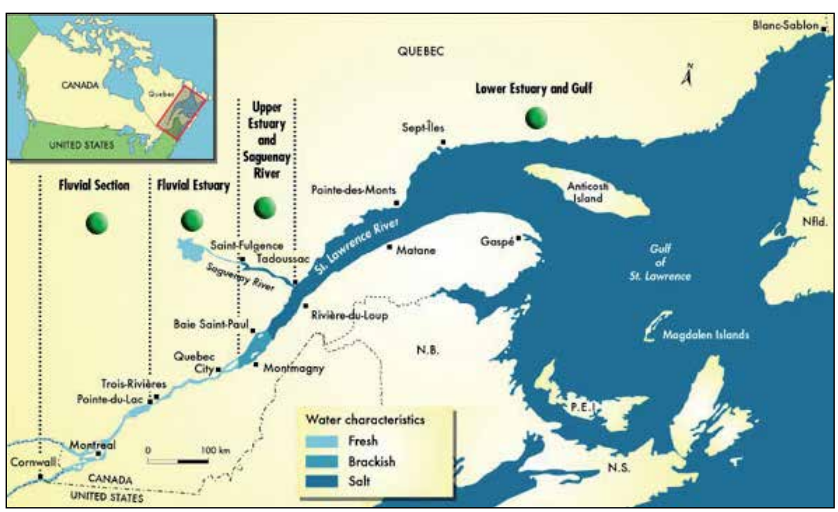


Figure 11.2. Hydrographic segments of the St. Lawrence River: the Fluvial Section, the Fluvial Estuary, the Upper Estuary and Saguenay River, and the Lower Estuary and Gulf. Source: St. Lawrence Centre, 1996.

southern Great Lakes. The mills, along with a growing workforce emigrating from Europe, would become the origin of the first industrial cities of the southern basin, e.g., Chicago, Cleveland, Detroit, Green Bay, Hamilton, and Windsor. Indirectly thus, the Canadian shield and the SLS were at the origin of the birth and development of the car industry in North America.

Up north in Quebec, the congruence of abundant hydropower driven by the steep fall in elevation, the steady flow rates of water and ready access to worldwide bauxite sources through the SLE led to the development of a large aluminum smelting industry on the Saguenay River, a major SLE tributary (the Saguenay fjord 170 km long, 280 m deep, is a significantly sized estuary *per se*).

To the south of the GL-SL basin, the climate is warmer, with deep, fertile soils deposited as glacial drift or as glacial lake and river sediments. Much of the original deciduous forests, prairies, and wetlands have been cleared and drained for agriculture and urban development, explaining that more than one third of the basin is exploited by agriculture. The GL-SL region is home to over 10 and 30% of the human populations of the US and Canada respectively (Government of Canada and US EPA, 1995) and most of that population is concentrated in industrial urban centers. Approximately 79% of the 33 million inhabitants live in 5 cities: Chicago, Toronto, Detroit, Montreal and Cleveland. Large industries are established in these major cities and in smaller ones, adding heavy metals, organic contaminants, and pathogens to the system, while at the same time removing ever-increasing amounts of water for drinking, sanitation, industrial activities and irrigation.

The SLE is among the largest estuaries in the world. Its downstream part, the "Lower Estuary", is much deeper than larger estuaries such as the Chesepeake Bay (U.S.A), Spencer Gulf (Australia) and, Gulf of Suez (Egypt). Its freshwater flow rate is second only to that of the Mississippi but brings 100 times less suspended matter because the GLs trap much of it. The SLE is considerably long, wide and deep: it extends over 1,500 km from its inland origin to the Atlantic Ocean, it is up to 60 km wide and its depth reaches over 350 m (El-Sabh and Silverberg, 1990). The Gulf is large enough to be influenced by the earth rotation (Coriolis effect), which creates a transverse current that superimposes on the longitudinal flow.

Table 11.1. Similarities between the Saint Lawrence system and the Baltic Sea. .

<table>
<tr><th colspan="2">Characteristics</th><th>St Lawrence River, Estuary and Gulf</th><th>Baltic Sea</th></tr>
<tr><td rowspan="2">Winter</td><td>Water temperature</td><td colspan="2">Low</td></tr>
<tr><td>Partial ice cover</td><td colspan="2">Yes</td></tr>
<tr><td colspan="2">Tide</td><td colspan="2">+</td></tr>
<tr><td colspan="2">Salinity</td><td colspan="2">Brackish</td></tr>
<tr><td colspan="2">Water area</td><td>211,517 km^2 (River, estuary, and Gulf)</td><td>375,000 km^2</td></tr>
<tr><td colspan="2">Emptying</td><td colspan="2">Slow</td></tr>
<tr><td colspan="2">Length</td><td colspan="2">1,600 km</td></tr>
<tr><td colspan="2">Watershed</td><td colspan="2">1.6 million km^2*</td></tr>
<tr><td colspan="2">Origin</td><td colspan="2">Last glaciation</td></tr>
<tr><td colspan="2">Biological diversity</td><td colspan="2">Low</td></tr>
<tr><td colspan="2">Cod population</td><td colspan="2">Threatened</td></tr>
<tr><td colspan="2">Contaminants</td><td colspan="2">+</td></tr>
<tr><td colspan="2">Marine mammals: contamination and lesions</td><td colspan="2">Severe</td></tr>
<tr><td colspan="2">Contaminants</td><td colspan="2" rowspan="5">+</td></tr>
<tr><td colspan="2">Invasions by foreign animal species (mostly pontocaspian)</td></tr>
<tr><td colspan="2">Toxic algal blooms</td></tr>
<tr><td colspan="2">Eutrophication</td></tr>
<tr><td colspan="2">Hypoxia</td></tr>
</table>

*: drainage area of the whole Saint Lawrence system, including the Great Lakes.

Overall, water circulation in the SLE, like in the GL, is slow: SLE surface waters are replaced over several months whereas deep waters are replaced over several years. This is partly due to the semi-enclosure of the SLE and Gulf water (similar in that regard to the Arctic Ocean) connected to the Atlantic Ocean by only two openings.

Along the SLE northshore, the SLE water moves like a conveyor belt. The upper part of the belt is composed of brackish, lighter water that runs downstream toward the Atlantic. In turn, the deep cold water of the North Atlantic is pumped upstream with numerous macrozooplankton crustaceans. This movement, the Labrador Current, flows southwestward within a 350-meter deep trench termed the Laurentian channel. Because cold water is denser, this current sticks to the bottom and, at the head of the Laurentian channel, at the level of Tadoussac, a small village on the SLE north shore, it hits a huge, almost vertical underwater wall where depth abruptly decreases from 300 to only 25 meters. Because of this configuration, the icy water surfaces locally and recreates a subarctic highly productive marine environment, which explains the presence of a unique population of beluga whales and of 19 other species of marine mammals.

The SL River, Estuary and Gulf on one hand and the Baltic Sea on the other hand share several physical, hydrographic and biological similarities (Table 11.1), explaining that they also share common problems. Specific to the province of Quebec, one of the largest producers of hydroelectricity in the world, is the anthropogenic disturbance of the freshwater flow rate caused by the unseasonal release of vast amounts of freshwater into the SLE from dams.

Some of the more pressing water-associated issues that plague the GL-SL basin today (and for many, the Baltic Sea as well) include persistent organic chemicals and metals, eutrophication (which promotes hypoxia and algal blooms) due to fertilizer and sewage inputs (caused among other factors by stormwater runoff and sanitary sewer overflow), invasive exotic species and increased shipping. Shipping leads to the erosion of the shoreline of the St Lawrence Seaway (SLS), which on the midterm may cause the disappearance of fluvial Lake St. Pierre (through which the SLS travels), a shallow widening of the SLR declared a World Biosphere Reserve by UNESCO in 2000. In addition, shipping necessitates continuous dredging, which leads to resuspension of contaminants. Other issues are shoreline development (which contributes to wetland loss) and removal of water for drinking, manufacturing and irrigation. Among these dire issues, the loss of recreational areas such as beaches, while seemingly benign, is perhaps the most significant manisfestation of environmental degradation present in the daily life of the ordinary citizen. This effect is pernicious: even on an island like Montreal (surrounded by water), many of the 2 million citizens become progressively unaware of the existence of large, nearby potential recreational aquatic spaces. This disinterest explains the lack of pressure that should be put on local governments to stop or mitigate environmental degradation, in order to reopen these spaces to the public. Global climate changes complicate these issues; for instance, warmer water, in addition to containing less oxygen, results in a shorter ice cover. Together with stronger storms in the fall and winter, this increases the erosion of the SLE shoreline by waves. More intense precipitations, also predicted by climate models, may have contributed to recent algal blooms (*Alexandrium tamarense*) in the SLE by augmenting freshwater discharges. Increased evaporation leads to lower water level in the SLR, which resulted in loss of wet land in the fluvial Lake St Pierre in 2000-2001 (Vis et al., 2007).

In summary, many serious environmental issues are common to the Baltic Sea and the GL-SL because of the human, physical and ecological similarities that exist between these ecosystems. These similarities may call for common solutions.

Eutrophication

12

Jeffrey M. Levengood
University of Illinois at Urbana-Champaign, Urbana, IL, USA
Daniel Martineau
University of Montreal, Saint-Hyacinthe, QC, Canada

The availability of nitrogen (N) and phosphorus (P) limits the growth of aquatic plants and plant-like organisms (macroalgae and phytoplankton, including 'harmful algae'). In freshwater, P is the least abundant (and thus the limiting factor), which would explain why reducing P input alone in lakes could be sufficient to prevent algal blooms whereas in estuarine and marine coastal environments, it is N that is often the least abundant, and thus limiting, factor.

Before extensive urbanisation and agriculture, the Great Lakes (GL) were primarily oligotrophic – they received only small inputs of N and P from streams and rivers. As a result, GL productivity was largely confined to the extensive system of marshes and estuaries located at their periphery.

Nutrient inputs into the GL increased, albeit modestly, as early as the 18th century as European settlers initiated deforestation, which increased erosion. Increasing human populations and the concomitant industrialisation, agriculturalisation and inputs of sewage by communities established on the shoreline injected massive amounts of N and P into the GL. These nutrients led to the overgrowth of phytoplankton and macroalgae, a process termed eutrophication, which perhaps is the largest, most pervasive global water-quality issue (Diaz and Rosenberg, 2008).

Normally, solar energy drives the growth of phytoplankton upon which the food chain is based. Under normal conditions phytoplankton are produced – and die – in moderate amounts because of the limited availability of N and P. Dead phytoplankton fall to the bottom where benthic animals such as aquatic crustaceans, insects and annelids feed on them. In turn these animals transfer this energy to upper trophic levels. When N and P are available in enormous amounts, massive amounts of phytoplankton and macroalgae are produced, die and sink to the bottom where their decomposition by bacteria consumes the oxygen dissolved in water, leading to a total (anoxia) or partial (hypoxia) lack of oxygen. In turn, the absence of oxygen may devastate entire populations of aquatic animals, crustaceans being the most sensitive (Vaquer-Sunyer and Duarte, 2008). In addition, benthic animals are deprived of the solar energy that was previously transferred to them through phytoplankton falling to the bottom. Instead, solar energy is monopolised by the massive bacterial population thriving in the overabundant decomposing algae.

Nowadays, all but the northernmost lake, Superior, have grown progressively more eutrophic since the mid-19th century (Meyers, 2006). It seems that restricting P input alone is sufficient (and economically realistic) to mitigate eutrophication in lakes (in contrast to estuaries where the limitation of both N and P should be considered). In the 1970s, regulations limiting phosphorus inputs into the GLs such as a ban on phosphate-containing

Figure 12.1. A satellite image showing algal blooms in Lake Erie in August 2009. Note large bloom in the western portion of the basin. Image courtesy of NOAA Great Lakes Environmental Research Laboratory.

detergents and improving waste water treatment greatly reduced blooms of the green macroalgae *Cladophora* in Lakes Erie and Ontario, although a 'dead' zone still develops during late summer in Lake Erie, which is now regarded as mesotrophic with some highly eutrophic areas. In estuaries (such as the St Lawrence) and in the Baltic Sea in contrast, it is argued that restricting both P and N inputs is necessary to stop algal blooms (Conley et al., 2009; Schindler and Hecky, 2009).

Hypoxia occurs naturally in bodies of water in which water circulation is limited by slow flushing or stratification, the latter occurring when layers of water separate because of their different salt concentration (haloclines) or temperature (thermoclines). Thus, both the BS and GL-SL are naturally vulnerable to hypoxia caused by eutrophication because of slow flushing and the presence of thermoclines and haloclines (the latter for the SL and BS).

Deep waters of the SL Lower Estuary and Gulf (>275 m), and deep waters of the mouth of the Saguenay River into the SLE, at the centre of the beluga whale's habitat, up to Anticosti island (Gilbert et al., 2005) are eutrophied and/or hypoxic because, like the GL waters, increased N and P are brought by rivers which carry municipal sewage, fertilisers and manure. In addition, warmer water also probably contributes to hypoxia. Indeed over recent years, oxygen-poor warmer water has penetrated the SL Gulf deep waters from the Gulf Stream, possibly because of global warming. The low oxygen levels that have resulted (warm water contains less oxygen than cold water), are lethal to cod, and thus may have contributed to the severe decline in cod populations that has economically devastated entire fishing villages in Newfoundland and the Gaspe Peninsula (Quebec). Cod populations have also dwindled in the Baltic since the 1960s for the same reasons e.g. eutrophication and hypoxia, the latter being permanent in the Baltic deep waters (Conley et al., 2002; Österblom et al., 2007). Hypoxia also has sublethal effects on aquatic animals: under hypoxic conditions, some aquatic animals breathe faster and thus absorb more of some lipophilic contaminants. Certain contaminants that damage gills and red blood cells and/or interfere with glycolytic metabolism may hamper the adaptation of aquatic animals to hypoxia (see review by Couillard et al., 2008).

The 21st century has seen an increase in the number and severity of harmful algal blooms (HAB) worldwide

(Heisler et al., 2008), including in both the Baltic Sea (Vahtera et al., 2007) and the GL (NOAA, 2010). Most HAB are composed of plant-like, photosynthetic algae such as cyanobacteria (or blue-green algae), but some, such as *Pfiesteria* spp., are animal-like protozoans not equipped for photosynthesis. Most HAB have been associated to varying degrees with eutrophication (Anderson et al., 2002). In freshwater, high nutrient levels mostly promote the growth of nitrogen-fixing cyanobacteria and thus are responsible for most HAB in freshwater lakes. Some species of cyanobacteria can produce cyanotoxins (hepatotoxins, neurotoxins, cytotoxins, dermatotoxins and or irritant toxins) that are highly toxic to humans, pets, livestock and wildlife (Mehra et al., 2009). Cyanobacteria can also cause hypoxia on their own, and can disrupt food webs. In addition, by fixing N they make it available for other algae, thus countering the efforts made to reduce the input of external N, as has been documented in the Baltic Sea (Vahtera et al., 2007; Schindler et al., 2008). HAB have also increased in the GL (NOAA, 2010a), especially in the shallower Lake Erie, and Lake Huron's Saginaw Bay (Vanderploeg et al., 2001). Using satellite and aerial monitoring, the National Centres for Coastal Ocean Science and National Coastal Centre of the National Oceanic and Atmospheric Administration, along with external partners, detect and monitor HAB in coastal regions of the US, including the GL (Figure 12.1; NOAA 2010b). In addition, research in experimental HAB forecasting is underway, to allow prediction of the location and intensity of current and expected HAB.

Evidence has suggested that selective filtration and elimination by the exotic zebra mussels (*Dreissena polymorpha*) may be promoting blooms of the cyanobacteria *Microcystis aeruginosa*, which produces mycrocystins, potent hepatotoxins, in some areas of the GL, by giving unpalatable toxic strains of this species a competitive edge (Vanderploeg et al., 2001). Later, Raikow et al. (2004) determined that the presence of zebra mussels promoted *Microcystis* production in lakes with low nutrient levels, whereas cyanobacteria increased in eutrophied lakes in the absence of the mussels.

Alexandrium tamarense, a photosynthetic dinoflagellate, produces saxitoxin, a toxin causing paralytic shellfish poisoning (PSP). It is a well-known cause of recurrent toxic blooms in the St Lawrence Estuary (Blasco et al., 2003); these blooms are usually restricted to brackish water plumes formed by freshwater input from the upper estuary and rivers in the absence of wind. Both the absence of wind and freshwater diminish the turbulence of water, which in turn favour the growth of *A. tamarense* growth and toxin production. Humic substances and possibly P and N released into rivers after heavy rains would also increase growth of *A. tamarense* (Fauchot et al., 2005). In 2008, with weather conditions such as those described above, a bloom of *A. tamarense* formed in the St Lawrence Estuary, at the heart of the beluga whale habitat. It caused the death of hundreds of fish, birds, dozens of seals and 9 beluga whales (actual mortality was probably much higher), an event previously unheard of in the SL Estuary.

The anticipated effects of global warming on eutrophication are currently of great concern. Higher temperatures are known to favour algal blooms (and thus to increase consumption of oxygen by bacteria decomposing dead harmful algae). Because warmer water can contain less gas than cold water, increasing temperatures will worsen hypoxia induced by eutrophication. Finally, reduced precipitation expected at mid-latitudes under global warming scenarios will decrease the volume of water flowing into the GL, which will increase the retention time of nutrients, thus exacerbating eutrophication.

Contaminants in Great Lakes Environs

13

David P. Macarus*
United States Environmental Protection Agency, Chicago, IL, USA

The five Great Lakes touch eight states in the US from Minnesota East to New York and the Province of Ontario, Canada (see map, Figure 13.2). All but Lake Michigan are shared with Canada. A full description of the history and characterisation of the Great Lakes, including sizes, water depths and flow patterns, can be viewed at http://www.epa.gov/glnpo. The lakes provide hydroelectric energy, cooling water for coal, oil, gas and nuclear fuelled steam generated electricity and water for numerous industries including lumber, agriculture, fishing and shipping. Over 30 million people live in the Great Lakes Basin, and use the lakes for recreation and drinking water. Some of the beneficial uses of the lakes are compromised by the numerous contaminants affecting the air, water and land of the Great Lakes Basin including pathogens, nutrients, chemicals and metals with varying toxicity, and non-native species. The contaminants are introduced to the basin from industrial activities, municipal sewage overflows, private septic systems, pesticide applications, contaminated run-off from urban areas and cropland, both wild and domestic animal waste, ship ballast water harbouring alien species and through the atmosphere (see http://www.epa.gov/glnpo).

There are thousands of chemicals and biological organisms, both naturally occurring and those resulting from human activity, that abound in the environment. Biota and ecosystems have evolved the means to alleviate the physiological stress of exposure to contaminants that have existed since pre-historic times. However, some of the more recent introductions of chemical and biological stressors have disrupted the natural accommodation and caused ecosystem damage (polluted air and water resulting in retarded bird reproduction, fish kills and other disruptions of wildlife), eutrophication and hypoxia from excess nutrients, human respiratory ailments, systemic poisoning, cancer and many other impacts to life on earth.

Many contaminants result from activities that provide useful products and processes for humans and society. The benefits of automobiles, electrical equipment, functional metals and polymers, fuels, fluids, pest control, coatings, adhesives, economical food production, etc. come with some hazards. We recognise some of the hazards (toxicity during manufacture and use, fertiliser run-off to waters, pesticide food residues, air and water emissions, mechanical hazards, potential explosions, accidental releases), which can be mitigated through design and various regulations for delivery and use. However, later we may find unanticipated contamination of air, water, avian and aquatic life, soil from wastes, long-range

*Deceased

transport from distant sources, the long-term presence of persistent contaminants, unknown health effects of continuous exposure to multiple chemical contaminants, and loss of beneficial uses due to ecosystem damage.

The following sections discuss some of the known effects of the various classes of pollutants on air, water and land, and provide references for more complete data and analysis. In the end, humans are concerned that the air they breathe is free of harmful pollutants, the water is safe to drink, the beaches are safe for swimming, the fish and wildlife contain no toxic residues, and the land they are exposed to is free of poisonous substances. The following contaminant groups were selected for consideration here, but there are others (e.g. nanoparticles) that may deserve research and regulatory consideration:

1. Currently used pesticides and other toxic chemicals
2. Persistent, bioaccumulative and toxic substances (PBTs)
3. Trace metals
4. Pharmaceuticals and personal care products (PPCPs)
5. Airborne contaminants

Figure 13.1. Industrial Pollution at the Calumet River flowing into Lake Michigan in Illinois. Photo by the US Environmental Protection Agency - Great Lakes National Program Office.

Currently Used Pesticides and other Toxic Chemicals in the Great Lakes

As with any toxic substance, we seek to answer the following:

1. What are the sources of the toxins of concern?
2. What quantities are being released?
3. What is the environmental fate? i.e. where do the toxins go?
4. Exposure and hazards to humans, wildlife and ecosystems.

Agriculture continues to rely on pesticides for the control of weeds, insects and fungal diseases. The 2007 State of the Great Lakes Report on Pesticide Management quoted an estimate of 21 million kg of pesticides used annually on agricultural crops in the Great Lakes watershed based upon a 1993 GAO report (GAO, 1993). A revised estimate was made using the US Department of Agriculture's National Agricultural Statistics Service Database for pesticide use on major crops in the principal states bordering the Great Lakes (MI, NY, OH and WI). Comparing the pesticide usage in 1993/94 and 2005, several significant changes in pesticide use occurred over the 11-year period. Reductions occurred due to changes in cropping patterns, integrated pest management and the expanded use of genetically modified crops. Herbicides continued to dominate pesticide use, with atrazine remaining the leading herbicide, with little change in use. However, metolachlor and alachlor use declined in favour of acetochlor and glyphosate. Increased use of glyphosate-tolerant crops (principally soybean) over the 11-year period was a contributing factor. Overall, herbicide use in the basin was estimated to have declined by about 20% to a level of around 16-18 million kg per year. Agricultural insecticide use also declined by about 70%, with the use of plant incorporated protectants, such as Bt corn, playing an important role.

Pesticides applied to agricultural crops move through the environment in several ways. Some of the pesticides applied are absorbed by the plants with residues remaining in the food crop; some vaporise and are carried away in air currents; some are retained in the soil; some move through the soil to groundwater, and some are washed away and mixed into surface waters. Most of the currently used pesticides are not persistent to the extent exhibited by the former highly chlorinated pesticides such as DDT and chlordane. Hence, currently used pesticides retained in the soil can eventually degrade by the action of soil chemicals and microorganisms. However, pesticides entering groundwater or surface waters are little degraded because of low concentrations of microorganisms and lower temperatures.

An analysis of the pesticides found in the nation's waters has been carried out under the by the US Geological Survey's NAWQA Program (National Water-Quality Assessment Program). The NAWQA studies included two areas of the Great Lakes basin: Lake Erie-Lake Saint Clair drainages, and western Lake Michigan drainage. The samples were taken from streams, but did not include finished drinking water. Over 30 pesticides were detected, with the most heavily applied herbicides, atrazine, metolachlor, acetochlor, and alachlor, being most frequently detected in agricultural areas, and the insecticides diazinon, chlorpyrifos and malathion more frequently found in urban areas. While the time weighted average of the concentrations of each pesticide are below the individual regulatory MCL (Maximum contaminant levels) for drinking water (http://water.epa.gov/drink/contaminants/index.cfm#List), peak values of atrazine following application and rainfall reach concentrations well above the MCL in May or June.

The US EPA has set the following standards for protection of human health and aquatic life at the following values for atrazine:

Drinking MCL	3 µg/L
Aquatic life criteria for acute effects (proposed)	350 µg/L
Aquatic life criteria for chronic effects (proposed)	12 µg/L

Environment Canada studied the presence of pesticides in the waters of the four Great Lakes that are shared with the US (http://www.on.ec.gc.ca). Their findings were similar to those of the NAWQA study, with 33 pesticides being detected. As expected, the herbicides in greatest use were found to have the highest concentrations. Atrazine showed peak values of 1 ug/L, metolachlor 0.7 ug/L, and simazine 0.28 ug/L. Lake Erie had the highest concentrations, followed by Lakes Ontario, Huron and Superior. No pesticide was found to exceed the water quality criteria established for the pesticide. The US Environmental Protection Agency's (EPA) Mass Balance Study (http://www.epa.gov/greatlakes/lakemich/study.pdf) for Lake Michigan measured PCBs, mercury, trans-nonachlor and atrazine in rivers, air, sediments, lake water and the food chain. A mathematical model was developed to predict the effects that increases or decreases of pollution will have on the lake and its large fish (lake trout and coho salmon). The study measured the concentration of atrazine as a marker for currently used pesticides in waters of the lake. Open water concentrations ranged from 33.0 to 48.0 ng/L, with average values increasing from 37.0 to 39.7 ng/L over the one-year period of the study.

The EPA Office of Pesticide Programs (http://www.epa.gov/pesticides) registers pesticides using a system of risk assessment that is designed to ensure that exposure is minimal when used according to pesticide labels. New information regarding toxicity, exposure, or environmental damage can result in more restrictive use, as has occurred recently with chlorpyrifos and diazinon, for which residential uses were removed. Pesticides are widely used on crops producing food. Because such applications have the potential to result in high residues of pesticides in the food products, the EPA sets tolerance limits for pesticide residues in food under the authority of the Federal Food, Drug and Cosmetic Act. The US Department of Agriculture (USDA) monitors crops and foods for residual pesticides annually. The latest report for 2006 includes five of the eight Great Lakes states, and continues to report a safe food supply, with residues rarely found above the allowed tolerance levels set by EPA (Pesticide Data Program, USDA Agricultural Marketing Service).

Legacy Pesticides and Other Persistent Bioaccumulative and Toxic Chemicals (PBTs)

PBTs are persistent, bioaccumulative and toxic compounds that continue to have an environmental presence, even though some are no longer in production. The past usage of 'legacy pesticides' (such as DDT, chlordane, toxaphene, etc.) and PCBs was large enough to cause widespread environmental contamination during the

Figure 13.2. Toxic sediment - wood mill, Keene Creek, Duluth, Minnesota. Photo: Minnesota Department of Natural Resources, Pat Collins.

years of their production, and, due to the resistance of these contaminants to degradation, they continue to be found in the environment. Despite bans on 'open uses' in 1973 and production in 1979, PCBs still remain in electrical transformers, capacitors and other equipment. The US production of DDT, the first pesticide to be used on a large scale, is reported to have reached a peak of 80-85 million kg in 1962. Polycyclic aromatic hydrocarbons (PAHs) are another important persistent and bioaccumulative group of concern. PAHs are formed by the incomplete combustion of organic matter, and occur both naturally (e.g. volcanoes, forest fires) and through anthropogenic activities such as combustion processes in industry, power plants, home heating, waste incineration (including rural backyard burning), etc.

To monitor the PBTs in air and precipitation, Canada and the US established the Integrated Atmospheric Deposition Network in 1990 to cover the Great Lakes region (http://www.ec.gc.ca/natchem/default.asp?lang=en&n=1590DD07-1). The network consists of seven major stations and several satellite stations. The latest IADN report covers the period up to 2003, and presents long-term temporal and spatial trends of atmospheric toxic substances in the Great Lakes Basin. Most contaminant groups (HCB, PCBs, organochlorine pesticides and PAHs) showed decreasing trends over time, with half-lives from 4 to over 15 years. Spatial analysis showed a high urban influence, with most contaminants increasing in the environment along with local human populations.

In another study, using an analysis of sediment cores from 38 lakes across the US to develop a historical record of persistent contaminant deposition, Van Metre and Mahler (2005) found that DDT (including metabolites) and PCBs have declined since production ceased in the 1970s, as expected, while PAHs have trended upward, with growing urban areas showing larger increases. These compounds are toxic to benthic biota, and PAH concentrations in sediments are increasingly rising above concentrations where adverse effects on benthic biota are likely.

Under the Clean Air Act (CAA), the EPA has listed 190 air toxics for consideration, and has selected 33 from a published list of 70 industrial sources for regulation of emissions (http://www.epa.gov/air/caa/). Six pollutants, called criteria pollutants, are used as indicators of air quality. By 2006, these pollutants (carbon monoxide, ozone, lead, nitrogen dioxide, fine particulate matter and sulphur dioxide), had declined by 49% from the base year of 1980. The Great Waters Program, as part of the 1990 amendments to the CAA, was put in place to determine whether the current provisions of the CAA are sufficient to prevent atmospheric deposition of pollutants to the Great Lakes. The Great Waters Program found that the bioaccumulative pollutants that move up in the food chain, as noted in the PBT, Critical Contaminants and Fish Advisory sections (http://www.epa.gov/ttn/atw/area/list33.html; http://www.epa.gov/air/airtrends/sixpoll.html), play an important role in human and ecological health.

Radon is a naturally occurring product of uranium decay, and is estimated to cause 20,000 lung cancer deaths annually (http://www.epa.gov/radon/). Although not regulated, EPA and state partners have engaged in a nationwide campaign to inform citizens of the hazards of radon and instructions on testing their homes.

While the concentrations of the PBTs found in waters are generally below drinking water standards, they bioaccumulate in aquatic life to levels of concern such that fish consumption advisories are published by the Great Lakes states. Organised international efforts to reduce the presence of many of these PBTs include the Great Lakes Binational Toxic Strategy (BTS) between Canada, and the US, and the international agreement to control 12 Persistent Organic Pollutants (POPs) (http://www.pops.int/). The substances included in the BTS and POPs lists are shown in Table 13.1, along with the

'Critical Contaminants' identified by the Lakewide Area Management Programs (LaMPs) for the Great Lakes. The dates of discontinuance of the legacy pesticides can be found in the Pesticide Report of the BTS (http://www.epa.gov/glnpo).

Mercury is a critical contaminant and a major source of fish consumption advisories (http://www.epa.gov/glnpo; http://www.epa.gov/waterscience/models/maps). Mercury is transported through the atmosphere to the earth's surface, where conversion to methyl mercury occurs in areas where certain anaerobic reducing conditions exist. It is the methylated form that is bioaccumulative in aquatic life and that leads to higher concentrations in fish. Coal combustion is an important source of mercury, but waste incineration, batteries, electrical switches, chlor-alkali production, thermometers, thermostats, medical equipment, dental amalgam and preservatives are additional uses that can have emissions to the environment.

Critical Contaminants for the Great Lakes

Of the thousands of potential contaminants, 12 have been identified by the programmes to protect the Great Lakes as critical on the basis of their toxicity, presence in the environment, and bioavailability. These critical contaminants are shown in Table 13.1 with sources and health concerns. Substances included in the BTS and the international agreement for POPs are listed for comparison. Table 13.1. Critical contaminants (condensed from http://www.great-lakes.net/humanhealth/fish/critical.html). Level I substances under the BTS are targeted for virtual elimination. Contaminants indicated as 'L2' are Level II substances listed for voluntary pollution prevention activities.

Over time, pollutants discharged to waters entering the Great Lakes are absorbed by particulate matter suspended in the waters. Such particles settle to the bottom of lakes, rivers and harbours and hold the contaminants in the bottom sediments until disturbed. Because of the need to dredge shipping channels and harbours, the toxicants bound to the sediments are re-introduced into the water column. As the result, there are 43 Areas of Concern (AOCs) in the Great Lakes basin, where approximately 200 contaminants have been detected in sediments. Concentrations of toxicants bound by the sediments are typically higher than concentrations found in the waters above. To address the problem of these highly contaminated areas, the US Congress has created 'The Great Lakes Legacy Act', which provides funds for the restoration of the AOCs (http://www.epa.gov/glnpo/sediment/legacy).

Fish Consumption Limits

Recommended limits on fish consumption may be necessary due to the bioaccumulation of contaminants through the food chain. Contaminants with water concentrations of just nanograms per litre can accumulate and magnify a million fold to concentrations of parts per million in fish. This is the case for toxaphene in Lake Superior where recommended fish consumption limits have been issued by the Province of Ontario, Canada. In the US, fish consumption limits for the ingestion of sport fish are produced by each state. The US EPA Office of Water (http://www.epa.gov/waterscience/) produces The National Listing of Fish Advisories (NLFA), based upon data from States and Tribes. Each state provides recommended limits for specific waters, and includes information on the species, size of fish, specific sub-populations (i.e. pregnant women), and recommended consumption rate. Five contaminants account for about 90% of the 2006 fish Advisories: mercury, PCBs, chlordane, dioxins, and DDT and its metabolites (DDE and DDD). While these substances are declining in the environment, they continue to be detected in fish, indicating that re-entry from contaminated sediments is likely still occurring (http://iisgcp.org).

Contaminants of Emerging Concern

Table 13.1 shows a list of the contaminants of current concern in the Great Lakes basin, the Great Lakes Binational Toxics Strategy and International list of 12 Persistent Organic Pollutants (POPs). However, in addition to these well recognised pollutants, there are other chemicals increasingly present in the environment**.**

Table 13.1. Critical contaminants (condensed from http://www.great-lakes.net/humanhealth/fish/critical.html). Level I substances under the BTS are targeted for virtual elimination. Contaminants indicated as 'L2' are Level II substances listed for voluntary pollution prevention activities.

Critical Contaminant (CC)	BTS List	POPs List	Uses and Sources	Health Concerns
Chlordane & (CC) Heptachlor	Yes L2	Yes Yes	Insect control for crops and building termite treatment. All uses cancelled by 1987.	Probable human carcinogen. Detected in Great Lakes waters, fish and wildlife.
DDT and (CC) Metabolites	Yes	Yes	Large use as crop insecticide and mosquito abatement. Most uses cancelled in 1973.	Cause of diminished reproduction of eagles and other wild life.
Aldrin & (CC) Dieldrin	Yes Yes	Yes Yes	Uses of both insecticides now cancelled. Aldrin converts to dieldrin in environment.	Dieldrin concentrations in fish exceed GLI criteria in many areas of the basin.
Dioxins & (CC) Furans	Yes Yes	Yes Yes	Inadvertent by-product of incin-eration, pulp & paper bleaching and chemical manufacturing	Poorly understood, but wildlife sensitive at low levels in laboratory studies.
Mercury and (CC) Methyl mercury	Yes	No	Released in combustion of coal. Use in batteries, electrical components and dental amalgam declining.	Toxic to human and animal foetuses. Fish consumption limits due to bioaccumulation of methyl mercury in fish.
Lead, nickel, copper, zinc and cadmium (CC)	alkyl Lead	No	Common in hazardous waste. Degradation of benthos and plankton in Lake Huron due to sediment concentrations.	Organ damage at low concentrations. Accumulate in food chain.
Mirex (CC)	Yes	Yes	Pesticide and flame retardant now cancelled in North America. Found in Niagara River and Lake Ontario.	Fish consumption limits for Lake Ontario due to exceedance of GLI criteria.
Polychlorinated biphenyls (PCBs) (CC)	Yes	Yes	Use began in 1929 and dis-continued in 1978. Used in electrical transformers, capacitors and switches.	Fish advisories due to bio-accumulation in the food chain. Health and reproduction problems in eagles, mink, etc/
Polybrominated biphenyls (PBBs) (CC)	No	No	Manufactured in Michigan as flame retardant. Plant closed and site being remediated.	Food chain contamination in 1973 from accidental mixing of PBBs with animal feed.
Nutrients: nitrogen & phosphorous (CC)	No	No	Discharged by wastewater treatment and run-off from urban and agriculture areas.	Not completely understood, but cause eutrophication in waters and hypoxia in Gulf of Mexico.
Sediments and suspended solids (CC)	No	No	Deposited in water channels from urban and agricultural run-off and stream bank erosion.	Carries pollutants downstream and covers fish spawning and aquatic life habitat.
Tritium (CC)	No	No	A radioactive by-product from water cooled nuclear reactors.	Exposure and health effects not well known.
Poly Aromatic Hydrocarbons (PAHs)	Yes	No	Unintended by-product of combustion and other high temperature processes	Benthic organisms
Toxaphene	Yes	Yes	Cancelled pesticide. Moves to GL via air-borne transport from legacy sites.	Bioaccumulative in aquatic life. Fish consumption limits for Lake Superior
Endrin	L2	Yes	Cancelled pesticide – isomer of dieldrin	See dieldrin
Hexachlorobenzene	Yes	Yes	Formerly used as fungicide. By-product of incineration.	Probable carcinogen. Produces many systemic symptoms.

Polybrominated biphenyl ethers (PBDEs) are increasingly found in humans and the environment. About 75 million pounds of PBDEs are used in the US annually as flame retardants in foams, plastics, fabrics, computer cases and circuit boards (http://www.epa.gov/fishadvisories/forum/2004/proceedings.pdf). PBDEs were first detected in the US in 1979 and in Sweden in 1981 (Alaee and Wenning, 2002). In further studies, PBDEs were found in fish-eating birds and marine mammals in the Baltic Sea, North Sea and Arctic Ocean, and in marine fish, shellfish and sediments in the Pacific region. PBDEs have now been detected in all parts of the US and Canada, including the Great Lakes. Being structurally similar to PCBs, the PBDEs have 209 possible congeners, of which the tetra-, penta- and hexa- brominated congeners are most often found in waters, wildlife and humans. In the atmosphere, PBDEs are found above the waters from 5 pg/m^3 at Lake Superior to about 52 pg/m^3 over Chicago (Strandberg et al., 2001). Partitioning of the congeners between gas and particles was noted, with about 80% of the tetrabrominated homologues being found in the gas phase, and about 70% of the hexabrominated homologues in the particle phase. Partitioning of the congeners was also noted in Great Lakes fish and Lake Michigan water (Streets et al., 2006). This higher absorption of the more highly brominated congeners to particles was noted in sediment cores as well, with decabromo (BDE-209) making up 95-99% of the PBDE load (Zhu et al., 2005). Bioaccumulation of PBDEs in fish was reported by the Wisconsin Sea Grant Program. An average level of 80 ppb in Lake Michigan salmon was measured in 1996, which is about six times higher than the levels reported for Baltic Sea salmon in 1999. Fish archived during 1980 to 2000, analysed for 15 PBDE congeners, were found to have PBDE concentrations increasing rapidly, with 3-4-fold annual increases (Zhu and Hites, 2004). By the year 2000, total PBDE concentrations in Lake Michigan lake trout were the highest at about 1,400 ng/g of lipid, followed by Lakes Superior (990 ng/g), Erie (600 ng/g), Ontario (550 ng/g) and Huron (370 ng/g).

As an indicator of potential human health effects, Dye et al. (2007) reported that serum PBDE concentrations of 23 cats ranged from 4.3 to 12.7 ng/ml; cats exhibiting hyperthyroidism had higher concentrations than young control cats. Analysis of cat food showed PBDEs were present in both dry and canned foods, with dry food having higher concentrations, on average. The presence in humans has been rising, with Johnson-Restrepo et al. (2005) reporting a median level of 77.3 ng/g of lipid fat in a group of 52 people. The highest values were 9630 and 4060 ng/g, which are higher than any previously reported and some initial data show that 10% of California residents have higher PBDE concentrations in tissue than PCBs. Rodent studies have shown adverse neurological effects of PBDE exposure, prompting current studies in humans (http://cfpub.epa.gov/ncer_abstracts/index.cfm/fuseaction/display.abstractDetail/abstract/8044/report/2008).

Perfluorinated compounds such as perfluorooctanoic acid (PFOA), perfluorooctane sulphonate (PFOS), and many other perfluorinated compounds of various carbon chain lengths and functional end groups are ubiquitous in the environment. These contaminants are released from the manufacturing and breakdown of products used for non-stick surfaces, stain repellents, fire fighting foams, surfactants and numerous other beneficial uses. The compounds resist degradation, are persistent in the environment and are found in water, humans and wildlife (Herbert et al., 2002). Perfluorinated compounds have been detected in wastewater treatment effluent in New York, Kentucky, Georgia, in drainage basins of North Carolina, Ariake Sea, Japan, in fishes, birds, benthic organisms and in the general human populations in many countries. (e.g., Nakayama et al., 2007; Lau et al., 2007; Sinclair et al., 2006; Moody et al., 2001; Taniyasu et al., 2003).

Perfluorinated alkylated compounds were detected in 89% of the water samples collected in waters of Lakes Michigan, Huron and Superior, with concentrations as high as 36 ng/L (Kannan et al., 2005). Concentrations of PFOS in the livers and muscle of chinook salmon, lake whitefish, brown trout and carp were 1,000 times or more higher than in water. Values in Lake Michigan chinook salmon ranged from 32 to 173 ng/g wet weight in livers, and up to 189 ng/g in muscle. Similar values were observed in Lake Huron, with the exception of Saginaw Bay carp, which had concentrations up to 297 ng/g. Lake Superior brown trout had lower values of up to 26 ng/g in liver, 46 ng/g in muscle and 75 ng/g in eggs. In other areas of the Great Lakes basin, perfluorinated compounds were found in livers of 10 species of waterfowl in the Niagara River, and elevated concentrations of PFOS were reported in the surface waters of Lake Onondaga (Sinclair et al., 2006).

Studies in rodents have shown dose-dependent responses in maternal and developmental toxicity of PFOA, such as early pregnancy loss, reduced postnatal survival, delays in growth and development, and sex-specific alterations in pubertal maturation (Lau et al., 2006). However, the long-term effects of exposure to perfluorinated organic compounds are largely unknown, prompting the US EPA, under the Toxic Substances Control Act (TSCA), to add 50 perfluorinated chemicals that have the potential to persist and bioconcentrate to the TSCA reporting requirement; this action solicits information on uses, exposures, ecological effects, environmental fate and human health effects.

In addition to the brominated flame retardants discussed earlier, there are other flame retardants that have an environmental presence, even though they have not

Figure 13.3. Lake Michigan Beach. Elberta, MI. Michigan Travel Bureau photo available from the Environmental Protection Agency - Great Lakes National Program Office.

reached the critical contaminant lists. Notable are the short- and medium-chain chlorinated paraffins and declorane plus. Although the chlorinated paraffins can only be metabolised slowly in rainbow trout, bioaccumulation still occurs, with concentrations reported to be over 100 ng/g wet wt in lake trout from Lake Ontario (Muir, 2006). Dechlorane plus, formed from hexachlorocyclopentadiene, is persistent, with a half-life of over 182 days in soil. However, it does not bioaccumulate and no toxicity data are available to rank the compound's hazards.

Bisphenol-A (BPA) is a key monomer in the formation of polycarbonate resins. Polycarbonate resins are widely used for shatter-proof bottles, including baby bottles, and as coatings to line containers such as liquid infant formula cans. It has been found that BPA leaches from the bottles and coatings, and is now found in the tissues of 90% of Americans. The National Institute of Environmental Health Sciences' National Toxicology Program Expert Panel concluded that 'The scientific evidence that supports a conclusion of some concern for exposures in fetuses, infants, and children comes from a number of laboratory animal studies reporting that 'low' level exposure to bisphenol A during development can cause changes in behavior and the brain, prostate gland, mammary gland, and the age at which females attain puberty' (http://ntp.niehs.nih.gov/ntp/ohat/bisphenol/bisphenol.pdf). Much research is ongoing and at the time of publication local, state or federal bans or restrictions on biphenol A use have been proposed or enacted. Millions of pounds of phthalate esters are used as vinyl plasticisers and solvents. They are a concern because some of the phthalate esters are emitted to the environment as by-products of manufacturing and use or breakdown of end-products. These esters are not currently found in all waters, and studies of four phthalate esters (dimethyl, diethyl, di-n-butyl, and butylbenzyl phthalate) have shown that concentrations are generally well below the predicted no-effect concentrations for aquatic life.

Samples collected during 1992 and 1993 showed that low concentrations of trace metals were widespread throughout the Great Lakes ecosystem (Nriagu et al., 1995; http://www.cprm.gov.br/pgagem/Manuscripts/pirronenatmospheric.htm). Concentrations of trace metals in Great Lakes water were generally below the drinking water MCL but much higher concentrations were found in diving duck livers, fish and zebra mussels (Table 13.2). Many of the metals are rapidly taken up by the suspended particulate matter; while concentrations vary by location, no systematic increase in concentration of the metals from Lake Superior to Lake Ontario was found. However, higher concentrations were observed in nearshore areas close to urban centres and river mouths. As to the fate of the metals entering the lakes, large amounts of dissolved copper, nickel and chromium exit the system through the St. Lawrence River, while the loadings of cadmium, lead and zinc appear to be retained in the basin.

Based on data collected via the Integrated Atmospheric Deposition Network (IADN) (http://www.ec.gc.ca/rs%2Dmn/default.asp?lang=En&n=BFE9D3A3-1), atmospheric deposition of trace metals was found to be an important source to the Great Lakes. Total deposition of most trace metals to the Great Lakes declined over the period 1976 to 1993-94. A very large decline for lead was likely due to the phase-out of alkyl lead in automotive gasoline. There was a considerable industrial-urban effect, with concentrations at NW Indiana during 1975-1980 being many times higher than the 1976 open-water values from Lake Michigan. Furthermore, deposition rates were higher for Lake Erie than for Lakes Michigan and Superior.

In another study, total atmospheric emissions of trace metals to the Great Lakes region increased until 1988-89, after which most showed little change except nickel, which

Table 13.2. Comparison of trace elements in Great Lakes water, diving ducks and mussels.

Element	Year taken	Where sampled	Water column conc. Mg/l	Drink-water MCL(4) mg/l	Animal conc. dry wt. µg/g	Animal tissue analysed
Ag				* 0.030		
Al	1993	S. Grt Lks		*0 .05-0.2	1,700	STZM(5)
As				0.010	0.55	DDL(1)
B	1993	S. Grt Lks S.Grt Lks			ND(2) 7.0	DDL STZM
Ba	1993	S. Grt Lks S. Grt Lks		2	ND 16.8	DDL STZM
Be	1993	S. Grt Lks		0.004	ND	DDL
Cd	1994 1993 1993	Great Lakes S. Grt Lks S. Grt Lks	0.0028-0.0045	0.005	 1.63 2.27	 DDL STZM
Cr				0.1	1.59	
Cu	1993 1993	S. Grt Lks S. Grt Lks		# 1.3	59.6 14.6	DDL STZM
Fe	1993	S. Grt Lks		* 0.3	2,050	DDL
Hg				0.002	0.18	DDL
Mg	1993	S. Grt Lks S. Grt Lks			659(3) 971	DDL STZM
Mn				* 0.05	16.0	DDL
Mo					2.7	DDL
Ni					8.62	STZM
Pb	1994 1993 1993	Great Lakes S. Grt Lks S. Grt Lks	0.0032-0.011	#0.015	 ND 0.82	 DDL STZM(5)
Sb				0.006		
Se				0.05	10-33	DDL
Sr					0.59	DDL
V	1993 1993	S. Grt Lks S. Grt Lks			LDQ(6) 4.59	DDL STZM
Zn	1994 1993	Great Lakes S. Grt Lks	0.087-0.277	*5.0	 104	 DDL

* Non-enforceable guidelines that may cause cosmetic or aesthetic effects (taste, colour, odour, etc.)
# Action levels for copper and lead that trigger the need for additional treatment.
1 Diving duck liver analysed in four species of migrating diving ducks sampled at migratory corridors in Western Lake Erie, Lake St. Clair, and two locations of Lake Michigan.
2 ND = Non Detect, i.e, below analytical detection limit.
3 Males, females had lower concentrations at 564 µg/g.
4 Maximum Contaminant Level for Primary Drinking Water Standards. See reference 3
5 STZM = Soft tissue of Zebra mussel
6 Detected at less than quantifiable level
7 Data are from Custer and Custer, 2000, except where otherwise noted as from Nriagu et al., 1995

declined (http://www.cprm.gov.br/pgagem/Manuscripts/pirronenatmospheric.htm). Again, emissions were higher near industrial and urban areas. Similarly, the analysis of juvenile fish in Lake Ontario showed the urban-industrial effect, with median concentrations of copper and zinc at Toronto area tributaries exceeding the concentrations found in other areas of the lake. A study of the blood and urine of 32 sport fish consumers in Lakes Michigan, Huron and Erie did not find most trace metals, with the exception of lead and mercury, to be elevated (Anderson et al., 1998) compared with a group outside the Great Lakes basin. Mean blood lead and mercury concentrations were consistently higher than the mean from the comparison group, suggesting contributions due to the bioaccumulation of these metals in fish. However, the levels were below the NCEH/CDC reference range (<30 µg/L).

Pharmaceuticals and personal care products (PPCPs) are a group of contaminants that include prescription and over the counter therapeutic drugs, veterinary drugs, fragrances, cosmetics, sunscreens, vitamins and antimicrobial agents. These substances mostly enter the waters through municipal wastewater treatment effluent, private septic systems, or animal waste run-off. Wastewater treatment does not remove all of the PPCPs, as degradation rates vary among the products. As the human population increases, so does the use of the thousands of chemicals providing beneficial uses, while the supply of water remains the same or declines. A US study in 1999-2000 detected 82 substances at low concentrations in the nation's waters (Kolpin et al, 2002; Daughton et al., 2005). The study sampled a network of 139 streams across 30 states that emphasised waters downstream of urban areas and livestock production and waste disposal (including the application to cropland) facilities. Table 13.3 shows the groups of chemicals and the compounds within these groups that were most frequently detected. Detection of many of these compounds resulted from the improved analytical methods, which greatly reduced the detection limits. While each of the compounds was detected in low concentrations, and generally below the health or aquatic criteria, long-term toxicity and the unknown effects of mixtures of 20 or more of such compounds in a given water supply are a concern. There are several hormones entering the waters, and some have been shown to exhibit endocrine disruption at levels as low as 1 ng/L (Ternes et al., 2004), which again raises the question of the cumulative effect of the large number of substances found in the waters.

Safety of Drinking Water and Beaches in the Great Lakes

For most communities consuming Great Lakes water, the toxic chemicals present in the water are at levels below drinking water standards. Some of the tributaries to the Great Lakes that receive wastes from industrial areas such as Lake Onondaga, New York are polluted to an extent requiring restoration (http://www.epa.gov/Region2/water/lakes/onondaga.htm). Community water supplies work with a network of government agencies to ensure the safety of public drinking water supplies (http://www.epa.gov/safewater/contaminants/index.html). Under the Safe Drinking Water Act (1974 and amended in 1986 and 1996) the USEPA sets drinking water Maximum Contaminant Levels (MCLs) and health advisory levels (HALs) for 14 inorganic chemicals including seven metals, over 50 organic chemicals, four radionuclides and six microorganisms. In addition, suggested standards for 15 secondary pollutants are listed, but not regulated. The levels are based upon toxicity information with risks estimated for human lifetime exposure. Monitoring is carried out by states, tribes, public water suppliers, and the federal government. Additional monitoring is performed by the United States Geological Survey (USGS). USGS monitors the quality of the nation's waters under the National Water-Quality Assessment (NAWQA) Program, and reports the concentrations of contaminants found within several categories. (http://water.usgs.gov/nawqa).

Surveillance monitoring of the open waters of the Great Lakes conducted by the United States and Canada report high water quality (http://www.great-lakes.net/humanhealth; Dreelin, 2005). However, nearshore waters that are affected by discharges from wastewater treatment plants, septic systems, combined sewer overflows, and run-off from agricultural and urban sources are subject to microbial contamination. The treatment rules (http://www.epa.gov/safewater/contaminants/index.html) for drinking water supplied from surface waters and groundwater require filtration and disinfection to remove several micro-

Table 13.3. Most Frequently Detected Chemicals and Groups in U.S. Streams (Kolpin et al. 2002).

Contaminant group/ approximate det freq. (% of samples)	Most detected chemical in group	Maximum concentration µg/L	Median concentration µg/L	MCL (M) HAL (H) aquatic crit(A) µg/L
Steroids (84)	coprostanol cholesterol	150 60	0.088 0.83	None None
Non-Presc Drugs (81)	acetaminophen ibuprofen caffeine	10 1.0 6.0	0.11 0.20 0.081	6,000 (A) None 40,000 (A)
Insect Repellent (72)	N,N-diethyltol-uamide	1.1	0.06	71,250 (A)
Detergent Metab (70)	4-nonyl phenol + metabolites	72 (Sum of NP + 4 Met)	3.1	13 (A for NP)
Disinfectants (65)	triclosan	2.3	0.14	180 (A)
Plasticisers (62)	bisphenol-A	12	0.14	3600 (A)
Fire Retardants (60)	tri(2chloroethyl) phosphate	0.54	0.1	66,000 (A)
Antibiotics (49)	chlorotetra-cyline + Metab erythromycin+M	0.69 1.7	0.42 0.1	88,000 (A) 665,000 (A)

organisms including cryptosporidium, giardia, lamblia, legionella and viruses. Disinfection is most commonly performed by chlorination. While drinking water supplies are in general safe, an incident of cryptosporidium contaminated drinking water in 1993, believed to be caused by animal waste runoff to drinking water supply areas of Lake Michigan, resulted in severe distress and illness in over 400,000 people in Milwaukee, Wisconsin, USA.

Disinfection by-products (DBPs), such as chloroform, trichloroethylene, and other halogenated compounds are present in drinking water as the result of the addition of chlorine or bromine to control bacteria and viruses, reacting with trace organic substances present from human processes and the breakdown of plant and soil substances. The use of chlorine has virtually eliminated water-borne diseases such as typhoid fever, cholera and dysentery, but the unanticipated creation of DBPs is a concern still under investigation. USEPA has not been able to link the exposure to low concentrations of DBPs in drinking water with health risks that are known for high concentrations of the substances. Still, under the Safe Drinking Water Act, USEPA regulates four DBPs: bromates, chlorites, haloacetic acids and trihalomethanes (http://www.drinktap.org/consumerdnn/Home/WaterInformation/WaterQuality/DisinfectionByproducts/DBPsFactSheet/tabid/189/Default.aspx).

There are thousands of acres of fine beaches on the Great Lakes. As noted above, microbial contamination of nearshore waters from waste treatment, septic systems, sewer overflows, and run-off from urban and agricultural areas does affect beach safety. Microorganisms cause beach closures to protect swimmers, using *E. coli* as a health marker for beach safety in communities that monitor beaches. Action levels are not uniform, but range from 100 organisms/100 mL for the Province of Ontario, Canada to 200 organisms/100 mL for the State of New York. In 2004, 13% of the Great Lakes beaches were closed at least 10% of the time due to contamination by pathogens. The closing orders lag behind the sample collection, as current testing methods require a day or more to provide results. More rapid tests for pathogen contamination are being developed in order to protect swimmers closer to real time. See http://www.glin.net/beachcast for health advisories and monitoring data for beaches.

Conclusions

In conclusion, many contaminants are found in the Great Lakes Basin, although not all are governed by drinking water standards or aquatic life criteria. For the contaminants that have drinking water standards, concentrations rarely exceed such standards, except for pathogens in nearshore areas, which require the filtration and disinfection of drinking water supplies. However, numerous wildlife species, particularly invertebrates and fish, are affected by even the low concentrations of the bio-accumulative substances present in the Great Lakes (critical contaminants). The bioaccumulation of such contaminants in fish requires Canada and the Great Lakes states to issue recommendations on fish consumption limits. Many contaminants are preferentially absorbed by particulate matter in the water, only to be deposited as contaminated sediments in stream and river beds, harbours and other connections to the Great Lakes. Because of urban, industrial and agricultural practices, chemical contaminants, eroded soils and nutrients continue to add materials to contaminated sediments. Storm run-off and sewer overflows from urban areas and agricultural run-off carry pathogens to nearshore areas that result in periodic beach closures. Still, many open questions remain, such as the cumulative effect of the sum of the pollutants present at any time. Nutrients, discussed in another Chapter, also contribute negative impacts to the Great Lakes such as algal blooms producing mycrosytin toxins and causing hypoxia in embayments and other slow moving waterways.

Undoubtedly there are other toxic substances that are at concentrations below detection limits. As analytical methods continue to improve, more contaminants are expected to be found. Under the Safe Drinking Water Act, EPA considers other candidates for regulation. At present, there are 42 chemicals and 9 microbial contaminants under consideration as 'candidates' for regulation (http://www.epa.gov/safewater/contaminants/index.html). The vigilant surveillance of waters, sediments, terrestrial and aquatic wildlife by the many Great Lakes Agencies and Organisations in both Canada and the United States must continue to provide data to judge the health of the Great Lakes system.

Contaminants in Colonial Waterbirds of the North American Great Lakes, 1955-2007

14

CASE STUDY
Canada

D.V. Chip Weseloh
Canadian Wildlife Service, Environment Canada, Toronto, ON, Canada

The Great Lakes of North America are comprised of five large individual lakes and are the largest body of fresh water in the world. They encompass an area equal in size to the United Kingdom. In addition to all that water, there are also approximately 35,000 islands in the Great Lakes (http://en.wikipedia.org/wiki/Islands_of_the_Great_Lakes). Although many of the islands have been developed and built upon, many have not and those islands are home to hundreds of thousands of colonially nesting waterbirds, birds such as cormorants, pelicans, herons, egrets, gulls and terns. In 2000, at the end of a three-year survey to count all nests of colonial waterbirds on the Great Lakes, it was estimated that there were over one million nests of 13 species of colonial waterbirds on the Great Lakes, i.e. more than two million breeding individuals (Cuthbert et al., 2001; Morris et al., 2003, 2007; Weseloh et al., 2003).

Figure 14.1. Nesting Herring gull (*Larus argentatus*). Photo: Author.

Unfortunately, the Great Lakes have not always been a healthy environment for these birds to breed. In the early-mid 1960s, a team of researchers from the University of Wisconsin, led by Dr. Joe Hickey, found that Herring Gulls (*Larus argentatus*) nesting in Lake Michigan, one of the Great Lakes, were highly contaminated with various kinds of toxic chemicals, notably DDT, PCBs and dieldrin. They also found that these gulls were suffering extensive reproductive failure and their populations were probably declining (Keith, 1966). Further research on the eggs of Great Lakes birds in various museums found that one of the effects from toxic chemicals, e.g. thinning of the birds' eggshells, had been present in Great Lakes birds since about 1955 (Hickey and Anderson, 1968).

Research on the Canadian side of the Great Lakes in the early 1970s, led by Environment Canada's Canadian Wildlife Service (CWS), found similar problems and chemicals in gulls, common terns (*Sterna hirundo*), black-crowned night-herons (*Nycticorax nycticorax*) and double-crested cormorants (*Phalacrocorax auritus*) on the other four Great Lakes (Gilbertson, 1974, 1975; Gilbertson and Hale, 1974a, b; Price, 1977; Teeple, 1977; Gilman et al., 1977).

This early 'preliminary' research into contaminants in fish-eating colonial waterbirds on the Great Lakes led to the establishment, by CWS, of a Great Lakes-wide Herring Gull Egg Contaminants Monitoring Program (GLHGECMP) in 1974, which has continued to this day (Mineau et al., 1984; Hebert et al., 1999; Weseloh et al., 2006). In addition to monitoring contaminant levels, several research studies into the effects of toxic chemicals on fish-eating birds, mainly herring gulls, were carried out by staff at the CWS National Wildlife Research Centre in Ottawa, Ontario. The purpose of this paper is to present a brief summary of the contaminant work done on colonial waterbirds on the Great Lakes by the Canadian Wildlife Service, as well as other studies which have evolved from this work.

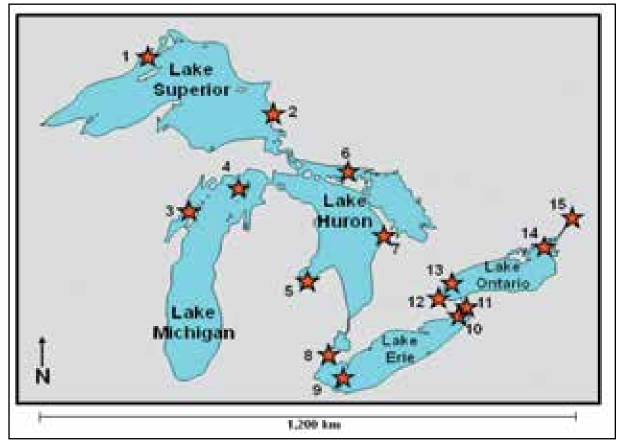


Figure 14.2. The 15 herring gull colonies where eggs are collected and analysed annually: 1. Granite I., 2. Agawa Rks., 3. Big Sister I., 4. Gull I., 5. Channel-Shelter I., 6. Double I., 7. Chantry I. 8. Fighting I., 9. Middle I., 10., Port Colborne Breakwater, 11. Unnamed island in the Niagara River, 12 Hamilton Harbour, 13. Tommy Thompson Park (Toronto Harbour), 14 Snake I., 15 Strachan I. Source: D. Moore, Canadian Wildlife Service, Burlington, Ontario.

Trends in Contaminants in Great Lakes Herring Gull Eggs, 1974-2007

The Great Lakes Herring Gull Egg Contaminants Monitoring Program began in 1974 after a three-year period of general study of contaminants in several species of colonial waterbirds (Gilbertson, 1975). Except for 1976, the programme has been conducted annually every year since and is now in its 35th year! The herring gull was chosen as the best available indicator species because it:

1. Bred on all five of the Great Lakes, making inter-lake comparisons very easy,
2. Was relatively common, so sampling its eggs would not jeopardise the population in any way,
3. Eats a diet consisting mainly of fish, which made it a top predator in the Great Lakes food web,
4. Produces eggs with relatively high lipid (fat) content, which is where most contaminants are stored in a bird's body and, hence, they would be relatively easy to measure, and
5. Appeared to be a relatively robust indicator species, meaning it could withstand reasonably elevated concentrations of many toxic chemicals without being extirpated from the Great Lakes.

If a species is very sensitive to contaminants, it can easily die out at elevated concentrations and no longer be available as an indicator, such as happened with both the peregrine falcon (*Falco peregrinus*) and the bald eagle (*Haliaeetus leucocephalus*). However, the best feature of the herring gull as an indicator for Great Lakes contaminants was that as an adult, it was non-migratory on the Great Lakes, i.e. it stayed on the lakes all winter. This meant that any contaminants found in the eggs of these gulls almost certainly came from the Great Lakes and not from any wintering ground areas with unknown levels of contaminants to which the birds might have been exposed (Gilman et al., 1979).

The GLHGECMP is made up of 15 sites, two sites in each of Lakes Superior, Michigan and Erie, three sites

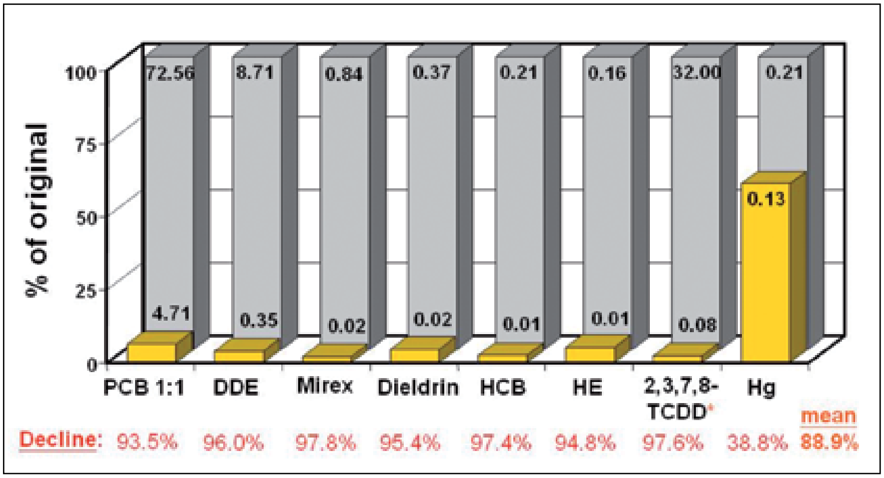


Figure 14.3. Percentage decline in concentration of eight contaminants in herring gull eggs from Port Colborne, Lake Erie, between 1974 and 2007 (earliest and most recent years of analysis). Inclusive years of analysis for TCDD are 1984 and 2005. Concentrations in the first year of analysis are standardised to 100%. Original concentrations given at top of histograms, most recent concentrations at bottom. All concentrations are ug/g (wet weight) except for TCDD, which is pg/g. Source: D. Moore, Canadian Wildlife Service, Burlington, Ontario.

in each of Lakes Ontario and Huron and single sites in each of the Detroit, Niagara and St. Lawrence rivers (Figure 14.2). Fresh herring gull eggs, one from each of 13 completed clutches, are collected at each site during late April or early May, early in incubation (Gilman et al., 1979). The eggs are sent directly to the CWS National Laboratory in Ottawa for analysis. Up until 1986, all eggs were analysed individually; from 1986 onwards, due to increasing analytical costs, eggs have been analysed as site pools, i.e. a single analysis of an egg pool made up of an aliquot from each of the 13 eggs from a given site (Turle and Collins, 1992). The eggs are analysed for over 75 contaminants, including congeners, most of the legacy contaminants (DDE, PCBs, mirex, dieldrin, etc.) but also dioxins, furans, brominated diphenyl ethers and mercury.

The chemical data from these eggs allow rigorous statistical analysis for spatial and temporal trends. Spatial trends, or patterns, in the distribution of contaminant concentrations in gull eggs have been used to identify areas of greater and lesser contamination. Means were calculated for each of seven contaminants at the 15 herring gull monitor sites for the five year period 1998-2002. The results were weighted according to fish flesh criteria for the protection of piscivorous wildlife. The three highest ranking (dirtiest) sites were in Lake Huron, the St. Lawrence River and Lake Michigan, while the three lowest ranking (cleanest) sites were in Lake Superior, Lake Huron and Lake Erie (Weseloh et al., 2006).

Temporal results and trends in contaminant levels are portrayed using two different methods. First, for example, for the Port Colborne site in Lake Erie, the eight contaminants illustrated in Figure 14.3 have declined from 38.8% (Hg) to 97.8% (Mirex) since they were first measured, usually in 1974. This portrays a simple 'then and now' comparison. Histograms such as these have also been calculated for individual sites (Weseloh et al., 2005). Second, temporal trends are assessed using a change-point regression analysis (Draper and Smith, 1981; Pekarik and Weseloh, 1998; DiMaio et al., 1999). This analysis not only indicates whether there has been a statistically significant change in contaminant concentration over time, but also whether there has been a change in the slope of the regression during the time series, i.e. for most of the data in the GLHGECMP, it will identify whether there has been a change in the rate of decline over time. Four different types of models were found from this analysis (Figure 14.4). Most contaminants are now decreasing as fast or faster than they were earlier in the study. This is very useful in alerting the researcher to a possible change in contaminant uptake by, or availability to, herring gulls during the study period. Combining this analysis with an analysis of stable isotopes and fatty acids in gull eggs, Hebert and Weseloh (2006) concluded that much of the recently noted increase in the rate of decline of several contaminants could be attributed to a change in the trophic level at which the gulls were feeding. There had been a

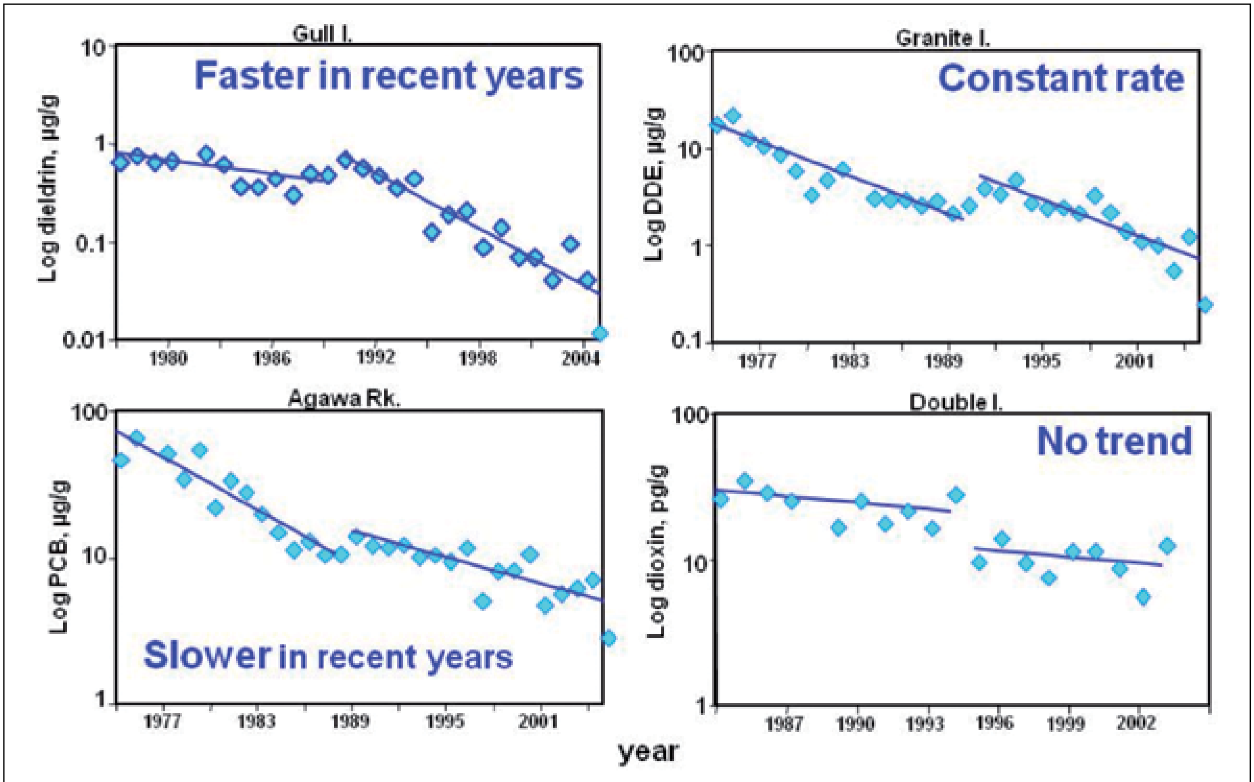


Figure 14.4. The four different models of temporal patterns in annual contaminant concentrations in herring gull eggs resulting from change-point regression analysis. The contaminant of concern for each model is shown on the y-axis; the herring gull colony for each contaminant is listed immediately above each model. Source: D. Moore, Canadian Wildlife Service, Burlington, Ontario.

switch from predominantly aquatic-based food to a more terrestrially-based diet. Hence, a portion of the decline we are seeing in the contaminant levels is due to a switch to a less contaminated food source rather than to a reduction in contaminant levels in the environment per se.

In addition to temporal and spatial trends, another important component of the Herring Gull Program is the search for new contaminants and their tracking. Gauthier et al. (2008) have shown dramatic changes in the temporal trends of polybrominated diphenyl ethers (PBDEs) in gull eggs. They found that in recent years BDE-209 showed much shorter doubling times than any of the other BDE congeners. The data illustrate that the PBDE concentrations and trends in congener pattern have changed greatly between 1995 and 2006 and this is of increasing concern.

Effects of Toxic Chemicals on Great Lakes Herring Gulls and other Colonial Waterbirds

With many early contaminants studies in the 1960s and 1970s, it was the appearance of an ‘effect’ which first alerted researchers to potential environmental problems. It is more cost-effective and provides necessary basic data on population size and productivity to go to the breeding colony and measure clutch size, hatching success and reproductive success than it is to conduct expensive chemical analyses before one knows what one might be looking for. The early herring gull studies on the Great Lakes showed that although the birds’ clutch size was normal, they were not producing very many young birds. Normal productivity should have been 1.0-1.5 young per pair of adults. What was being produced on the Great Lakes was often less than 0.5 young per pair, or even less than 0.1 young per pair (Keith, 1966; Gilbertson and Hale, 1974a, b; Teeple, 1977). Shortly after the use of DDT and PCBs was restricted in both the US and Canada, in the early 1970s, concentrations of these compounds in gull eggs declined dramatically and productivity of the gulls returned to normal (Mineau et al., 1984, Hebert et al., 1999).

Although contaminant conditions greatly improved during the 1970s and 1980s, very specific field and laboratory studies showed that there were still several effects of chlorinated organic compounds present at the embryonic and molecular levels in herring gulls (Hebert et al., 1999). Using an embryonic viability detector (Mineau and Pedrosa, 1986), Craig Hebert was able to show that there was still a relationship between the viability of the gull embryo in the egg and the contaminant concentrations in the egg (Environment Canada, 2003). Laird Shutt showed that male gulls which nested in more contaminated areas were more prone to carry vitellogenin, a female egg protein, in their blood stream than those from much less contaminated areas – a sign that feminisation was occurring. Keith Grasman was able to show a depressed immune system in young herring gulls and black-crowned night-herons (*Nycticorax nycticorax*) from more contaminated areas (Environment Canada, 2003).

In the late 1980s and early 1990s, hepatic porphyria (a serious liver disease) was reported in herring gull embryos. It was most likely caused by exposure to certain chlorinated organic compounds (i.e. highly carboxylated porphyrins, HCPs) that are known to cause porphyria in laboratory-exposed birds (Gilbertson and Fox, 1997, Kennedy and Fox, 1990). The most recent work by this group has provided strong evidence that the cause of elevated HCPs in herring gull liver was exposure to polychlorinated biphenyls (PCBs) (Kennedy et al., 1998). In the early 1990s, Gilbertson et al. (1991) suggested a Great Lakes Embryo Mortality Endocrine Deformity Syndrome (GLEMEDS) to describe contaminant-induced conditions in several species of colonial waterbirds in the Great Lakes. They hypothesised that GLEMEDS in colonial fish-eating birds resembled chick oedema disease of poultry and had been caused by exposure to chick oedema active compounds that had a common mode of action through the cytochrome P-448 system.

In the late 1990s, Lorenzen et al. (1999) examined the relationships among organochlorine residues, plasma corticosterone concentrations and intermediary metabolic enzymes in Great Lakes herring gull embryos. They found that chlorinated organic compounds were present in gull embryos at concentrations that may have affected the hypothalamo-pituitary-adrenal axis and associated intermediary metabolic pathways.

Recent research by Kennedy and colleagues has moved toward determining the effects of some of the new persistent organic pollutants in herring gulls. The new pollutants include brominated flame retardants (BFRs) and perfluoroalkyl compounds (PFCs). Their research includes both egg injection studies using fertile herring gull eggs and studies on the effects of PFCs on selected molecular targets in cultured avian hepatocytes, cardiomyocytes and neuronal cells. For example, a recent report by Crump et al. (2008) indicates the promise of using primary cultures of herring gull neuronal cells for determining the toxic and biochemical effects of BFRs in this species.

These recent studies show that although contaminant levels in gull eggs have declined by more than 90% for many of the legacy compounds over the last 35 years, and productivity has returned to normal at many locations, many subtle effects are still associated with herring gulls from the relatively more contaminated areas on the Great Lakes.

Botulism in Colonial Waterbirds on the Great Lakes

In the summer of 2004, large numbers of waterbirds were found dead and dying in eastern Lake Ontario. A tri-monthly survey to assess the numbers and species of birds dying was established on six islands in eastern Lake Ontario. During July to October, 2004 and 2005, over 3,000 dead birds were recorded. In terms of numbers, the double-crested cormorant was the most frequently encountered dead bird species, with more than 2100 sick and dead individuals recorded. Other species which contributed greatly to the total number of dead birds found included herring, ring-billed and great black-backed gull and Caspian tern. More dead black-backed gulls were found than were known to breed in Lake Ontario. Necropsies confirmed that botulism was present in all specimens tested. The great black-backed gull was wiped out as a nesting species in Lake Ontario, which had been its stronghold in the Great Lakes (Shutt et al., 2008)

Population Trends of Colonial Waterbirds on the Great Lakes

With approximately 35,000 islands, the question of how many colonial waterbirds nest on the Great Lakes islands and their distribution has always been an interesting one. Unfortunately, early records are quite sparse. Explorers and fur traders visited the Great Lakes as early as the 1600s, the southern shores of Lakes Ontario and Erie were settled initially in the early to mid-1700s, the north shores in the late 1700s and there was much military activity on those lakes during early 1800s. Unfortunately, we do not know of any quantitative records from that time. Commercial fishing and waterborne commerce developed soon after the War of 1812 and there were various ornithological treatises in the late 1800s (Wheaton, 1882; McIllwraith, 1894). Still, there were few if any quantitative records on the distribution or numbers of colonial waterbirds nesting on the Great Lakes.

One of the first related studies to evolve from the GLHGEMCP was initiated independently by Hans Blokpoel, on the Canadian Great Lakes, and by Bill Scharf on the US Great Lakes (Blokpoel, 1977; Scharf et al., 1978). It was their goal to visit all the waterbird colonies on the Great Lakes and record the numbers of nesting birds. These inventories evolved, in part, to answer the broader question of whether there were any population-wide impacts from toxic chemicals. A partial survey of the upper lakes had been completed by Ludwig (1962). The project has since become a decadal survey, being conducted every 10 years; it is a huge undertaking usually taking three years to complete. In the most recent survey, 1997-2000, it was estimated that approximately 1.8 million individuals of nine colonial waterbird species nested on the Great Lakes. Estimating the number of offspring and migrants which also used the Great Lakes in any one season increased the number of colonial waterbirds which used the Great Lakes annually to approximately 3.8 million. Ring-billed gull, herring gull and double-crested cormorant were the three most numerous species, while black-crowned night-heron, great egret and great black-backed gull were the least numerous. In terms of temporal trends (at Canadian sites), cormorants and black-crowned night-herons have been increasing, common terns have been declining steadily and the gulls and large herons have fluctuated, usually showing a substantial increase between the 1970s and the 1980s but then a decline between the 1980s and the 1990s. The census for the 2000s is currently underway. The surveys were begun too late to show a decline in population levels associated with increasing contaminant levels, which presumably occurred in the 1950s and 1960s, but they have captured a strong recovery for many species from the 1970s to present. Clearly, the census of all colonial waterbirds on the Great Lakes is an important programme for tracking fluctuating population levels and identifying management issues with both superabundant and rare species.

Effects and Remediation of Oil Spills on Wild Birds

The St. Lawrence Estuary and Gulf Experience

15

Guy Fitzgerald
University of Montreal, Saint-Hyacinthe, QC, CAN

Québec's first contingency plan for oiled wild birds was set up by the federal government and the oil industry in 1985. A bird of prey rehabilitation programme was initiated at the College of Veterinary Medicine, Université de Montréal, in 1987. In 1993, we were approached by a federal governmental agency (Environment Canada) to manage all oiled bird aspects of the contingency plan. In 1999, 49 tonnes of fuel were spilled near Havre-Saint-Pierre in the St. Lawrence seaway. Over 1,000 birds (mainly common eiders, *Somateria mollissima*) were affected even with such a relatively small oil spill. After this major event, strategies and planning were reviewed with the assistance of the international community of oiled wildlife rehabilitators to improve the preparedness and efficiency for future responses.

The St. Lawrence Seaway includes the St. Lawrence Gulf, Estuary and River (Figure 15.1 and Chapter 11), a huge body of water about 1,700 km long. The coastline represents more than 4,300 km and is a difficult area to navigate. Officers of most ships are assisted by an experienced pilot when entering this seaway. About 3,000 ships navigate through the Seaway every year. Nearly 24 million tonnes of oil in cargo and 9 million tonnes of propulsion fuel are moved in the seaway annually. Nearly 50 oil spills occur annually along the St. Lawrence River

Figure 15.1. The St. Lawrence Gulf, Estuary and River.

Table 15.1. National organisations involved during an oil spill along the St. Lawrence Estuary and Gulf.

Name	Description	Function related to wildlife
Canadian Coast Guard	Governmental (federal agency) http://www.ccg-gcc.gc.ca/eng/CCG/Environmental_Response	Major role in incident command system (ICS); regulation; interaction with the party responsible
Canadian Wildlife Service (Environment Canada)	Governmental (federal agency) http://www.qc.ec.gc.ca/faune/faune/html/spills.html	Major role in ICS, bird surveys, modelling bird population impact, recommendations on intervention
Eastern Canada Response Corporation Ltd (ECRC)	private corporation http://www.simec.ca/en/home/default.asp	Interaction with the party responsible; coordination of field operations (pumping, hazing buoys, shore cleaning); major role in ICS; facilitator with capture and rehabilitation of birds
Fondation Les Oiseleurs du Québec	private non-profit organization http://www.oiseleurs.ca/ang/index.html	Sub-contractor under ECRC; role in ICS; bird surveys; bird hazing; bird capture; coordination of the rehabilitation centre
Centre québécois sur la santé des animaux sauvages	Veterinary College, Université de Montréal	Wildlife health

but only one or two involve large deployment logistics. Wildlife is rarely involved.

Depending on the season, however, 200,000 to 1,000,000 birds of 129 species are present along the St. Lawrence River, Estuary and Gulf. At times, the concentration of birds can be important in a given area and even a minor spill could have a major impact if occurring in such an area. For instance in the autumn, Cap Tourmente, near Québec City, can harbour over 200,000 snow geese (*Chen caerulescens*). In addition, the season of occurrence is crucial. In cold water and weather in winter, the mortality of oiled birds is higher even in the presence of a minimal amount of oil on the plumage.

For an efficient response to a crisis, preparedness plays a major role. Managing the event needs a multi-disciplinary approach; it needs among others an army-like 'incident command system'. Then comes the expertise of different organisations and their networking. Tables 15.1 and 15.2 describe the involvement of different local and international organisations. Some organisations offer training opportunities.

Oil Spill Response in Quebec

1) Spill Detection

Spill events are first communicated to Environment Canada and the Canadian Coast Guard. A sequence of actions is initiated including: contacts with the responsible party and the ship insurances, chemical and toxicity tests on the product spilled and contacts with responders.

The oil source may be unknown. 'Mystery' offshore oil spills have been responsible for the death of more than 300,000 sea birds annually in international seas near Newfoundland, Canada (Wiese and Robertson, 2004). The same authors developed a general mathematical Oiled Seabird Mortality Model (OSMM) to assess seabird mortality due to chronic oil pollution. The south of Newfoundland provides food to near tens of millions of seabirds year-round and is among the highest oil polluted area in the world, just at the entrance of the St. Lawrence Gulf.

Subsequently to those studies and to non-governmental pressure, in 2005, the Canadian Government adopted bill C-15, an act to amend the Migratory Birds Convention Act and the Canadian Environmental Protection Act against the dumping of oily bilge wastes, or other pol-

Table 15.2. International oil spill response organisations or training opportunities.

Name	Description
International Bird Rescue Research Center (IBRRC)	Private non-profit organisation, USA http://www.ibrrc.org/
Oiled Wildlife Care Network (OWCN)	University of California Davis, School of Veterinary Medicine, USA http://www.vetmed.ucdavis.edu/owcn/
Tri-State Bird Rescue and Research	Private non-profit organisation, USA http://www.tristatebird.org/response
Focus Wildlife	Private non-profit organisation, Canada and USA http://www.focuswildlife.net/
Office of Spill Prevention and Response (OSPR)	http://www.dfg.ca.gov/ospr/
Southern African Foundation for the Conservation of Costal Birds	http://www.sanccob.co.za/
Oil spill response (OSRL)	International organisation in oil spill response http://www.oilspillresponse.com/ http://osrlearl.com/pdf/Published%20Papers/Rob%20Holland/paper_full_oiled_wildlife_response.pdf
International Petroleum Industry Environmental Conservation Association (IPIECA)	http://www.ipieca.org/
Sea Alarm	International organization in oil spill response http://www.sea-alarm.org/
Centre of Documentation, Research and Experimentation on Accidental Water Pollution	Oil spill response in France http://www.cedre.fr/index_gb.html
International Alliance of Oiled Wildlife Responders (IAOWR)	Group of oiled wildlife responders meeting periodically to promote best practices and to promote networking in the domain.
Effects of Oil on Wildlife (EOW)	International meeting occurring every two years to improve communication of best practice in oiled wildlife response. Tallinn, Estonia October 2009 : http://www.eowconference09.org/

lutants, into the ocean. This law resulted in an increased aerial monitoring of oil spills and improved possible prosecutions.

2) Restraining of Oil and Environment Restoration

A variety of heavy equipment is directed to the spill including pumps, storage barges, booms, skimmers and shoreline cleaning gear. After a spill, the party responsible must contract an organisation (usually ECRC) to control the spill. The teams mobilise a tremendous amount of logistics (boats, trained personnel, safety equipment, housing, food supply).

3) Assessment of Impact on Birds

In Quebec, the Canadian Wildlife Service maintains a computerised database on the bird population along the St. Lawrence River. It includes species and its conservation status, number of birds related to the location and time of the year. Thus, the potential impact of the spill can be roughly estimated. Nevertheless, a bird inventory is started in the affected area by volunteers, biologists on shore or by aerial surveys. Gulls are used as indicators because oil is easily detected visually on their white plumage.

4) Deterring Wildlife

The impact on birds is not necessarily proportional to the amount of the oil spilled. A small spill can be dramatic if a large flock goes into it. Thus, it is crucial to prevent the birds from coming into the spill. Different deterrent techniques exist in order to keep birds away. Pyrotechnical devices (propane cannons, starter pistols) and horns can be used from the shore or from boats. Helicopters, falconry and even small radio control airplanes can be useful. The Breco Bird Scarer© (Whisson and Takekawa, 1998).

was created in Quebec in the mid-1990s. It follows the oil spill as it is displaced by the tide, wind and waves. This equipment can be efficient if used in a very short time after the spill. The best approach would be to have them ready to launch from the ship involved in the spill.

5) Initiation of Wildlife Rescue

Undertaking a bird rehabilitation operation is not yet a legally regulated decision in Canada. Governmental, media and public pressures usually influence the party responsible to financially support the operation. The challenge lies more in the logistics than in the basic principles of cleaning a contaminated bird. Best practices are described by US Fish and Wildlife Service (2003, 2005) and by Beaulieu and Fitzgerald (1996). Other documents are in the process of being completed by IBRRC. The rehabilitation centre (permanent or temporary) must meet these basic requirements to be efficient:

- water softeners must be installed if the hardness of local tap water is > 3 grains (50 mg/litre);
- nearly 3,000 litres of hot water (40-41°C) per hour may be necessary, thus high-performance water heaters are essential;
- water pressure for rinsing with a hose must be near 40-60 psi;
- large supply of liquid soap: Dawn© is preferred at concentration of 1-4%;
- ambient temperature must stay around 25-30°C and ventilation should recycle 8-10 volumes per hour.

The preparation of the facilities can be started while the capture operation takes place. It often takes a few days after the spill to capture the first birds. Throughout the whole rehabilitation process, veterinary supervision should be provided. Post-mortem of dead birds should be carried out, as well as some clinical laboratory tests on live birds to detect infectious diseases. Diving birds are generally sensitive to aspergillosis and preventive antifungal treatment, e.g. itraconazole at 10 mg/kg once a day orally (Sporanox©, Janssen-Ortho, Toronto, Canada), is recommended. Rehabilitators should be aware of contagious diseases such as avian cholera, avian influenza and duck herpes virus. The stress induced by rehabilitation and confinement could aggravate the spread of these diseases. Nobody would like to introduce a contagious disease into the wild population with the release of infected birds that are only temporarily without symptoms. With the new concerns about avian influenza, it is also recommended that no volunteer should be allowed to work with the birds while having symptoms of human influenza. Security and safety should always be part of the planning and the training on the premises. We do not want animals to get hurt, nor people.

6) Capture

Oiled birds can be captured using nets from boats or on the shore. In some cases nocturnal captures with intense lighting can be performed but staff safety should not be compromised. Other situations may necessitate the use of traps. In any cases, the skills of the capture team will have an impact on rehabilitation efficiency. The capture must be carried out as soon as possible because the health of oiled birds is compromised by the lost of thermoregulation, their energetic status (oiled birds stop feeding and consistently preen their feathers) and the ingestion of toxic products.

7) First Aid and Triage

On admission to the rehabilitation centre, birds have to be examined. Birds not only need to be cleaned but also need to be treated for hypothermia (hyperthermia if birds are confined in small boxes in a truck on a hot sunny day), for body condition (thin or emaciated) and for chemical irritation or intoxication. Life support must be administered including warm fluids, digestive tract protectors, warmth and food. Euthanasia may have to be performed on severely debilitated birds. Only when the bird's health status is considered sufficient to cope with the procedure may it go to the cleaning section.

8) Cleaning

It generally takes 2-3 persons to hold the bird and to perform the cleaning in successive water baths. After 3-4 soap baths, it may take 4-5 clean hot water baths to rinse the soap off the birds. Feathers are then rinsed with an appropriate hose (40-60 psi water pressure). There should not be any oil, detergent or calcium carbonate residues remaining on the feathers (Bakken et al., 2006). The formation of water droplets on the feathers is an indicator that the feathers are recovering their water repellency. Finally, the bird should go into a drying enclosure. Commercial pet air dryers are usually connected to small pens to achieve this step.

A bird washing machine has been developed (http://www.fost.fr/mlo_en.html) to facilitate cleaning operations but its efficiency is not recognised by all rehabilitators. This machine necessitates fewer people to perform the cleaning but the stress experienced by the bird is not decreased and cleaning results are inferior to those achieved by manual washing. Again, the issue of the whole process is not to clean birds but to rehabilitate them. Most of the energy must be devoted to the stabilisation first, and then reconditioning after the cleaning.

9) Reconditioning

All steps are important but this step is crucial. After the cleaning steps, the bird must recover its full water repellency. Indeed, the feathers are water repellent because of their structure. The massage effect of the water pressure during the rinsing step and feather preening are essential to the recovery of the normal feather structure. The bird needs to be able to move and eat in a pool (diving, floating, swimming). The salt gland of a pelagic bird must be efficient to allow the bird to drink salt water without getting intoxicated. The supraorbital glands (also known as salt gland or nasal gland, located ventrally to both eyes) concentrate and excrete salt. These glands secrete a solution up to 5% NaCl. When a pelagic bird is kept in fresh water for a while, the gland becomes atrophied and needs to be reconditioned before regaining its normal function.

Efficient pools are required. A water overflow system helps keep the water clean but necessitates a significant flow rate. Dirty water contaminated with bird faeces and fish oil may make a second cleaning necessary because it is detrimental to restoring feather waterproofing.

10) Release

Before considering the release of a rehabilitated bird, some health criteria must be looked at: body condition, weight, haematocrit and other blood chemistry parameters (Anderson et al., 2000). A location appropriate for releasing rehabilitated birds should be selected; it is important to avoid the contaminated area or areas where the food chain has been severely affected. If the breeding site is involved, the birds may try to come back to the spill site before the habitat is cleaned and become re-contaminated. Bear in mind that if rehabilitated animals are game birds, they could subsequently be ingested by (human) hunters.

11) Post-release Survival Assessment

Oiled wildlife rehabilitation still has to make progress, even if tremendous improvements have been achieved over the last 20 years. The survival and reproductive success of the rehabilitated birds are the parameters measured to evaluate the success of the rehabilitation. Banding (ringing), transmitters (implantation or harness) and subsequent tracking, along with breeding success surveys, can all help documenting these parameters.

Bird Population at Risk and Priority Species List

In general, people would rather not discriminate between species and do whatever can be done to save every single bird affected by the spill. However, it is well accepted that the number of birds that must be treated may overwhelm the capabilities of the facilities. It is possible that difficult decisions have to be made: selecting the birds with the best chances of surviving the whole process or choosing the species with a particular status (endangered as opposed to abundant species).

The Canadian Wildlife Service has developed a cleaning priority index of species in the event of a spill (Daigle et al., 2006). The concept of a population management approach is not largely accepted in the oiled wildlife responder community, but the Canadian Wildlife Service index takes into consideration the population, its productivity, its conservation status, and its relations with economic activities. For instance, over recent years the snow goose (*Chen caerulescens*) population has expanded so much in Canada that spring hunting has been launched. In 2000, an inland oil spill affected a group of around fifty snow geese near Victoriaville in Quebec. The decision was taken to capture and to humanly perform euthanasia on this group instead of setting up a rehabilitation operation.

By contrast, the St. Lawrence River harbours some vulnerable species such as harlequin duck (*Histrionicus histrionicus*) and Barrow's goldeneye (*Bucephala islandica*). With a wild population of 2-4 thousands individuals living on North America's East Coast, a major oil spill occurring in an area densely occupied by one of these, during wintering for instance, could be dramatic. The authorities would then probably do everything they possibly could to save as many individuals as possible.

Environmental Cost and Benefits of the Response

Some studies have shown that aggressive habitat restoration (cleaning shores with high pressure and washing devices using high water pressure) can affect or be more detrimental to the micro-environment than if the impacted area were left alone (see Whitfield, 2003 and Arnold, 2011). Research has been conducted to develop micro-organisms that could be dispersed in the spill and degrade petroleum products.

Regarding wildlife rehabilitation, the amount of money, human resources and tap water needed to manage a crisis professionally is tremendous. The use of fresh water is becoming a worldwide sensitive issue. It may become impossible to operate with success in certain remote areas. For instance, on the Iles-de-la-Madeleine located off the Quebec Atlantic coast, water availability would not support a bird rehabilitation centre during an oil spill. There is no technology available yet that allows sea water to be used to clean oiled birds. Such a technology would probably be prohibitive energy- and cost-wise. In some countries burning the oil spill and killing (gun shooting) affected birds have been considered the best option.

Furthermore, it has been reported by some scientists that rehabilitated oiled wildlife may not survive after the operation or that the reproduction rate is compromised, rending the whole process useless (Anderson et al., 1996). However other field studies (using transmitters) on the survival of those animals and evaluating breeding success have justified rehabilitation efforts (Underhill et al., 1999). There is no doubt that where an oil spill affects an endangered species, at least an attempt to rehabilitate is recommended. In these instances, the skills and expertise acquired through the years with more common species will be significantly beneficial.

Conclusions

Prevention is the key to protecting wildlife from oil spills. It is generally accepted that most infamous oil spills, such as that from the Exxon Valdez, could have been avoided. Regulations (local as well as international) promoting safe tankers and ways to manage the oil industry will never be too strict to protect both environment and wildlife. Improvements in prevention have been carried out by more stringent regulations regarding boat safety and training requirements for pilots. Statistical trends show a decrease in major spills in recent decades. (see : http://www.itopf.com). However, as long as dangerous products are widely transported, we will never be totally safe from environmental disasters. To be ready to cope with those, preplanning and networking are essential. The response needs to be performed professionally or not at all!

Contaminants in Semi-aquatic Wildlife

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Lessons from the Laurentian Great Lakes

Matthew Zwiernik, Frouke Vermeulen and Steven Bursian
Michigan State University, East Lansing, MI, USA

Semi-aquatic wildlife species have been important sentinels of environmental health and remain important biomonitors concerning emerging contaminant issues. The semi-aquatic wildlife of the Laurentian Great Lakes (LGL) has led the way in this regard and continues to do so. This is because the LGL and associated land mass possess a set of unique characteristics that make it an ideal laboratory for identifying the presence of environmental contaminants and their effects on semi-aquatic wildlife. The LGL basin and its diverse habitats support a wide array of wildlife with a significant number of species being dependent on the preponderance of coastline habitats. These species living in the transition areas of aquatic and terrestrial environments are exposed to contaminants from broad sources. As a result, semi-aquatic wildlife species often experience exposure to contaminants that are greater than for either terrestrial- or aquatic-based species and thus have been key sentinels of environmental health, both in the LGL basin and elsewhere. For the LGL, human activities including industrial, agricultural and recreational have resulted in significant inputs of anthropogenic contaminants. The combination of contaminant input and shoreline habitats with slow water turnover rate results in a great potential for chronic contaminant exposure of semi-aquatic wildlife species. In 1962, environmental awareness was brought to the forefront when the observations of a LGL scientist were published in Rachel Carlson's *Silent Spring*. The assumptions that environmental contaminants were not very toxic and that the dilution potential of the LGL was infinite were clearly in error. Over the next 30 years, declines in individual and population health of LGL semi-aquatic species including mink, otter, bald eagles, terns, cormorants and numerous amphibian species warned the world of the potential for widespread, adverse impacts of contaminants on the environment. Compounds such as DDT, polychlorinated biphenyls (PCBs), polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and a great number of pesticides were being identified in the tissues of these species at concentrations associated with adverse effects. Today, through the combined work of scientists, government and industry, exposure to identified contaminants has been reduced within the LGL. As a result, populations of mink, eagles and cormorants and other semi-aquatic wildlife species have seen increases over the last two decades and the lessons learned have pushed this trend worldwide. Today more than 600 programmes are utilised to monitor the environmental health of the eco-laboratory we call the LGL (www.glc.org/monitoring/greatlakes/). Semi-aquatic wildlife species continue to be a key component of that effort.

Laurentian Great Lakes

Lakes Erie, Huron, Michigan, Ontario and Superior comprise the Laurentian Great Lakes, which together form the world's largest fresh water system. The Great Lakes have a total volume of 23,000 km^3, a surface area of 244,000 km^2 and a drainage area of 750,000 km^2. The Great Lakes represent 80% of the total surface water of North America and 20% of the world's freshwater supply (Li et al., 2006; Ward et al., 2008). These collective waters are a vital economic and environmental resource for both the United States and Canada. Economically, the area generates over $330 billion in US/Canada trade, accounting for 18% of the gross domestic product of the two countries (Zhu and Hites, 2004; Ward et al., 2008). The basin is rich in terrestrial and aquatic wildlife, including over 100 species that are considered to be rare, endangered or at risk.

Because of the economic benefits associated with being in close proximity to a natural resource of this magnitude, the area surrounding the Great Lakes is densely populated and highly industrialised. Forty million people, representing 10% of the United States population and 30% of the Canadian live within the Great Lakes basin. As a result of contaminant runoff from urban, agricultural and industrial sources and atmospheric deposition, in excess of 1,000 chemicals have been identified in the water column or resident biota. Approximately 350 of these chemicals are present in significant quantities (Ward et al., 2008).

Impact of Contaminants on LGL Wildlife

In the mid-1960s, the impact of chemicals on the health of semi-aquatic wildlife species within the LGL began to concern biologists. Impaired health of individuals, reproductive failures and population declines of mink, eagles and a number of colonial fish-eating birds were noted. For avian species, the signs observed were consistent with those of 'chick oedema disease' in domestic chickens (*Gallus domesticus*), which included oedema, hydropericardium, ascites, liver enlargement, porphyria, hepatic necrosis with fatty degeneration and a high rate of mortality following exposure to PCBs (Vos and Koeman, 1970; Vos, 1972; Gilbertson, 1983). A similar relationship was established for PCB exposure and impaired reproductive performance of mink on commercial fur farms within the LGL basin (Aulerich et al., 1971). Later, Ludwig et al. (1996) determined that there was a relationship between exposure to PCBs and the structurally similar PCDDs and PCDFs and the increased incidence of embryonic deformities and mortalities in double-crested cormorants (*Phalacrocorax auritus*) and Caspian terns (*Hydroprogne caspia*) nesting in the LGL. Regional studies indicated decreased hatching success and increased incidence of nestling deformities in a colony of double-crested cormorants located in an area along Lake Michigan contaminated with PCBs when compared with a reference colony (Larson et al., 1996). Threshold concentrations of PCBs and PCB-like contaminants in eggs were suggested, based on a study examining the contaminant burden and the impaired reproductive success of LGL Forster's terns (*Sterna forsteri*) (Kubiak et al., 1989). The severity of reproductive failure observed in colonies of herring gulls (*Larus argentatus*) throughout the LGL appeared to be directly related to contaminant concentrations in eggs, although a casual relationship could not be established (Gilbertson, 1983). While similar observations of impaired wildlife health were noted elsewhere, this suite of signs became collectively known as the Great Lakes embryo, mortality, oedema and deformities syndrome (GLEMEDS) (Gilbertson et al., 1991).

Polychlorinated Contaminants

As suggested above, PCBs, PCDDs and PCDFs have been scrutinised for the past 40 years as potential contaminants responsible for a number of the effects observed in wildlife residing in the LGL basin. The PCDDs, PCDFs and PCBs are widely distributed into the global environment and can be very resistant to environmental degradation and me-

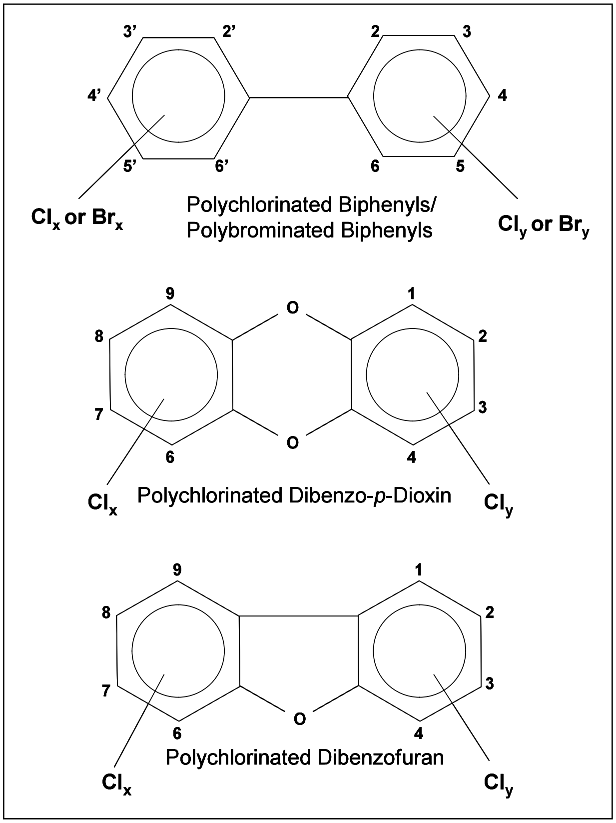

Figure 16.1. Structure and numbering of generic polychlorinated/polybrominated biphenyl (PCB/PBB), polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran molecules.

tabolism. As a result, they readily accumulate in the food chain, with the greatest tissue concentrations being found in species at the higher trophic levels. Residues have been detected in a variety of animal species, including humans (Van den Berg et al., 1994; Safe, 1998). In some situations, the environmental concentrations of these contaminants are such that there are concerns about the health of wildlife and humans. Because of this concern, there continues to be an effort on the part of state, federal and international regulatory agencies to minimise exposure to this significant class of environmental contaminants.

PCBs are formed by substituting chlorine for hydrogen on the biphenyl molecule, which consists of two benzene rings (Figure 16.1). Theoretically, there are 209 possible PCB congeners considering the five chlorine-binding sites on each ring. Each congener has been assigned a unique number from 1 to 209 in accordance with the rules of the International Union of Pure and Applied Chemistry (IUPAC). Commercial PCB products are mixtures of congeners that differ with respect to the extent and positions of chlorination. Polychlorinated dibenzo-*p*-dioxins are composed of two benzene rings connected by two oxygen atoms and contain four to eight chlorines, for a total of 75 congeners (Figure 16.1). Polychlorinated dibenzofurans are also composed of two benzene rings. The rings have a single oxygen between them and have four chlorine binding sites available on each ring (Figure 16.1). There are 135 different PCDF congeners (Di Carlo et al., 1978; Safe, 1990, 1998; Headrick et al., 1999; Huwe, 2002; Mandal, 2005; Schecter et al., 2006).

Certain approximate stereoisomers in this group, often referred to collectively as dioxins and dioxin-like compounds, induce a common suite of effects and have a common mechanism of action mediated by binding of the polychlorinated ligand to a specific high-affinity cellular protein. This group of dioxins and dioxin-like chemicals includes seven PCDD congeners, 10 PCDF congeners, and 12 PCB congeners. The prototype for the dioxins is 2,3,7,8-tetrachlorodibenzo-*p*-dioxin or TCDD. Toxicity and persistence of the polychlorinated compounds are determined by structure, with lateral substitutions on the ring resulting in the highest degree of toxicity. For the PCDDs and PCDFs, congeners with chlorines in the 2, 3, 7 and 8 positions fall into this category. The dioxin-like PCB congeners are the non-ortho- and mono-ortho-substituted compounds with no chlorines or one chlorine, respectively, on the 2, 2', 6 or 6' position (Safe, 1990; Headrick, 1999; Huwe, 2002; Mandal, 2005; Schecter et al., 2006).

Mechanistic studies indicate that the toxic and biochemical effects associated with exposure to TCDD and its approximate stereoisomers are mediated by initial binding of the chemical to the cytosolic aryl hydrocarbon receptor (AhR) present in target tissues and organs. There is a correlation between the AhR binding affinity of these chemicals and their structure-toxicity relationships, which supports the idea that the Ah receptor is involved in the mediation of responses induced by the TCCD-like PCDD, PCDF and PCB congeners (Okey et al., 1994; Hahn, 1998, 2002; Safe, 1998; Denison et al., 2002; Denison and Nagy, 2003; Mandal, 2005).

The common mechanism of action of TCDD and related compounds has led to the use of a toxic equivalency factor (TEF) approach to estimate the TCDD-like toxicity of complex mixtures containing chemicals that resemble TCDD. The TEF is a consensus value based on multiple data sets in an attempt to predict the potency of an individual congener relative to TCDD. Using the TEF concept, TCDD toxic equivalents (TEQs), which is the sum of the product of the concentration of each congener and its respective TEF, can be calculated for any complex mixture containing TCDD-like chemicals to provide an estimation of the total TCDD-like toxicity (Safe, 1998; Huwe, 2002).

Commercial production of PCBs, primarily by the Monsanto Corporation, began in the United States in 1929 until 1977, (Tanabe, 1988; Kimbrough, 1987, 1995; Headrick et al., 1999). They were used in closed use systems including electrical transformers, capacitors and heat transfer and hydraulic systems. For a period of time, PCBs also had a large number of open-ended applications. They were used in paints, polymers, and adhesives, as lubricants, plasticisers, fire retardants, immersion oils, vehicles for pesticide application and as agents for the suspension of pigments in carbonless copy paper (Safe, 1990; Headrick et al., 1999). The PCB products that were manufactured by Monsanto in the United States had the trade name Aroclor followed by four digits that identified the particular mixture. The first two digits referred to the 12 carbon atoms of the biphenyl molecule and the last two digits referred to the percentage of chlorine in the mixture, by weight. Aroclors 1221, 1232, 1242, 1254 and 1260 were the commercial PCB products that were produced by Monsanto, containing 21, 32, 42, 54 and 60% chlorine by weight, respectively. Similar commercial PCB mixtures were produced by other manufacturers worldwide including the Clophens (Bayer, Germany), Pheoclors and Pyralenes (Prodelec, France), Fenclors (Caffro, Italy) and Kanechlors (Kanegafuchi, Japan) (Kimbrough, 1987, 1995; Safe, 1994).

The physical and chemical properties of PCBs, such as high stability, inertness and dielectric properties, that were advantageous for many industrial purposes, led to the international use of PCBs in large quantities (Tanabe, 1988). For example, the estimated cumulative production of PCBs in the United States between 1930 and 1975 was 700,000 tonnes, and worldwide production was approximately 1.2 million tons. Domestic sales of PCBs in the United States during this time period totalled 627,000 tons (Kimbrough, 1987, 1995; Tanabe, 1988). As a result of widespread use, PCBs were identified in environmental media and biota as early as the 1960s. After the discovery of their widespread environmental contamination in the 1970s, PCB production decreased and eventually ceased (Tanabe, 1988). In 1971, Monsanto voluntarily stopped production of PCBs for open-ended uses and subsequently only the lower chlorinated biphenyls were produced (Aroclor 1242 and 1016). In 1977, Monsanto ceased production entirely (Kimbrough, 1987, 1995). Although PCBs are no longer used commercially because of their persistence, they are still present in the environment. About 31% of all the PCBs produced are present in the global environment. It is estimated that 780,000 tons are still in use in older electric equipment and other products, deposited in landfills and dumps or in storage (Tanabe, 1988).

Polychlorinated biphenyls were first detected in the Great Lakes in 1968. Since the cessation of PCB production in 1977, PCB concentrations in sediments of all the Great Lakes have either levelled off or declined. Li et al. (2009) estimate a current total accumulation of approximately 300 metric tons of PCBs in the sediment of the Great Lakes, which is 30% less than the 1980 estimate. However, Li and associates (2009) state that even if PCB degradation were substantial and widespread, the ultimate elimination of PCBs from the Great Lakes would take decades or centuries to complete. Furthermore, He et al. (2001) concluded from a 20-year survey of trends in sport fish consumption that serum PCB concentrations among consumers of sport-caught Great Lakes fish increased in the 1970s and did not subsequently decline in the 1980s, presumably because of continued low-level exposure through fish consumption and the long half-life of PCBs. Hickey et al. (2006) reported on trends of chlorinated organic contaminants, including PCBs, in Great Lakes trout and walleye from 1970 to 1998. They concluded that although concentrations of most contaminants in predator fish in the Great Lakes have continued to decrease during the 1990s, the rapid decrease in concentrations observed through the 1970s and 1980s has slowed. Furthermore, concentrations of several contaminants, including PCB congener 126 (3,3', 4,4', 5-pentachlorobiphenyl), which is the most toxic dioxin-like PCB congener, appear to have stabilised. Kannan et al. (2000) reported the results of a comprehensive analysis of PCB congeners for fish originating from Michigan waters, including the Great Lakes. Concentrations of total PCBs in walleye and carp were approximately double the US Food and Drug Administration recommended tolerance limit of 2,000 ng/g, weight wet (ww) for human consumption. When expressed on a TEQ basis (Van den Berg et al., 1998), which reflects the dioxin-like toxicity, fish fillets contained from 0.46 to 58 pg TEQ/g (ww) derived from PCBs, with PCB 126 accounting for 50 to 83% of the TEQs.

Polychlorinated dibenzo-*p*-dioxins and PCDFs have never been produced for commercial sale. Their presence in the environment is the result of their formation as by-products of commercial and natural processes (Safe, 1990). Some of the important industrial sources of PCDDs and PCDFs have included their formation as by-products in the production of PCBs, chlorinated phenols and chlorinated phenol-derived chemicals, hexachlorobenzene, technical hexachlorocyclohexanes, and chlorides of iron, aluminium and copper. PCDDs and PCDFs have also been identified in effluents, wastes and pulp samples from the pulp and paper industry and in finished paper products. Emissions from municipal and hazardous waste incinerators as well as home heating systems that use wood and coal, diesel engines, forest and grass fires and agricultural and backyard burning contain PCDDs and PCDFs. Another contribution might come from naturally formed PCDDs and PCDFs, which have been detected in deep soils and clays from the southern United States and Germany (Safe, 1990; Huwe, 2002).

Historical data suggest that PCDDs and PCDFs entered the environment in the 1930s and 1940s, with releases peaking in the 1970s. Since the 1970s, emissions and abundance have been decreasing (Bhavsar et al., 2008). The United States Environmental Protection Agency (EPA) estimated that annual emissions of PCDDs and PCDFs decreased by 75% from 13.5 to 2.8 kg TEQ/year between 1987 and 1995. This was due primarily to improvements in incinerator performance and removal of incinerators that could not meet emission standards. Other regulations, including bans or restrictions on the production and use of chemicals such as the wood preservative pentachlorophenol (PCP), the phase-out of leaded gasoline that contained halogenated additives, and the elimination of chlorine bleaching in the pulp industry also contributed to reducing concentrations of PCDDs and PCDFs (Huwe, 2002). There has been a further 50% decline in emissions between 1995 and 2000 from known sources in the US (Bhavsar et al., 2008). Bhavsar and associates (2008) measured the concentrations of 17 of the most toxic PCDDs and PCDFs in lake trout or lake whitefish collected between 1989 and 2003 from the Canadian Great Lakes. The results of the study indicate that PCDDs and PCDFs are ubiquitous in lake trout and white fish in the Canadian Great Lakes, with the greatest concentrations occurring in Lake Ontario fish (54 pg TEQ/g in 1989 and 22 pg TEQ/g in 1999). TEQ concentrations in lake trout from Lakes Superior and Huron are 60 to 95% less than those from Lake Ontario. Temporal trend data suggest that TEQ concentrations are decreasing in Lake Ontario lake trout at approximately 1.5 pg/g/yr. TEQ concentrations also appear to be decreasing in Lake Huron lake trout, while they have stabilised in Lake Superior lake trout.

Despite a substantial drop in the emissions and environmental concentrations of PCDDs and PCDFs as well as the decline in environmental concentrations of total PCB concentrations, these chemicals are currently responsible for a substantial proportion of sport fish consumption limit recommendations for both the Canadian and US Great Lakes (Bhavsar et al., 2008). In addition, the TEQ concentrations associated with PCBs, PCDDs and PCDFs in the Great Lakes basin continue to be within a range associated with potential deleterious effects in a number of wildlife species.

Sentinel Species

Because humans and vertebrate species that share the same environment often have similar responses resulting from exposure to toxic substances, certain animal species can be used to monitor environmental contaminant exposure and effects. These species are considered to be sentinel species, which are defined by Grove et al. (2009) as those that are used to evaluate environmental contamination and its implications on environmental health based on their chemical sensitivity, position in the biotic community, exposure potential and geographical distribution or abundance. For an animal to serve as a sentinel species it must meet certain requirements as presented in Basu et al. (2007). These criteria include (1) widespread distribution, (2) high trophic status, (3) ability to accumulate contaminants, (4) maintained and studied in captivity, (5) captured in sufficient numbers, (6) restricted home range, (7) well-known biology, and (8) sensitive to contaminants. The mink (*Mustela vison*) is a wildlife species that satisfies all of the above criteria. It is a fish-eating mammal that, because of its high trophic status, accumulates numerous contaminants of concern including PCBs, PCDDs and PCDFs, as indicated by numerous field (Haffner et al., 1998; Millsap et al., 2004; Martin et al., 2006a) and laboratory studies (Ringer et al., 1972; Hochstein et al., 1988, 1998, 2001; Tillitt et al., 1996; Halbrook et al., 1999; Bursian et al., 2006a,b,c; Martin et al., 2006b). Agencies and organisations such as the US Environmental Protection Agency, US Fish and Wildlife Service, US National Academy of Sciences, Environment Canada and the Swedish Environmental Protection Agency recognise the mink as a sentinel species (Basu et al., 2007), making it one of the most commonly selected receptors in ecological risk assessments for sites involving aquatic habitats with elevated concentrations of PCBs, PCDDs, PCDFs and related compounds (Blankenship et al., 2008).

Effects of PCBs, PCDDs and PCDFs on Mink

The identification of the mink as a potential sentinel species and its relation to the Great Lakes can be traced back to 1968, when coho salmon collected from Lake Michigan tributaries during the 1967 spawning run and incorporated into mink feed, which was fed prior to and during the breeding and whelping periods on commercial fur farms, resulted in abnormally high kit mortality (Aulerich et al., 1971, 1973). Studies conducted at Michigan State University by Richard Aulerich and associates confirmed that Lake Michigan coho salmon, when fed to breeder mink at 30% of the diet, caused either complete reproductive failure or increased kit mortality. Chlorinated pesticide (DDT, DDT isomers and dieldrin) contamination was ruled out by mink feeding experiments that indicated no reproductive effects at dietary concentrations in excess of what was detected in the fish (Aulerich et al., 1971, 1973). Further analysis of the fish indicated the presence of PCBs at concentrations up to 15 parts per million (ppm) or µg/g tissue.

Figure 16.2. Running Mink. Courtesy Adam Ahlers, University of Illinois.

Subsequent feeding studies by Aulerich and colleagues demonstrated that mink were very sensitive to PCBs. Thirty µg/g of commercial PCB mixtures (10 µg each of Aroclors 1242, 1248 and 1254) in feed proved lethal to adult breeders within four months. Fifteen µg/g feed caused reproductive failure and some adult mortality. Diets containing 10 µg of a commercial PCB mixture (Aroclor 1254)/g feed resulted in reduced weight gain when fed continuously to growing mink. The clinical signs and gross lesions, which included anorexia, bloody stools, fatty livers and haemorrhagic gastric ulcers, of mink fed diets containing the commercial PCB mixture were very similar to the signs noted in mink fed feed containing coho salmon (Aulerich et al., 1973; Aulerich and Ringer, 1977).

Subsequent studies with commercial Aroclor mixtures showed that a dietary concentration of 2 µg Aroclor 1254/g feed adversely affected reproduction based on the number of females whelping and litter size, but similar dietary concentrations of Aroclors 1221, 1242 and 1016 had no effect on reproduction (Aulerich and Ringer, 1977). Based on the results of this study, the no observed adverse effect level (NOAEL) and lowest observable effect level (LOAEL) for Aroclor 1254 based on reproductive impairment is 1 and 2 µg/g feed, respectively. A dietary NOAEL and LOAEL of 10 and 25 µg/g feed has been reported for Aroclor 1016 based on decreased kit weight at 4 weeks of age (Aulerich and Ringer, 1980). Bleavins et al. (1980) reported that a dietary concentration of 5 µg Aroclor 1242/g feed resulted in complete reproductive failure in that no bred females in this group whelped. Because this was the least dietary concentration used, a NOAEL cannot be determined.

A number of mink feeding studies have been conducted utilising fish collected from waters associated with the Great Lakes basin. Hornshaw et al. (1983) reported a reduction in litter size and kit survival through four weeks of age in mink fed diets containing yellow perch from northern Lake Erie and white suckers from Saginaw Bay, Lake Huron. The LOAEL for this study was 0.66 µg total PCBs (tPCBs)/g feed. Heaton et al. (1995) fed ranch mink diets containing up to 40% carp collected from Saginaw Bay, Lake Huron. These diets provided tPCB concentrations ranging from 0.72 µg/g feed (10% carp) to 2.56 µg/g feed (40% carp). The LOAEL in this study was 0.72 µg tPCBs/g feed (10% carp) based on reduced kit survival and body weight. Bursian et al. (2006a) conducted a study similar to that by Heaton et al. (1995) in that mink were fed diets containing carp collected from the Saginaw River, which empties into Saginaw Bay. They reported that inclusion of up to 30% carp in the feed, which provided 1.7 µg tPCBs/g feed, had no significant effect on reproduction and offspring survivability and growth. Diets in the studies by Heaton et al. (1995) and Bursian et al. (2006a) were analysed for individual PCB, PCDD and PCDF congeners, which allowed for calculation of TCDD toxic equivalents. In the Heaton et al. (1995) study, the LOAEL of 0.66 µg tPCBs/g diet corresponds to 16.8 pg TEQ/g diet using 2005 World Health Organization (WHO) TEF values (Van den Berg et al., 2006) and the reproductive NOAEL of 1.7 µg tPCBs/g diet in the Bursian et al. (2006a) study is equivalent to 56.6 pg TEQ/g diet.

One complication of feeding mink diets containing fish collected from contaminated waters is that it is probable that other unaccounted for contaminants are influencing the overall toxicity of the mixture. Furthermore, it is possible that the PPCB/PCDD/PCDF congener profile is influencing toxicity in a way that is not accounted for using the TEF approach. This may explain why Heaton et al. (1995) reported effects on kit survival and growth at dietary tPCB and TEQ concentrations that were 2.5-fold and 3.4-fold less, respectively, than the NOAEL reported by Bursian et al. (2006a). Similarly, Bursian et al. (2006 b) reported that diets containing 3.7 µg tPCB/g feed or 50.4 pg TEQ/g feed provided by carp collected from the Housatonic River, Massachusetts, caused reduced kit survivability, resulting in a LOAEL that is 5.6 and 3 times greater than the LOAEL reported by Heaton et al. (1995) when expressed on a tPCB and TEQ basis, respectively.

There have been a few mink studies conducted with single PCB, PCDD and PCDF congeners that avoid the complexities associated with exposure to a mixture of congeners as well as other contaminants. Hochstein and associates conducted studies that involved exposure of mink to TCDD. The first study (Hochstein et al., 1988) established a 28-day LD_{50} of 4.2 µg TCDD/kg body weight. A 28-day LC_{50} of 4.8 ng TCDD/g feed and a 125-day LC_{50} of 0.85 ng TCDD/g feed was reported in the second study (Hochstein et al., 1998). Beckett et al. (2008) reported that dietary concentrations of 24 and 2.4 ng 3,3',4,4',5-pentachlorobiphenyl (PCB 126)/g feed resulted in complete reproductive failure (2,400 and 240 pg TEQ/g feed, respectively), while a dietary concentration of 0.24 ng PCB 126/g feed (24 pg TEQ/g feed) was the NOAEL. Interestingly, Zwiernik et al. (2009) reported that an equivalent TEQ concentration (242 pg/g feed) provided by 2,3,7,8-tetrachlorodibenzofuran (TCDF) had no effect on reproduction and survivability of offspring. Blankenship and associates (2008) conducted a comprehensive review of mink feeding studies involving PCBs, PCDDs and PCDFs, including those studies mentioned above, in order to derive dietary and tissue residue-based toxicity reference values (TRVs) expressed as TEQs. Developmental and reproductive effects were considered to be ecologically relevant endpoints. Dietary TRVs ranged from 12 to 57 pg TEQ/g feed for the NOAEL and from 50 to 242 pg TEQ/g feed for the LOAEL. The authors stated that the effects of PCDFs could not be accurately predicted from the use of TEQ-based TRVs developed from studies using PCBs or PCDDs.

Figure 16.3. Dental malalignment and increased interdental spacing in mink induced by consumption of a diet containing 24.0 μg PCB 126/kg feed. (A) Inflammation of gingival tissue and moderate spreading of incisors. (B) Extended canines and missing incisors (mandibular and maxillary); gums are bloody and swollen Source: Zwiernik et al., 2011.

Mandibular and Maxillary Squamous Epithelial Proliferation

While effects on mink reproduction and offspring survivability and growth are generally considered to be the important ecologically relevant endpoints (Blankenship et al., 2008), mandibular and maxillary squamous epithelial proliferation in the mink has attracted the attention of regulators as a potential endpoint indicative of comprised health. Render et al. (2000a) observed maxillary and mandibular osteoinvasive squamous epithelial proliferation in 12-week-old mink fed a diet containing 24 ng PCB 126/g feed. Gross examination revealed mandibular and maxillary lesions consisting of swollen tissue of the lower and upper jaws with nodular proliferation of the gingiva and loose teeth with increased gingival surface area. Radiographs indicated osteolysis of the maxilla and mandible, and histological examination documented extensive osteoinvasion by squamous epithelial cells (see Figures 16.3 to 16.5). A subsequent study by Render et al. (2001), in which 6- and 12-week-old mink were fed 24 ng PCB 126 or 2.4 ng TCDD/g feed, verified induction of maxillary and mandibular osteoinvasive squamous epithelial cell proliferation by PCB 126 and demonstrated

Figure 16.4. Osteolysis induced by exposure to PCB 126. (A) Front view of skulls from a control mink (left) and a mink fed a diet containing PCB 126 (bottom). (B) Side view of skulls from a control mink (top) and a mink fed a diet containing PCB 126 (bottom). Source: Zwiernik et al., 2011 (A) ; Render et al., 2000a (B).

its induction by TCDD. This latter study also showed that the lesion could be detected histologically after only two weeks of dietary treatment. Examination of adult female mink that were fed 5.0 ng TCDD/g feed for six months indicated no gross abnormalities of the maxilla or mandible, but histologically there was proliferation of squamous epithelial cells (Render et al., 2000b). The proliferation resulted in focal loss of alveolar bone or osteolysis, but not to the extent that was observed in 6- and 12-week-old mink fed PCB 126 or TCDD. The studies by Render et al. (2000a,b, 2001) suggested that juvenile mink exposed to PCB 126 or TCDD are more susceptible to proliferation

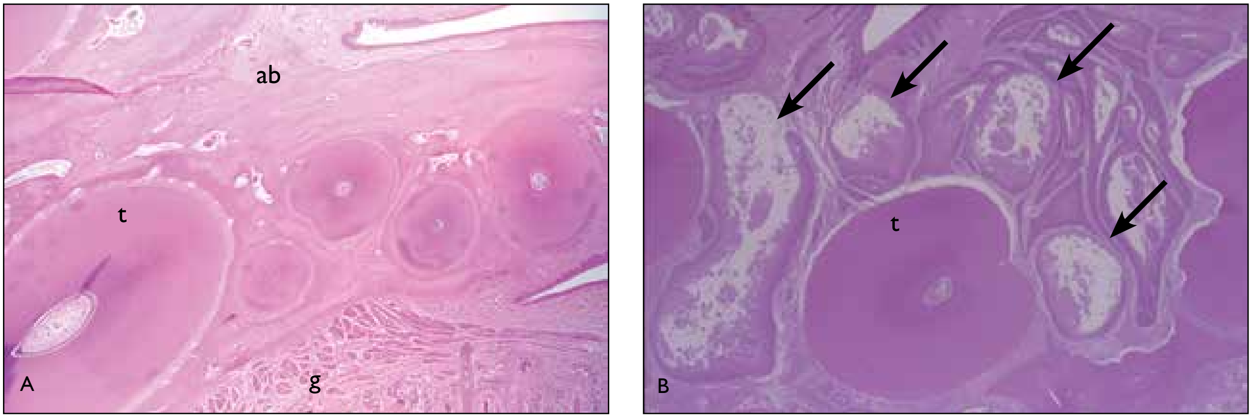

Figure 16.5. Mandibular and maxillary squamous epithelial proliferation in mink exposed to PCB 126. (A) Cross-section through the mandible of a control mink shown for reference. Note the solid appearing alveolar bone (ab) surrounding the teeth (t), which are above the gingiva (g). (B) Cross-section through the mandible of a mink fed a diet containing PCB 126. Note the infiltration of cysts of squamous cells throughout the alveolar bone (arrows), resulting in loosening of teeth. Source: Beckett et al., 2005 (A) ; Zwiernik et al., 2011 (B).

of squamous epithelia than adult mink. Mink exposed to PCBs/PCDDs/PCDFs through consumption of diets containing contaminated fish also had histological evidence of the lesion.

Bursian et al. (2006c) reported that juvenile mink that were exposed from conception through 31 weeks of age to tPCBs concentrations as low as 0.96 µg/g feed (9.2 pg TEQ/g) provided by fish collected from the Housatonic River developed mandibular and maxillary squamous cell proliferation. Maxillary and mandibular squamous epithelial proliferation was evident in four of seven juveniles exposed from conception through to 27 weeks of age to 1.1 µg tPCBs/g feed (36.5 pg TEQs/g) and in six of eight juveniles exposed to 1.7 µg tPCBs/g feed (56.6 pg TEQs/g) derived from carp collected from the Saginaw River (Bursian et al., 2006a).

Beckett et al. (2005) reported on the incidence of mandibular and maxillary squamous epithelial proliferation in mink trapped in the Kalamazoo River area of concern (KRAOC). The Kalamazoo River Superfund site includes approximately 125 km of the Kalamazoo River from the city of Kalamazoo in southwestern Michigan, USA to Lake Michigan. This area became contaminated with PCBs by waste discharge from the recycling and processing of carbonless copy paper. Four of nine mink collected from the KROAC had histological evidence of the lesion, while mink trapped in an upstream reference area did not have the lesion. Lesion severity was positively correlated with hepatic tPCB and TEQ concentrations in mink collected from the KROAC. Hepatic tPCB and TEQ concentrations in mink having the lesion ranged from 2.9 to 6.0 µg/g and 0.21 to 1.3 pg/g, respectively.

Contaminant Load in LGL Mink

Because of recognition of the mink as a sentinel species, there have been efforts to assess spatial and temporal trends of PCBs in mink residing in the Great Lakes basin. Mink trapped in areas adjacent to Lake Erie during the 1970s and 1980s were reported to have PCB concentrations in excess of those causing growth and reproductive effects in controlled exposure studies (Proulx et al., 1987; Haffner et al., 1998). Giesy et al. (1994) assessed TEQ concentrations in fish collected above and below hydroelectric dams on selected rivers that flow into Lake Michigan and Lake Huron and concluded that, on average, if more than 10% of a mink's diet consisted of fish from these rivers, there was an increased risk of deleterious effects. Martin et al. (2006a) compared contaminant concentrations in liver tissue of mink trapped from 1998 to 2003 in the Lake Erie and Lake St. Clair basins to those of mink similarly obtained in 1978/1979. They reported that while concentrations of PCBs and other chlorinated hydrocarbons in mink generally decreased over the past two decades, PCB concentrations tended to increase in western Lake Erie mink over the same time period. Furthermore, hepatic PCB concentrations in mink were within the range associated with reproductive impairment, as determined from captive mink studies, in approximately 12% of all animals collected from the Lake Erie and St. Clair basins overall, and in almost 40% of individuals from western Lake Erie.

Emerging Contaminants of Concern

In addition to PCBs, PCDDs and PCDFs, there are emerging contaminants of concern that have been detected in the LGL environment. One such group of chemicals is the brominated flame retardants (BFRs), which are chemical compounds that inhibit the combustion of organic materials by scavenging free radicals that are involved in the combustion process (D'Silva et al., 2004; Hites, 2006). BFRs are incorporated in a wide variety of materials including paints, plastics, textiles, furniture and electronics by covalent bonding to the polymer or by addition into the final product (Ward et al., 2008). One type of BFR is the polybrominated diphenyl ethers (PBDEs), which gained prominence after the manufacture of polybrominated biphenyls (PBBs) was stopped in the US and Canada in the late 1970s (Hites, 2006). PBDEs are a class of additive BFRs made up of 209 possible congeners containing between 1 and 10 bromine atoms (Alaee et al., 2003). Of these 209 congeners, 23 are of environmental significance (Ward et al., 2008). Until early 2005, three commercial PBDE mixtures were widely distributed for use in North America: Pentabromodiphenyl ether (pentaBDE), octabromodiphenyl ether (octaPBDE), and decabromodiphenyl ether (decaBDE). Due to public concerns and for economic reasons, the only North American producer of octaBDE and pentaBDE voluntarily stopped producing these products in December 2004. Despite the fact that the use of penta- and octaBDE products has been effectively eliminated in North America, these chemicals will continue to enter the environment through the disposal of PBDE-containing electronics and furniture (Hites, 2006; Ward et al., 2008). In contrast, there are no restrictions on the decaPBDE commercial mixtures, which largely dominate the demand and use of PBDEs as an additive flame retardant in the global market (Gauthier et al., 2008).

More than 70,000 metric tons of PBDEs have been produced annually worldwide, 50% of which have been used in the US and Canada, including almost all of the pentaBDE manufactured (Renner, 2000; Hites, 2006). The major source of PBDEs in the LGL is atmospheric deposition (Li et al., 2006; Ward et al., 2008). PBDEs also enter the LGL through recycling and disposal of products containing PBDEs (Watanabe and Sakai, 2003; Ward et al., 2008), manufacturing output and fluvial deposition from tributaries (Samara et al., 2006; Ward et al., 2008) and wastewater treatment plants (La Guardia et al., 2007; Ward et al., 2008). Long-term studies of birds and fish in the LGL have indicated that PBDE burdens in wildlife have increased exponentially (Norstrom et al., 2002), doubling approximately every three years (Zhu and Hites, 2004). Recent trends indicate that burdens may now be levelling off in some of the lakes (Luross et al., 2000; Zhu and Hites, 2004), but if the temporal trajectory of PBDE accumulation in the LGL is maintained, concentrations of PBDEs could soon exceed the upper limit of safe levels of consumption of contaminated fish (Ward et al., 2008).

Because the mink is a fish-eating mammal that accumulates PCBs, PCDDs and PCDFs, it was not unexpected that structurally similar PBDE congers were detected in livers of animals fed diets containing fish collected from the LGL basin. Bursian et al. (2006a) reported that 6-week-old mink kits whelped by females fed diets containing 30% carp collected from the Saginaw River had an average hepatic total PBDE concentration of 23 µg/kg. The predominant congener was BDE-47 (89%) followed by BDE-100 (5%) and BDE-209 (4%). Both BDE-47 and BDE-100 are components of DE-71, which is a commercial penta-BDE mixture, while BDE-209 is the principal component of the deca-BDE commercial mixture DE-83R.

In response to the demonstration of bioaccumulation of PBDE congeners in mink, a series of studies were conducted to determine the effects of dietary DE-71 in this sentinel species. In the initial study, juvenile mink were fed diets containing up to 10 µg DE-71/g feed for eight weeks, which resulted in reduced feed intake and decreased body mass at dietary concentrations of 5 and 10 μg/g feed. In addition, there was an increase in antibody production, an increase in relative masses of the spleen, adrenal glands and liver and induction of liver microsomal ethoxyresorufin-O-deethylase (EROD) activity. Spleens of mink exposed to 10 μg DE-71/g feed had significantly increased germinal centre development and incidence of B-cell hyperplasia. Haematocrits in mink fed 5 and 10 μg DE-71/g feed were significantly less than in controls, while the percentage of neutrophils increased and the percentage of lymphocytes decreased (Martin et al., 2007).

To assess the effects of DE-71 on mink reproduction and development at environmentally relevant concentrations, adult female mink were fed diets containing 0, 0.1, 0.5 or 2.5 μg DE-71/g feed from 6 weeks prior to breeding until weaning of kits at 6 weeks post-parturition. Offspring in each group were maintained on their respective diets for an additional 27 weeks. A dietary concentration of 2.5 μg DE-71/g feed resulted in complete reproductive failure. Developmental effects in offspring were evident in 33-week-old juveniles, which were more sensitive to effects than their respective dams. At 0.5 μg DE-71/g feed, total triiodothyronine (T3) was significantly decreased in both male and female juveniles while thyroid follicular epithelium cell height was elevated when compared with controls. The NOAEL and LOAEL for T3 disruption based on juvenile hepatic PBDE concentrations were 1.2 and 6.4 μg/g. Hepatic EROD activity was significantly induced in all exposed offspring at 33 weeks (Zhang et al., 2009). This study also demonstrated that environmentally relevant exposures to DE-71 did not affect key parameters of the cholinergic neurotransmitter system in the brain of ranch mink (Bull et al., 2007).

These studies indicate that exposure to a commercial pentaBDE mixture can cause deleterious effects in mink. Biomonitoring of wild mink in the LGL region indicated that most populations have hepatic concentrations of total PBDEs that are currently less than those expected to affect thyroid hormone homeostasis, but margins of safety are small and hepatic PBDE concentrations in mink around Hamilton Harbour, Ontario exceeded the NOAEL for T3 disruption (Zhang et al., 2009).

Conclusions

Chemical contaminants have had deleterious effects on numerous wildlife species residing in the LGL basin over the past half-century. In many cases, awareness of the negative impact has resulted in a reduction or discontinuation of the use of many of these chemicals. This in turn has led to a partial, if not full, recovery of certain species. We should point out, however, that some of these historical contaminants, such as PCBs, continue to occur in concentrations that can induce effects in sensitive species such as the mink, despite the fact that production of PCBs ceased over 30 years ago. Furthermore, contaminants that have been detected relatively recently, such as the brominated flame retardants, continue to increase in the LGL, approaching concentrations that have been demonstrated in laboratory studies with mink to cause potentially deleterious effects. Considerably more research is needed to better characterise the occurrence of these chemicals in the LGL and their effects on wildlife so that effective measures can be taken to reduce the impact of the emerging contaminants of concern on this vital resource.

Contaminants and Health of Beluga Whales of the Saint Lawrence Estuary

17

Daniel Martineau
University of Montreal, Saint-Hyacinthe, QC, CAN

Defining the causes of mortality in wild animal populations is difficult, largely owing to their widespread distribution and poor spatial delineation of these populations. It is especially problematic in marine mammals, because their environment is opaque to direct visual observation. Because of these limitations, the respective roles played by natural factors and human activities in mortality remain intricate. An exception is the beluga population that inhabits a stretch of the SLE roughly centred on the mouth of the Saguenay River in Quebec, eastern Canada, only 500 km from Montreal. This estuarine habitat is reminiscent of that used by Arctic belugas, which enter the estuaries of Arctic rivers in summer. The SLE drains the GL, the most industrialised part of North America, and thus receives a heavy input of contaminants. The slow circulation of the SL water exacerbates problems of environmental contamination. SLE beluga became isolated from their original Arctic population at the end of the last glaciation, 10,000 years ago, when the region emerged isostatically, relieved from the weight of glacial ice.

This origin is similar to that of Baltic harbour seals (*Phoca vitulina*), which were probably imprisoned in the Baltic 8,000 years ago. Because both populations have been isolated for a thousand years from Arctic populations, they show a high degree of inbreeding, expressed by lack of genetic variation (Murray et al., 1999; Patenaude et al., 1994). Other factors shared by both populations and that may have contributed to population bottlenecks are that both have been the object of extensive hunting in the early 20th century, even being the object of bounties by the corresponding government. In addition, both populations, because they are top predators in the food chain, have been contaminated by high levels of stable lipotrophic (fat-loving) industrial contaminants, many of which are organochlorine compounds that may decrease immune functions (which may have contributed to a severe viral epidemic that struck Baltic harbour seals at the end of the 1980s), endocrine disruption, lower rates of reproduction (Helle and Olsson, 1976; Lair et al., 1997; Martineau et al., 1987, 1988; Mikaelian et al., 2000; Reijnders, 1986).

SLE beluga, which constitute the southernmost beluga population worldwide, are unique by their accessibility to investigation and geographical isolation from the Arctic, the natural habitat of most beluga. The isolation of SLE beluga from Arctic populations has been confirmed by genetic analysis and by the distinctly higher levels of industrial contaminants found in their tissues (Gladden et al., 1999).

Together, the position of SLE beluga at the top of the food chain, its high body lipid content (around 40% of body weight) and its long lifespan[1] explain the high con-

[1] up to 70 years according to a recent paper (Stewart et al 2006)

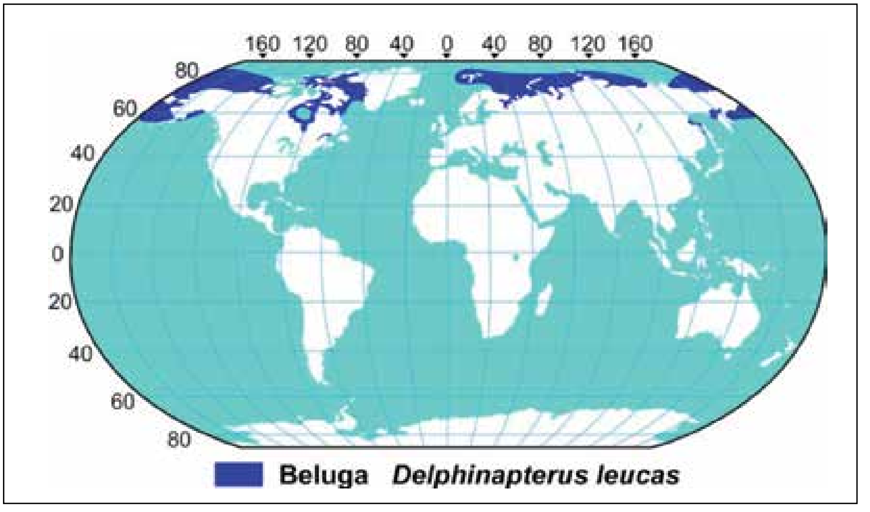

Figure 17.1. Worldwide distribution of beluga whale (*Delphinapterus leucas*). Map by Uko Gorter for American Cetacean Society (http://www.acsonline.org/factpack/BelugaWhale.htm).

centrations of persistent lipophilic contaminants in its tissues. In contrast, contaminant levels in fin whales visiting the area in summer are much lower, partly because these animals mostly eat plankton, and thus occupy a lower position in the food chain (Metcalfe et al., 2004).

The SLE beluga population has dwindled from an estimated 10,000 at the beginning of the 20th century to the current estimate of anywhere between 400 and 1200 animals (the maximum number of animals ever counted directly is 459) (Kingsley, 1998). In 1980, SLE belugas received the status of endangered species from the Canadian Government (Sergeant, 1986). To explain the decline and apparent lack of recovery, a study was initiated in 1982 to carry out systematic post-mortem examination of dead SLE beluga that drift ashore, in order to determine the cause of death of these animals, measure tissue levels of chemical contaminants and examine the hypothesis that contaminants contribute to death.

To evaluate the diseases and causes of mortality in a human population, the logical places to look are hospitals and morgues. Similarly, to evaluate diseases and causes of death in wildlife, one has to examine animals found dead in the wild. The bodies of juveniles and newborns are harder to locate than those of adults, and thus some are likely missing from our data. There is no other obvious source of bias beside emaciation. The latter could be caused by chronic diseases, and would make carcasses less buoyant, which would make them more likely to sink and to be recovered. From 1983 to 1999, an average of 15 beluga carcasses were reported stranded every year and of these, about half were examined at the College of Veterinary Medicine of Université de Montréal because they seemed in a reasonable state of preservation (Martineau et al., 2002). This probably does not represent the entire mortality, because the territory is immense and the shoreline is rarely visited by people in winter. Using a realistic annual 6% death rate from a population of 960, Kingsley (2002) estimated the mortality rate at 58 deaths/year. Thus with an average of 15 carcasses reported per year, about 43 carcasses or 74% of dead animals escape detection. Other researchers have found higher annual death rates in free-ranging tooth whales, closer to 10% (Stolen and Barlow, 2003), which would result in around 100 beluga carcasses stranded every year, implying an even lower recovery rate.

The three primary causes of death of SLE beluga are metazoan parasites, (22%), cancer (18%) and infectious agents (bacterial, viral, or protozoan, (17%) (Martineau et al., 2002). Cancer caused the death of 27% of adults, a rate of cancer higher than in any other population of wild terrestrial or aquatic mammals with the possible exceptions of virally induced tumours in rodents and woodchucks. Epithelial cancers of the gastrointestinal (GI) tract were strikingly predominant (Martineau et al., 2002). In addition, two additional cancers involved the liver, and one case involved the salivary glands (adenocarcinoma) (Girard et al., 1991).

SLE beluga are commonly affected by multisystemic infections with agents that have generally been associated with immune suppression in humans and other animals. For instance, infections with opportunistic agents such as *Aeromonas* sp, *Edwardsiella tarda*, *Nocardia* spp., *Toxoplasma gondii*, and viruses such as herpes virus and papilloma virus have all been found in SLE beluga (De Guise et al., 1994; Martineau et al., 1988; Mikaelian et al., 2000).

Persistent Organic Pollutants

Persistent Organic Pollutants (POPs) are defined as global toxic contaminants found 'distant from sources' for long periods, and accumulate in the fat of humans and free-ranging mammals. Many countries have decided to restrict POPS fabrication and distribution through the Stockholm Convention on Persistent Organic Pollutants[2] and the United Nations ECE POPS Protocol to the Convention on Long-range Transboundary Air Pollution[3]. Thus PCBs and DDT are sometimes designated 'legacy' contaminants because their production has stopped in many parts of the world. Many POPs, such as PCBs, DDTs and PBDE, are organochlorine compounds (OCs), composed of paired carbon rings with chlorine or bromine attachments (see chapters 13 and 16). SLE beluga are contaminated by all these compounds, as well as by other OCs such as chlordane, hexachlorocyclohexane (lindane insecticide (HCH)), their metabolites and heavy metals (Martineau et al., 1988; Martineau, 1987; Muir et al., 1996; Wagemann et al., 1990). These large halogen atoms protect the carbon rings from the normal enzymatic degradation (metabolism) that takes place inside vertebrate organisms (most often in the liver and intestine). As a general rule, the resistance to degradation is proportional to the number of halogen atoms composing the molecule. Together, this protection along with the 'fat loving' properties of these compounds explain their lifelong accumulation in lipids.

Figure 17.2. Post-mortem examination of a beluga whale found dead on the the St Lawrence Estuary shoreline. College of Veterinary Medicine, University of Montreal. Source: Martineau et al., 2002. Reprinted by permission from Environmental Health Perspectives.

Most degradation-resistant chemicals find their way into water. Thus it is not surprising that OCs released from leaking equipment, poor storage conditions, accidental spills (as for PCBs) or deliberately discharged into the terrestrial or aquatic environment (as for DDT) have all ended up in water. Because the SLR drains the most industrialised area of North America, SLE belugas are among the most heavily OC-contaminated marine mammals, surpassing only transient killer whales of the Canadian Pacific coast and striped dolphins from the Mediterranean (Martineau et al., 1987; Ross et al., 2000). SLE beluga adipose tissues have 25 and 32 times more total PCB and

[2] http://www.pops.int/

[3] http://www.unece.org/env/lrtap/pops_h1.htm (United Nations Economic Commission for Europe)

total DDT, respectively, than Arctic beluga (Table 17.1). Concentrations found in SLE beluga are also higher than those found in humans exposed to PCB during the Yusho industrial accident (Letcher et al., 2000 a, b).

Pathological Effects on the Endocrine System

Many OCs and their metabolites severely damage the adrenal glands in animals and people. Degenerative and proliferative changes consistent with chronic stress and dioxin-related POP (DRPOP) intoxication are commonly observed in the adrenal cortex and medulla of SLE and Western Hudson Bay beluga whales, and the severity of these lesions increases with age in both populations (Lair et al., 1997). Adrenocortical cysts are also often found in SLE beluga. These are rare in other marine mammals except in white-sided dolphins (Geraci and St. Aubin, 1979; Lair et al., 1997). OC metabolites may be responsible for these cysts. Most strikingly, a DDT derived compound, o,p'DDD, has long been used in human and veterinary medicine to destroy tumour or hyperplatic adrenocortical cells, when these cells secrete pathological amounts of cortisol (Cushing syndrome). In grey and harbour seals from the Baltic Sea, adrenocortical hyperplasia has been attributed to PCB and DDT contamination based on epidemiological data (Bergman and Olsson, 1985; Olsson, 1994; Olsson et al., 1994). Both SLE beluga and Atlantic white-sided dolphins have been exposed to high levels of adrenotoxic OC metabolites for decades, suggesting that OCs may be responsible for the adrenal lesions seen in these two species (Martineau et al., l987, 2003; McKenzie et al., 1997; McKinney et al., 2006a,b; Muir et al., 1996a,b; Troisi et al., 1998).

Proliferative and degenerative lesions (adenomatous hyperplasia and follicular cysts) have been found in the thyroid of SLE beluga (Mikaelian et al., 2003). Similar lesions were found in Arctic beluga but the absence of older age groups in an Arctic population precluded a comparison between older whales from the two populations. The thyroid gland has long been known to be a major target of POPs toxicity. Thyroid hyperplasia (e.g. an increased number of cells) has been found in fish and birds contaminated with PCBs in the Great Lakes (recall that the GL are drained by the SLE) (Campbell et al., 2003; Fox et al., 2007). The thyroid gland produces two major hormones, T3 and T4, called thyroxines. These hormones are responsible for maintaining body temperature by regulating heat production through oxygen consumption, and maintaining and using adipose tissue for increased energy demand. The blubber of marine mammals is a thick, deep skin layer composed of adipose tissue. Thus the thyroid glands probably play a major role in the use and maintenance of blubber, which is critical for insulation, heat production, energy storage and buoyancy. Thyroid functions are especially important for Arctic marine mammals whose body must produce heat in cold water, a medium that would quickly suck every calorie produced if it were not for the blubber. The role of the thyroid gland is further complicated in marine mammals by these animals slowing down their metabolism in order to dive at great depths (this conflicts with the need for high heat production levels in a cold medium). For all the above reasons, thyroid integrity is most likely central for the survival of beluga whales. Finally, PCBS and PBDE are known to affect brain development through their action on the thyroid gland and cetaceans might be particularly vulnerable in that regard given their very high relative brain size, second only to that of humans (Marino, 1998; Montie et al., 2009).

Immunosuppression

PCBs have long been demonstrated as immunosuppressive compounds in laboratory and farm animals, and have been associated epidemiologically with immunosuppression in marine mammals (Bull et al., 2006; Jepson et al., 2005; Safe, 1994; Thomas et al., 1978). The finding of generalised infections by microorganisms that normally do not cause disease in immunocompetent animals has been taken as circumstantial evidence of PCB-induced immunosuppression (Inskeep et al., 1990; Martineau et al., 1988). Experimental evidence has supported this hypothesis. Young harbour seals fed with OC-contaminated fish for 2.5 years showed compromised immune functions when compared with a group of seals fed non-contaminated fish (Van Loveren et al., 2000).

Table 17.1. Mean PCB Levels (ng/g) in blubber of male belugas from selected stocks (Modified from Andersen et al., 2001a)
[1]Muir et al. (1996a); [2]Martineau et al. (1987); [3,4]Muir et al. (1997); [5]Andersen et al. (2001); [6]Stern et al. (1994); [7]Krahn et al. (1999); [8]Muir (1994, 1996) compiled by Muir et al. (1997); [9]Becker et al. (1995) compiled by De March et al. (1998); [10]Muir et al. 1990; [11]Krahn et al. (1999)
[a]Only geometric mean was available.

Location:	Average Level	Range
St. Lawrence Estuary (Canada) [1,2a]	78,900	
SE. Baffin Island (Canadian Arctic) [4]	6,794	± 2,171
S. Hudson Bay (Canada) [4]	6,768	± 2,346
N. West Greenland (Nuussuaq) [6]	5,580	± 2,500
E. Chukchi Sea (Alaska) [7]	5,200	± 900
Svalbard (Norwegian Arctic)[5]	5,100	± 1,870
Mackenzie Delta, Beaufort Sea (Canadian Arctic) [8]	5,010	± 1,618
E. Bering Sea (Alaska) [9]	4,170	± 644
Jones Sound (Canadian High Arctic) [10]	2,530	± 570
Cook Inlet (Alaska) [11]	1,490	± 700

OC-induced immunosuppression has been suspected to play a role in rendering harbour seals more sensitive to phocine morbillivirus in an epizootic that has caused more than 20,000 deaths in these animals in 1988 in the Baltic Sea, whereas in contrast, grey seals were not affected by the epizootic even though they were infected by the virus. The resistance of grey seals to morbillivirus may be explained by the fact that functions of grey seal leukocytes (white blood cells) are less affected by PCBs than those of harbour seals (Hammond et al., 2005). These findings are highly relevant for SLE beluga for several reasons: 1) PCB mixtures affect beluga lymphocyte proliferation and phagocytosis *in vitro*; 2) SLE beluga have no antibodies against morbilliviruses (Mikaelian et al., 1999); 3) pilot whales (*Globicephala melaena* and *G. macrorhynchus*) occasionally enter the beluga habitat and are asymptomatic carriers of morbilliviruses, and could thus transmit the virus to immunologically naïve beluga, a role similar to that grey seals are thought to have played in the transmission of phocine morbillivirus to harbour seals (Duignan et al., 1995; Kingsley and Reeves, 1998; Mikaelian et al., 1999).

Based on the results from experimental exposure in aquatic mammals (minks, otters, seals) and *in vitro* studies on dolphin lymphocytes, Kannan et al. (2000) estimated that marine mammals are likely to start suffering from the adverse effects of PCBs at a blubber concentration > 17 μg/g. Examining harbour porpoises stranded in the UK, Jepson et al. (2005) supported these findings: in a series of porpoises whose blubber total PCB concentration was > 17μg/g, animals dead of infectious diseases had higher total PCBs levels than animals dead from trauma. Below that concentration, there was no correlation, suggesting that PCB-induced immunosuppression favours infections in porpoises with concentrations > 17 μg/g. Using the same series of animals, Bull et al. (2006) showed that harbour porpoises with a total blubber PCB concentration > 50 μg/g showed a significant, positive association between PCB levels and the number of gastric nematodes. In the same series of porpoises, there was a 2% increased risk of being affected by an infectious disease for each 1 mg/kg increase in blubber, and the risk was doubled risk at around 45 μg/g lipid (Hall et al., 2006). Importantly, average PCB concentrations in the SLE beluga population are higher than these putative thresholds (Table 17.1). The threshold in PCB concentration might be even lower in beluga because the high lipid percentage of the beluga body makes the total body burden of beluga higher than that of the leaner porpoises.

One approach to study the impacts of contaminants on immune function is to measure the *in vitro* response of immune cells from a presumably 'normal' population to pollutants added in concentrations identical or similar to those found in the tissues of contaminated free-ranging animals from the same species. Using this approach, the proliferative response of lymphocytes from Arctic beluga to mitogens and their spontaneous proliferation were impaired *in vitro* by exposure to p,p'-DDT and PCB 138 concentrations similar to those found in SLE beluga tissues (PCB 138 is one of the most abundant PCB congeners present in SLE beluga tissues) (De Guise et al., 1998). Because beluga and other marine mammals are not contaminated with one or two pollutants but rather with a complex mixture of congeners and distinct compounds, De Guise's laboratory has also explored the effects of mixtures of different OC (Levin et al., 2004, 2005a, b, c; Mori et al., 2006).

Temporal Trends in Contaminant Loads

The use of PCB and DDT was banned in Europe and North America in the mid-1970s. PCBs are still in use in Russia, however and γ-HCH (lindane) is in restricted use in North America. In addition, legacy compounds such as toxaphene, DDT and PCBs continue to leach into waters from contaminated soils, explaining the continued elevated concentrations of these contaminants in marine mammals tissues (Braune et al., 2005). In SLE beluga, levels of most well known OC have decreased at least 2-fold between 1987 and 2002 with the important exceptions of chlordane, Mirex and γ-HCH. DDT metabolites have not decreased. In Arctic Canadian beluga, PCB levels declined 1.7 to 2.8 fold depending on PCB congeners considered from 1982 to 2002. In contrast to concentrations of the parent compounds, temporal trends in tissue levels of PCB metabolites have not been monitored in Canadian beluga populations (Lebeuf et al., 2007), a significant gap in our knowledge, since the toxicity of PCB metabolites may be higher than that of the parent compounds.

Polycyclic Aromatic Hydrocarbon Contamination

SLE beluga are heavily contaminated by polycyclic aromatic hydrocarbons (PAHs), a large family of compounds produced by the incomplete combustion of organic molecules (Martineau et al., 1988). PAHs, once metabolised by cytochrome oxidase enzymes, are degraded in metabolites that are unstable, strongly reactive and mutagenic, explaining why the PAH family includes members such as benzo[a]pyrene (BaP), which are powerful carcinogens based on their demonstrated carcinogenicity in rodents (IARC) (Fitzgerald et al., 2004).

The SLE waterway receives the effluent from much of northeastern North America, one of the most industrialised regions of the world. In addition, 60 miles (100 km) upstream and upwind of the SLE beluga habitat, on the Saguenay River shore[4], is a large complex of aluminium smelters developed in that region as early as 1926 because of the access to the sea and the availability of vast and cheap amounts of hydroelectricity, two conditions required for aluminium production. Because of this rare conjunction of characteristics, 4% of world aluminium production[5],[6] takes place in the Saguenay region. Aluminium is produced by the electrolysis of alumina (e.g. aluminium oxide) using the Söderberg method, a process that releases huge amounts of PAHs into the atmosphere. Alumina itself is chemically extracted on-site (which generates abundant liquid residues) from bauxite, a rock absent from Canadian soil. Because the Saguenay River provides direct access to the sea, bauxite is shipped directly from Australia, Brazil, Guinea and Jamaica to the Saguenay smelter. In addition, electricity is inexpensive for the aluminium smelters because of generous discounts provided by the Quebec state-run power company, and because the aluminium smelter company has operated its own private hydroelectric dams for almost a century. The Söderberg process, used by the smelter until 2004 for extracting aluminium from alumina, is highly polluting because electrolysis involves the passage of strong electrical current through an anode. This is a compact mass of tar which burns when the current passes through, releasing large amounts of PAHs into the atmosphere (other metallurgical processes, not only the Söderberg process, produce PAH, but generally in lesser amounts).

[4] The Saguenay fjord is a 104-km long inland extension of the SLE, 200-300 m deep, having strong tidal currents at its mouth. It is the only fjord on the North American mainland.

[5] 63% of Canada's production

[6] The production of 1 tonne of aluminium requires 13 kilowatts. The aluminium company in the Saguenay region produces 1950 megawatts annually using its own dams, that is, roughly enough electricity for 3 medium-sized cities (a medium-sized city of 100,000 inhabitants consumes about 600 megawatts/year).

Thus, the light weight of aluminium, often promoted as an energy saver for motor vehicles, came in fact with a high environmental cost until very recently, when the Söderberg process was replaced (partly) by a different process using prebaked anodes. Note that the amount of PAHs currently released into the environment has been monitored solely by the smelter since 1998, e.g. there is no independent monitoring by the government or by any independent organisation since 11 years. Two chemists at Canada's Department of Fisheries and Oceans estimated that the local aluminium smelters have released 40,000 tons of PAHs downwind into the Saguenay river watershed since 1926 (Smith and Levy, 1990). PAHs released by the smelters have contaminated the Saguenay river sediments along with beluga, and the aquatic and terrestrial animals that live in that region (Blondin et al., 1992; Lun et al., 1998; Martel et al., 1986; Martineau et al., 1988; Shugart et al., 1990; Smith and Levy, 1990; White et al., 1997).

Beluga are among the rare species of tooth whales that feed significantly on invertebrate animals living in sediments, particularly on annelids (worms), and these annelids have been contaminated by PAHs. Unlike PCBs, PAHs are biodegraded quickly once released into the environment but they may accumulate in benthic annelids because some annelid species are naturally deprived of CYP enzymes (Rewitz et al., 2006). The metabolites resulting from CYP-mediated degradation of PAHs are often more toxic or carcinogenic than the parent compounds. Within the cells (generally but not exclusively liver cells) of contaminated animals/human, many pollutants, among which PAHs and PCBs, induce the proliferation of smooth endoplasmic reticulum, the cell organelle in which CYPs reside. Thus the generation of carcinogenic PAHs metabolites is accelerated by the ingestion of PAHs and PCBs.

It is likely that beluga become contaminated with PAHs by ingesting contaminated annelids. PAHs have been detected in *Nereis virens*, a benthic annelid living in the sediments of the SLE beluga habitat (E. Pelletier, manuscript in preparation). The daily amount of fish ingested by an adult beluga is equivalent to 2 to 3 % of its body mass, i.e. between 13 and 20 kg of fish (Robeck et al., 2005). A series of beluga was dissected in the 1930s to determine whether beluga eat significant amounts of commercial fish species (the government was responding to fishermen's complaints that beluga competed with fisheries) (Vladykov, 1946). Worms found in their stomachs were identified, weighed and counted. Twenty-five to nearly 1,400 worms were present in individual stomachs; individual worms weighed 1.3 to 9.8 g[7] as determined from worms collected live in the sediments of that region. Thus roughly, the stomach of a single beluga could contain up to 14 kg of worms. On the average, *Nereis* contains 0.68 µg/kg of BaP. Thus 14 kg of worms contain roughly 9.5 µg of BaP.

It has been estimated that 5.6 µg of BaP is the maximum daily amount of BaP that could be ingested by a 70-kg human adult to 'safeguard human health' (Fitzgerald et al., 2004). Based on this estimation, the maximum amount of BaP 'safely' ingested daily by a 900-kg beluga would be 72 µg, an amount about 8 times larger than that contained in 14 kg of worms. However, the local smelter claims that PAHs emissions into the Saguenay River have decreased about 8-fold since the 1970s, implying that BaP concentrations in worms were 8-fold higher at that time. Thus the BaP dose ingested by a 900 kg beluga in 14 kg of worms is near the estimated maximal daily dose 'safely ingested' by a beluga (although it is not known over what period this amount of worms would have been eaten).

Local aluminium workers are affected by high prevalences of urinary bladder and lung cancers, and these have been epidemiologically related to PAH exposure within the smelter. Because the epidemiological relationship between PAH exposure and these cancers is particularly strong[8], workers affected by urinary bladder cancers and exposed for more than 20 years are now compensated by the Quebec Worker Compensation Board (Tremblay et al., 1995). Increased cancer rates have also been observed in the rest of the human population living in that area, where drinking water is taken mostly from surface waters, and is also contaminated by PAHs (Martineau et al., 2002) (Figure 17.3).

The aluminium industry has argued that there is no proof that PAHs cause cancer in beluga whales. This argument ignores the historical and, more importantly, the

[7] Also in Vladykov "Although beluga of both sexes eat *Nereis*, lactating or pregnant females show a marked preference for this type of food"

[8] Aluminium workers in the Saguenay region have up to 42 times more urinary bladder cancer than the general population (Tremblay et al., 1995; Theriault et al., 1984)

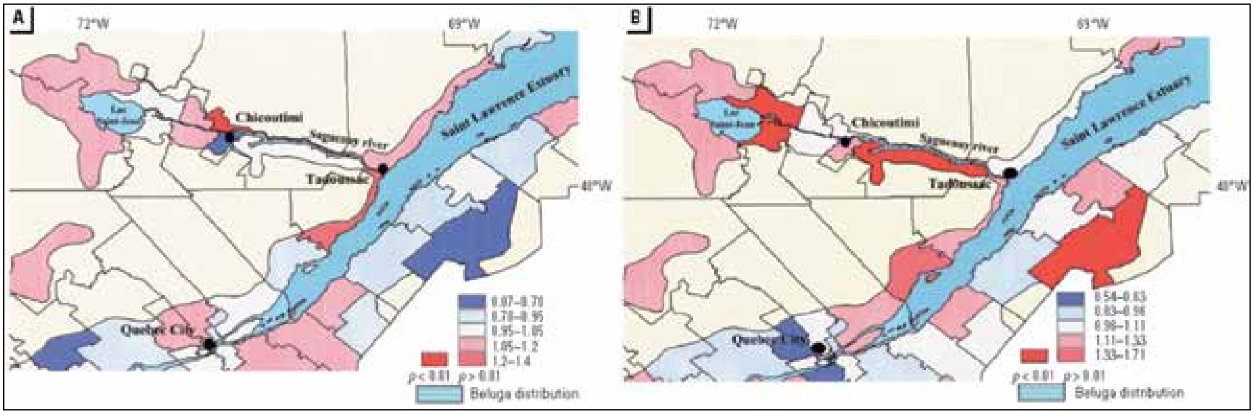

Figure 17.3. Distribution of beluga in the SLE and incidence of standardised rate ratio for digestive system cancer in (A) men and (B) women in the Saguenay River area, Québec. Source: Martineau et al., 2002. Reprinted by permission from Environmental Health Perspectives.

experimental evidence provided by laboratory rodent models and *in vitro* cellular and molecular models used to demonstrate PAH carcinogenicity. These results have been traditionally extrapolated to humans because people (like rodents and beluga whales) are mammals and as such, share most cellular machineries with other mammals, including beluga (Fitzgerald et al., 2004).

That argument – roughly that the absence of evidence is equivalent to the evidence of absence – is also reminiscent of that used by tobacco companies, which have long invoked the absence of proof regarding whether tobacco smoke causes lung cancer, despite overwhelming evidence supporting the carcinogenicity of tobacco smoke in animal and *in vitro* models (the weight of evidence now points at PAHs as the major culprit for the carcinogenicity of tobacco smoke; Alberg et al., 2005). The role of PAHs has also been demonstrated in the aetiology of GI tract cancers in chronic carcinogenicity studies in rodent models (reviewed in Martineau et al., 2002).

Emerging and Recently-Emerged Contaminants

Butyltin (Bt) and its metabolites have been found in SL beluga (Saint-Louis et al., 1999). Pelletier's group also recently detected butyltin in invertebrates living in Saguenay sediments (Viglino et al., 2006). This compound is the active component of antifouling paints used to protect ship hulls. It is immunotoxic, hepatotoxic, and a powerful endocrine disruptor (Ogata, 2001). The toxicity of butyltin metabolites remains to be determined. In the Saguenay River, gastropods naturally exposed to butyltin had significant sexual changes ('imposex'). Annelids recovered from the Saguenay sediments contained an average of 289 ng Sn/g of total Bt compounds, implying the possible ingestion of 4 mg of these compounds (in 14 kg of worms) by a beluga (see PAHs discussion; Viglino et al., 2006). For a 1,000-kg beluga, this represents a 4 μg/kg dose, which is 16 times higher than the maximum dose tolerable in humans, based on immunological toxicity (0.25 μg/kg) (Penninks, 1993). Liver concentrations measured in SL beluga approach those found to be lethal for the lymphocytes (white blood cells) of Dall's porpoise *in vitro* (100 ng/g). In addition, liver lesions caused by Bt in rats are consistent with those found in SL beluga (Martineau et al., 2003).

As PBDEs are used to reduce the risk of fire (in polyurethane foams, computer casings, textiles used in furniture and automobiles), they must be quickly released from burning material to rapidly make oxygen unavailable to flames. Thus these compounds are only mixed with, not chemically bound to, plastic. Consequently, they readily leach out from these materials, ending up in the environment. Canada's regulations on PBDE production and use are lagging, however (Health Canada, 2007). Thus it is not surprising that the concentration of these compounds in wildlife is increasing, in contrast to other OC such as PCBs, the production of which has generally been banned.

PBDE are found in SL beluga tissues at lower concentrations than PCBs but their concentrations are rapidly increasing (Hobbs et al., 2003; Lebeuf et al., 2004; Letcher et al., 2000a; McKinney et al., 2006a; Metcalfe et al., 1999). Between 1988 and 1999, concentrations have doubled in males and females every 3 and 2 years respectively (Lebeuf et al., 2004). In contrast to PCB levels (males have higher PCB levels than females), males and females have similar PBDE levels, most likely because of the relatively recent massive input of these contaminants in the beluga's environment, which has not given sufficient time for females to transfer their load to newborns. In Arctic beluga, the recent temporal increase in PBDE levels has been similar (6.5 fold from 1982 to 1997, 1.6 fold between 1998 and 2001 (de Wit et al., 2006). In Arctic ringed seals, levels increased 9-fold between 1981 and 2000 (Ikonomou et al., 2002).

The toxicity of PBDEs probably adds to that of PCBs, because both groups of compounds affect the development of the nervous system and lower vitamin A and thyroxin serum levels. In by-caught Baltic and North Seas porpoises, thymic atrophy has been associated with elevated PBDE levels (Beineke et al., 2005). The rapid increase in PBDE levels in beluga tissues along with the current use and production of these compounds and the lack of regulations almost guarantee that PBDE concentrations will continue to increase (Lebeuf et al., 2004).

Among fluorinated organic compounds (perfluoroalkyl compounds (PFC)), two have been studied in some detail: PFOA, a major ingredient in the manufacture of Teflon, composed of a 8-carbon chain ornate with fluorine atoms; and PFOS, very commonly used as a surfactant in fabric stain repellents (3M Scotchgard). In 2000, worldwide production of PFOS by the US 3M company was around 3700 tonnes. In 2001, 3M discontinued PFOS production. At the time of writing, however, the compound was still being produced in Germany, Switzerland, Russia and Japan (OSPAR, 2006).

Although PFCs are more polar than PCBs, and therefore much less lipotrophic (fat-loving), they are biomagnified from fish to odontocetes. They do not accumulate in fat but rather build up in liver and kidneys because they bind to proteins, because of their polarity. The lack of lipophilicity also explains that generally, and in contrast to PCB, mammalian males and females have similar tissue levels. PFOA and PFOS have been found in SLE beluga and show similar temporally increasing trends (M. Lebeuf, pers comm.). PFOS has been found in the liver of Baltic Sea seals (Kannan et al., 2002). So far porpoises from the UK show the highest concentrations among marine mammals (Law et al., 2008). PFCs have been detected even in the Arctic fauna. Their concentrations have increased exponentially over recent years in polar bears, which are the most contaminated marine mammals in the Arctic (Bossi et al., 2005; Butt et al., 2008; Sonne et al., 2008; Martin et al., 2004; Tomy et al., 2004).

Because the molecular structure of PFCs resembles that of endogenous fatty acids, PFCs disturb lipid metabolism (laboratory rodents become hypocholesterolaemic and show fat atrophy (Xie et al., 2003)). In humans, low birth weight has been recently associated with high levels of PFOA in maternal plasma, perhaps because the disturbed lipid metabolism deprives the developing foetus of lipid input, resulting in low weight at birth (Fei et al., 2007). Such disturbance in lipid metabolism may have a strong impact on Arctic mammals considering the central role played by fat in energy storage, buoyancy and heat insulation in these animals. PFCs also downregulate the expression of genes involved in inflammation and immune functions in rodents (Guruge et al., 2006). This feature might amplify PCB-induced immunosuppression in beluga and in other marine mammals: this hypothesis is supported by the epidemiological association between high PFOA tissue levels and infectious diseases in sea otters (Kannan et al., 2006).

Future Research

Cancer has been observed in SLE beluga but not in other SLE marine mammals or in marine mammals of the Baltic Sea or the Arctic. This observation indicates that SLE beluga are specifically exposed to chemical or biological mutagens or both, possibly through unique ecological features such as feeding on benthic invertebrates. Out of many other fish species from the approximate same habitat, cancer has been found in only a single fish species, lake whitefish. This is the only salmonid that feeds significantly on benthic invertebrates. Thus, two aquatic vertebrates, phylogenetically very distant, may be affected by cancer because they share unique feeding habits within their respective taxonomic groups. Because cancer is an ultimate but rare consequence of chemical mutagenesis, the epidemiological association of xenobiotics with carcinogenesis requires the examination of large numbers of animals exposed to environmental concentrations of carcinogens. To demonstrate the role of xenobiotics in carcinogenesis in SL beluga or in any other free-ranging mammal, convincing statistics would require the examination of much larger numbers of beluga and/or the follow-up of SLE beluga for many more decades. In addition, the observation of high prevalences of cancer in other populations of marine mammals similarly exposed to carcinogens would strengthen an etiologic relationship with chemical carcinogenesis. Few odontocetes species feed on benthic invertebrates: the Amazon River (*Inia geoffrensis*); Franciscana (*Phocoena blainvillei*); Susu (*P. gangetica*); and Irrawaddy (*Orcinus brevirostris*) dolphins (Ridgway et al., 1989). Because these species generally inhabit rivers that are often more contaminated than the open ocean, they might also be affected by high rates of cancers of the GI tract. Several chemical carcinogens leave a signature on the host genome by causing mutations at specific sites in genes involved in cell proliferation. The finding of the same signature in tumours of SLE beluga, fish and people would strongly support the aetiological role of contaminants, especially PAHs, in carcinogenesis,.

New experimental approaches for demonstrating the chronic effects of OC on cetaceans are emerging. For instance, the immune systems of several cetacean species have been reconstituted in SCID mice. These 'cetaceanised' mice have been used successfully to demonstrate the validity of a morbillivirus vaccine (De Guise and Levin, 2004). OC immunotoxicity can now be tested on these mice, a task not feasible *in vivo*.

There are complex interactions between the numerous pollutants and the associated metabolites which contaminate SLE beluga. Interactions between contaminants may be positive – for instance in rats, p,p'-DDE seems to counterbalance the reduction of thymus weight by tributyltin – or the interactions may be negative – for instance, PCBs induce CYP, which increases the formation of carcinogenic PAHs metabolites. PCB congeners have no or little effects on lymphocyte proliferation *in vitro* when tested individually, whereas they significantly reduce this immune function when mixed together (De Guise et al., 1998). Characterising these interactions will remain a major challenge. Biopsies (samples taken from live animals) will occupy an increasingly important role in monitoring contamination. Already, CYP induction has been visualised by immunohistochemistry in skin biopsies taken from free-ranging cetaceans, and has been correlated with OC levels measured from the same biopsies (Branchi et al., 2002).

Fisheries, Aquaculture and Exotic Species of the Laurentian Great Lakes

18

David J. Jude
University of Michigan, Ann Arbor, MI, USA

The Great Lakes range in size from 19,000 (Lake Erie) to 82,100 km^2 (Lake Superior) and exhibit maximum depths from 64 to 407 m, respectively (Jude and Leach, 1999). Early in their history the Great Lakes were mostly oligotrophic. Lake Erie had an eastern basin that was oligotrophic, but its western and central basins were mesotrophic. Lake Superior, because of its size, depth and most northerly location, and Lake Erie, because it is the shallowest and most productive, have characteristics that are unlike those of the other three Great Lakes. The lakes most affected by humans are those closest to the St. Lawrence River (Erie, Michigan, Ontario), which provided the easiest access to European settlers in the early 1800s. Historically the main fish species exploited were lake trout (*Salvelinus namaycush*), lake whitefish (*Coregonus clupeaformis*) and a coregonine complex (*Coregonus* spp.) in the four more oligotrophic lakes (Michigan, Huron, Ontario, Superior), while blue pike (*Sander vitreus glaucus*)(now extinct), walleye (*Sander vitreus*) and yellow perch (*Perca flavescens*) predominated in Lake Erie. The commercial production of fish from Lake Erie is approximately equal to that of the other four Great Lakes combined. The total number of native fish species in the Great Lakes ranges from 67 in Lake Superior to 112-114 in Lakes Michigan, Huron, and Erie. The Great Lakes are only some 10,000 years old and have undergone four major phases in their existence: pristine times, a period of extensive exploitation and pollution, resurrection and now a period of oligotrophication.

Fish Community Commonalities Among Lakes

The original species assemblage was similar in Lakes Superior, Michigan, Huron and Ontario (salmonine complex), but markedly different in Lake Erie (percid complex). The most abundant fish species in Lake Erie were lake whitefish, lake herring (*Coregonus artedii*), lake sturgeon (*Acipenser fulvescens*), blue pike, walleye, sauger (*Sander canadensis*), yellow perch, freshwater drum (*Aplodinotus grunniens*) and channel catfish (*Ictalurus punctatus*) (Leach and Nepszy, 1976). The other lakes contained lake trout, lake whitefish, lake sturgeon, lake herring, several species of deepwater ciscoes, burbot (*Lota lota*), deepwater sculpin (*Myoxocephalus thompsoni*), slimy sculpin (*Cottus cognatus*), emerald shiner (*Notropis athernoides*) and (in Lake Ontario only) Atlantic salmon (*Salmo salar*); Green Bay in Lake Michigan and Saginaw Bay in Lake Huron also contained large populations of walleye and yellow perch (see review papers in Loftus and Regier, 1972). In Lakes Superior, Michigan and Huron, top predators were lake trout and burbot, of which lake trout was the more abundant; prey were mainly coregonines, comprising perhaps 11 species in Lake Michigan and somewhat fewer in Lakes Huron and Superior (Koelz, 1929), and sculpins. Lake trout became the dominant

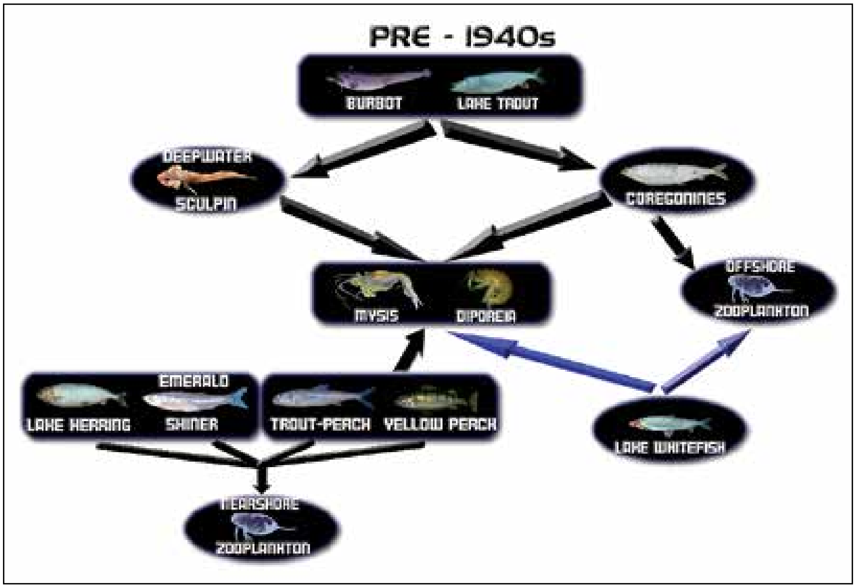

Figure 18.1. Lake Michigan food web during pristine times (pre-1940s) showing the two top predators, burbot and lake trout, their major prey items, deepwater sculpin and coregonines, and a nearshore complex of lake herring, emerald shiners, and yellow perch. Major invertebrate prey include *Mysis* and *Diporeia*. Source: Jason Jude.

predator in Lake Ontario after 1850 when Atlantic salmon disappeared from that lake (Christie, 1974). Top predators in Lake Erie were walleye and blue pike, although lake trout may have been the dominant predator of the limnetic zone of eastern Lake Erie, at least until about 1890 (Regier and Hartman, 1973); primary prey were probably yellow perch and freshwater drum.

After settlers arrived, nearly all but the smallest native species became heavily fished. Perhaps the most valuable as human food have been lake trout, lake whitefish, walleye and blue pike. Most native fish populations in the Great Lakes suffered dramatic declines between the mid-1800s and mid-1900s (see Loftus and Regier, 1972). For example, lake sturgeon had become scarce in all the lakes by 1900, while by the 1950s, lake trout were extinct in Lakes Michigan, Ontario and Erie, nearly extinct in Lake Huron, and at low levels of abundance in Lake Superior. Lake herring collapsed in Lake Erie in the 1920s, followed by declines in lake whitefish, blue pike and walleye. Declines in key predators, such as lake trout, resulted in an explosion of prey species and a constantly changing assemblage of fishes. One result of these changes was the emergence of a large biomass of rainbow smelt (*Osmerus mordax*) and alewives (*Alosa pseudoharengus*), which provided the opportunity to stock salmonines. Stocking has had a profound effect on the ecology of the lake. Large-scale stocking of salmonines began in 1965, and has continued to the present time. The rationale was to: (1) control abundance of alewives, (2) initiate recreational fishing, and (3) rehabilitate the lake trout population (Tody and Tanner, 1966; Holey et al., 1995; Rutherford, 1997). The five species of salmonines currently stocked in Lake Michigan include lake trout, Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus kisutch*), rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*).

Consequently, fish assemblages now largely consist of pelagic species (Figure 18.1), a condition that probably did not exist 200 years ago when nearshore littoral communities were in dynamic equilibrium with pelagic, offshore communities (Regier, 1979; Figure 18.2). Small, fecund pelagic species (alewife, rainbow smelt) now predominate over the larger species that were originally more closely associated with the benthic and coastal habitats of the lakes. Pacific salmon (Chinook, coho, pink and sockeye) have been introduced into all the lakes (Mills et al., 1993a). Self-sustaining populations of pink salmon (*Oncorhyncus gorbusa*) exist in all the Great Lakes and residual stocks of reproducing sockeye salmon persist in Lake Huron. White perch (*Morone americana*) have re-

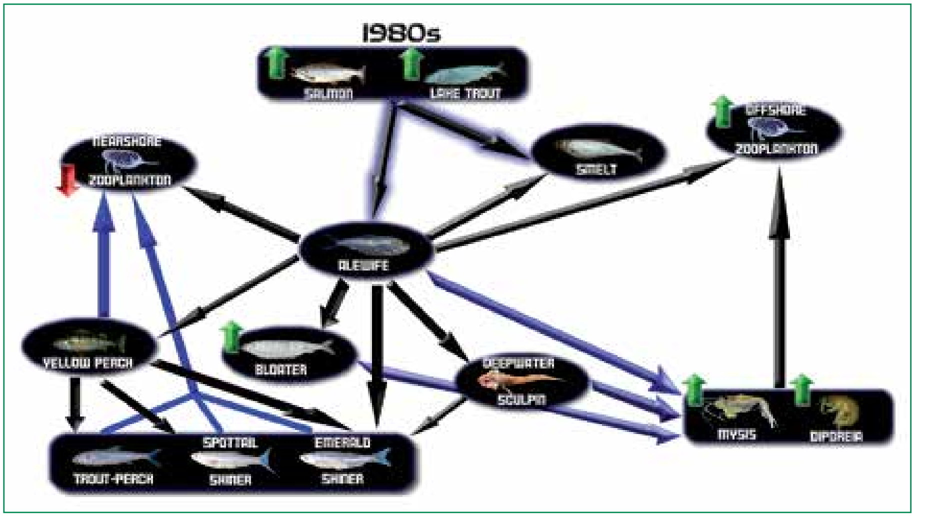

Figure 18.2. Lake Michigan food web for the 1980s-1990s showing the dominance of the non-indigenous alewife, predation effects by salmonines (salmon and lake trout), which reduced alewife abundance, and the rebound (upward arrows) of many native species after being released from predation on their larvae by alewives. Source: Jason Jude.

cently invaded and prospered in lakes Erie and Huron. In spite of major changes in species composition of the Lake Erie fish assemblages, fish yield (total biomass) has remained approximately the same as it was prior to the major cultural stresses (Regier and Hartman, 1973). Most of the Great Lakes fish communities are shadows of their formerly diverse communities of 200 years ago.

Because of human exploitation, exotic species, habitat destruction, toxic substances and eutrophication, the fish species complexes in the Great Lakes have changed dramatically over their history. There has been the addition of several exotic species and the extinction of many native species, while efforts to stock salmonines have also contributed to major changes in the fishery complex in each lake. There are common species and fish community trends among the lakes, with Lake Superior being the least affected because there were fewer people, while Lake Erie and Ontario were probably those most heavily affected by anthropogenic changes.

Lake Superior Fish Community

Lake Superior, because it is cold and deep, was once dominated by lake trout, burbot, lake whitefish, lake sturgeon, lake herring, sculpins, suckers (*Catostomus* sp) and several species of deepwater ciscoes. Overfishing, exotic species (sea lamprey especially), and some pollution of nearshore tributaries and bays were the main stressors for fishes. Rainbow smelt were common in the 1930s, while the sea lamprey entered by 1950, making Lake Superior the last lake to be colonized (Lawrie and Rahrer, 1972; Lawrie, 1978). Lake trout, one of two top predators with burbot, were being overfished and additional mortality imposed by sea lamprey caused them to decline sharply in the 1950s. There are several deepwater cisco species in the lake and they collapsed early and remained in low abundance throughout the late 1980s. Lake whitefish were overfished, some of their spawning habitat was degraded and they were also affected by sea lamprey. Lake herring were also overfished and stressed by the exotic rainbow smelt and began to decline precipitously between the 1930s and 1960s (Selgeby, 1982; Anderson and Smith, 1971).

In the late 1990s, rainbow smelt have remained at low levels, and lake whitefish and lake herring have increased (Selgeby, 1985; MacCallum and Selgeby, 1987; Hansen, 1994). No other lake herring stocks in the Great Lakes have ever recovered after reaching such low levels and there were no obvious reasons advanced (MacCallum and Selgeby, 1987). In an effort to restore top predators and promote lake trout rehabilitation, splake (lake trout-brook trout cross), rainbow trout, brown trout, brook trout, and Chinook, sockeye and coho salmon were planted voluntarily, while pink salmon were inadvertently introduced in 1956 (Emery, 1985). All contribute to sport fisheries in nearshore areas. Strategies for rehabilitation of Lake Superior included control of overfishing through quotas, banning of gill nets by Minnesota, sanctuaries and sport-fishing-only zones (MacCallum and Selgeby, 1987). Tribal groups continue to fish commercially, primarily with gill nets, throughout the lake. In addition, TFM, a lampricide, was introduced to kill sea lamprey ammocetes, and lake trout were stocked, which improved natural reproduction (mainly from native fish) (Krueger et al., 1986). By the early 2000s, lake trout in Lake Superior were declared restored and no more stocking of salmonines was allowed. However, walleye, lake sturgeon and brook trout stocks are still depressed (Williams et al., 1989; Hansen, 1994).

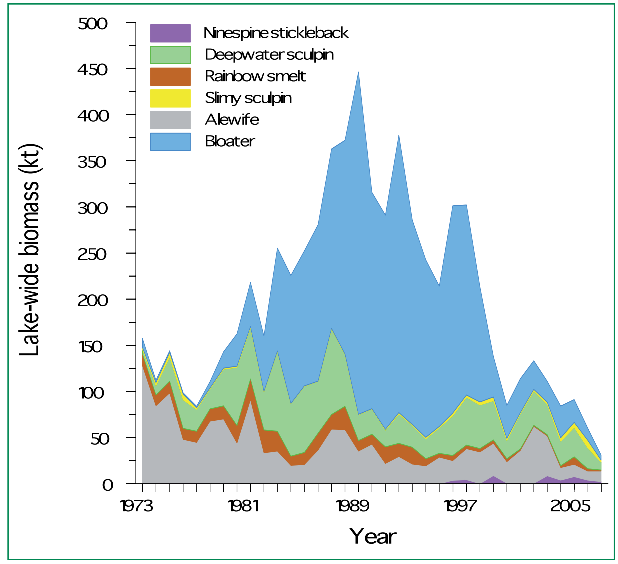

Figure 18.3. Lake Michigan forage fish catches (USGS, Madenjian). Prey fish community biomass (kilotonnes) in the main basin waters of Lake Michigan, 1973-2007. Data from standardized trawl hauls provided by Charles Madenjian, USGS Great Lakes Science Center, Ann Arbor, MI. Note the low biomass and reduced biomass of alewives during 2007.

Lake Michigan Fish Community

Native fish stocks in Lake Michigan have undergone massive changes in abundance and diversity (Wells and McLain, 1973), with overfishing important through the 1940s, and then the exotic species the sea lamprey (*Petromyzon marinus*) and alewife colonized and caused wild fluctuations in populations (Eck and Wells, 1987; Coble et al., 1990). As with all the Great Lakes, overfishing sequentially reduced the abundance of lake trout, lake whitefish, lake sturgeon and the largest species of a seven-species complex of coregonines, the deep water ciscoes (*Coregonus* spp). Even though the lake supported a robust lake trout fishery, overfishing and sea lamprey predation eventually caused the extinction of lake trout and reduced the abundance of lake whitefish. With no predators and abundant zooplankton food supplies, the alewife soon entered and flourished, reaching a peak in the 1960s. The alewife ate the larvae of native pelagic fish species, thus reducing their abundance (Eck and Wells, 1987; Krueger et al., 1995b), including lake herring, which were extirpated in the late 1950s, and walleyes in the Muskegon River around 1970 (Schneider and Leach, 1977). More recently, cultural eutrophication and contaminants have also degraded the fish community, especially in southern Green Bay. Many changes have occurred in native stocks. Six of seven species of deepwater ciscoes, lake herring and lake trout have been declared extinct (Smith, 1964). Burbot are now common due to sea lamprey control. Spoonhead sculpins (*Cottus ricei*), once thought to be extinct, have been found again. The rise in rainbow smelt and alewife in the early 1960s prompted the Michigan Department of Natural Resources to stock top predators in the mid-1960s, including lake trout, but there is little evidence of natural reproduction (Jude et al., 1981). Planting of coho and Chinook salmon (and brown and rainbow trout later) began in 1966-67, which resulted in a spectacular sport fishery (Rakoczy and Rogers, 1987). These predators ate large quantities of alewives and rainbow smelt and have contributed to the alewife decline (Jude et al., 1987; Figure 18.2). The alewife populations have continued to decline since their peak in 1966, resulting in the recovery of many native species, such as deepwater sculpin, bloaters (*Coregonus hoyi*) and yellow perch (Figure 18.3). However, overall forage fish biomass declined precipitously in 2007, probably due to reduced phosphorus levels and transfer of energy to the ecological dead end zebra (*Dreisena polymorpha*) and quagga mussels (*Dreisena bugensis*). Bloaters have declined from peaks in the mid-1980s, and round gobies (*Neogobius melanostomus*), an exotic species, have begun to increase in abundance since 2008.

Lake whitefish stocks have improved to pre-lamprey years (Eck and Wells, 1987). Yellow perch are the most important (numerically) harvested sport fish in Lake Michigan. Yellow perch populations remained low through the 1970s, but with the decline in alewives in the 1980s, they made a substantial rebound (Jude and Tesar, 1985; Shroyer and McComish, 1998). Around the late 1980s-1997, there was recruitment failure attributed to high early-life stage mortality (Francis et al., 1996). Coincidently, zebra mussels first appeared around this time, suggesting that impacts on zooplankton, first food for larval yellow perch, might be part of the problem, along with overfishing of adults. *Diporeia*, important prey of YOY perch, also declined during this period. In 1998 and 2005 (warm years), moderate recruitment of juvenile yellow perch was observed lakewide, resulting in a modest recovery of the stocks. In the early 2000s, additional dramatic change has come to Lake Michigan, as the overall biomass of forage fishes had declined (Figure 18.3), quagga mussels have expanded throughout the lake, alewives have remained at low levels, *Diporeia* remains scarce, and chlorophyll-*a* levels have declined to less than 1 as the oligotrophication process accelerates in this lake (Fahnenstiel et al., 2010).

Lake Huron Fish Community

Lake trout have also gone extinct in Lake Huron, except for a remnant stock in Georgian Bay, and scattered natural reproduction based on USGS trawl data (Eshenroder et al., 1995). Sea lamprey remain a threat to Lakes Michigan and Huron from ammocetes residing in the St. Marys River (Eshenroder et al., 1987, 1995; Ebener, 1995). However, lake whitefish populations have flourished in the 1980s. Lake herring was an abundant component of the fish community, but collapsed in the 1940-50s due to deterioration of spawning grounds in Saginaw Bay and overfishing. There is a viable population in the St. Marys River and Georgian Bay that could seed Lake Huron, now that alewives have collapsed in the lake (Ebener, 1995; Figure 18.4). Overall forage fish biomass has decreased along with alewife, round gobies have been increasing since 2005 and emerald shiners have expanded their populations for the first time since alewives entered the lake (Schaeffer et al., 2008). Walleyes have shown increased recruitment in Saginaw Bay (Fielder et al., 2007).

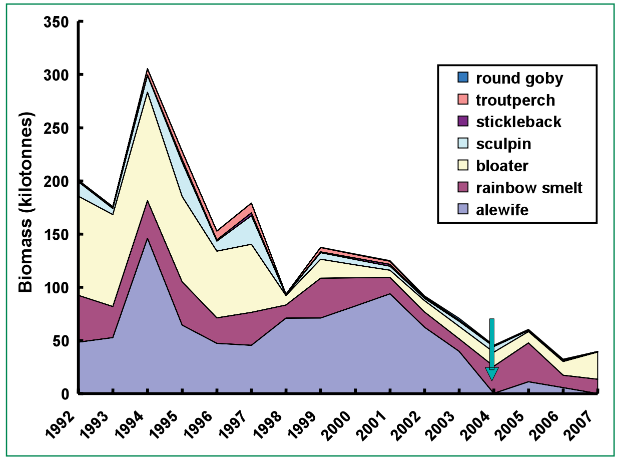

Figure 18.4. Prey fish community biomass (kilotonnes) in main basin waters of Lake Huron, 1992-2007 (USGS fall bottom trawl survey). No sampling occurred during 2000; biomass estimates for that year represent interpolated values. Data based on standardized trawling provided by Edward Roseman, USGS Great Lakes Science Center, Ann Arbor, MI.

The deepwater ciscos flourished until about 1940 and then declined, probably due to overfishing combined with alewife predation on larvae and competition for zooplankton prey (Brown et al., 1987). Now, only the bloater persists, as the five larger ciscoes were exterminated. Yellow perch are an important sport and commercial species; they are harvested from Saginaw Bay and southern Lake Huron. Abundance declined in the 1970s due to overfishing (Eshenroder, 1977), then production increased, but growth has been slow. The lack of *Hexagenia* in the bay (Hayward and Margraf, 1987; Ebener, 1995) and invasion of dreissenids around 1990 are undoubtedly inhibiting perch abundance. The other important percid in Saginaw Bay is the walleye, the second largest population in the Great Lakes (Schneider and Leach, 1977). The population collapsed in the 1940s due to Saginaw River pollution, which degraded spawning habitat in the river and bay, while overfishing contributed as well. Walleye commercial fishing was banned in Saginaw Bay in 1969, water quality improved, and in the 1980s, a large year class was produced based on stocking. Large numbers of walleye larvae are produced in the Saginaw River, but few survive to contribute to the fishery (Jude, 1992). However, with the alewife collapse in Lake Huron in the early 2000s, there is high survival of naturally produced walleye larvae (Fielder et al., 2007), indicating the damaging effect of alewife predation on pelagic larvae.

Lake Erie Fish Community

Lake Erie is the shallowest and most productive of the Great Lakes, undergoing dramatic fish population changes. The lake has three distinct basins (western, central, and eastern), ranging from oligotrophic in the east basin to eutrophic in the western basin. The central mesotrophic basin is adapted for percids (Leach et al., 1977). Exploitation of fish, eutrophication, exotic species, habitat destruction, siltation and inputs of toxic substances have led to many changes in fish populations, a dead zone with varying degrees of anoxia, loss of *Hexagenia*, blue pike, and lake herring. Many of these changes first occurred in Lake Erie because of its human population and shallow nature. Lake trout, lake herring, blue pike and longjaw cisco (*Coregonus alpenae*) all went extinct in Lake Erie. Lake herring collapsed in 1925, probably due to overfishing (Regier et al., 1969). Lake trout were overfished and their spawning habitat degraded, causing them to be extirpated by the 1930s (Hartman, 1972). Lake whitefish collapsed in the 1950s, but staged a dramatic comeback in the 1990s (Ebener, 1997). Stocking efforts have failed to restore lake trout. Other common species in Lake Erie include: channel catfish, emerald shiner, white bass (*Morone chrysops*), gizzard shad (*Dorosoma cepedianum*), spottail shiner (*Notropis hudsonius*) and common carp (*Cyprinus carpio*). Lake trout (moderately abundant in the eastern basin) began to decline in the 1950s-60s (Applegate and VanMeter, 1970) followed by lake sturgeon, lake whitefish, lake herring, sauger, blue pike, walleye and yellow perch. Rainbow smelt replaced the lost productivity by 1970 and became the dominant species. Rainbow smelt have been trawled commercially since the 1950s, while the yellow perch has been the most valuable commercial species (Hartman, 1988). Alewives colonised Lake Erie in 1931 (Smith 1970a), but because of the lack of a cold water refugium and high predation pressure, they have not done well (Hartman, 1972). White perch, an east coast estuarine species, was first found in 1953, then exploded in the late 1970s (Boileau, 1985). It has expanded its range to Saginaw and Green Bay, where it suffers high overwinter mortality. Extinction of the blue pike, which was ideally adapted to the central basin and sustained some of the highest catches in the Great Lakes, occurred around 1958. Overfishing, degraded summer habitat and introgression combined to eliminate this species (Regier et al., 1969; Leach and Nepszy, 1976). Saugers suffered the same fate. Walleyes remain a very important sport fish in Lake Erie. The species declined in the 1950s and 60s due to overfishing, nutrient overload effects, loss of *Hexagenia*, and exotic species, such as the rainbow smelt (Schneider and Leach, 1977; Regier et al., 1969). Rehabilitation efforts for walleye have been successful, as limits to harvest and reduced phosphorus loading have increased population levels (Knight, 1997).

Lake Erie in the 2000s continues to recover. Zebra and quagga mussels invaded along with round gobies, which now dominate the lake. Chlorophyll levels have been reduced as a result of lower phosphorus loading and filtering of dreissenids, anoxia has been reduced allowing *Hexagenia* to rebound, and water clarity is at all-time high levels, resulting in high abundances of aquatic plants and associated fish fauna in embayments (Makarewicz and Bertram, 1993). Walleye, smallmouth bass (*Micropterus dolomieu*), lake whitefish, burbot and lake sturgeon have surged in abundance, helping to stabilise the ecosystem.

Lake Ontario Fish Community

Lake Ontario, the only Great Lake directly connected to the Atlantic ocean, has a unique fish fauna (Atlantic salmon and American eels (*Anguilla rostrata*)), and it has had high levels of nutrient loading (Beeton, 1965; Ryder and Edwards, 1985). Because Lake Ontario was colonised by humans first, major changes were recorded in this lake prior to any of the others. Lake Ontario probably has had the most fish extinctions: lake herring, lake trout, blue pike, Atlantic salmon around 1900, deepwater sculpin (Brandt, 1986), and four coregonines in the 1940s-50s. Most extinctions were caused by overfishing, including reductions in lake whitefish and lake sturgeon (Christie, 1972). Only lake whitefish populations have rebounded (Christie et al., 1987b; Casselman et al., 1996), while some deepwater sculpins have been found recently. The alewife is common to abundant in the lake and that, plus the lack of terminal predators, prompted stocking of salmonines in the 1970s (Kerr and LeTendre, 1991). Introduction of the zebra mussel caused increased water clarity and movement of alewives deeper, but the fish community remains almost extinct to depauperate in the deep abyss, while the remaining community remains in flux (Jones et al., 1993). Lake trout are currently being stocked but little reproduction has been recorded (Krueger et al., 1995). Attempts are also being made to introduce Atlantic salmon, but dams and tributary degradation are hurdles. In Lake Ontario alewife are being consumed by salmonines, but there have not been substantial declines as observed in Lake Huron until the 1990s (Jones et al., 1993). It is not clear what species may respond if alewives decline more, but O'Gorman et al. (1987) state that yellow perch, white perch and remnant stocks of lake herring are candidates. Such a decline would also suggest that the endemic fauna, especially bloaters and deepwater sculpin, should also be re-introduced. In response to changes in prey and lower phosphorus concentrations, Chinook salmon stocking was reduced by 50% (Jones et al., 1993). Declines in the salmonine fishery in Lake Ontario and Lake Huron and elimination of stocking in Lake Superior are new directions of management toward more native species. Lake Ontario has been partially resurrected in the late 1990s and early 2000s. Dreissenids and lower phosphorus concentrations have combined to increase water clarity, deepwater sculpin, formerly thought to be extinct, have been collected, there has been a reduction in contaminants and sea lamprey predation, lake whitefish are at historical high levels (Casselman et al., 1996), walleyes have staged a resurgence (Bowlby et al., 1991), there has been some natural reproduction by lake trout in conjunction with the alewife decline, thereby reducing their predatory effect on lake trout, and yellow perch have increased (Jones et al., 1993).

Aquaculture

The major aquaculture activity in the Great Lakes basin involves production of: 1) Salmonines by the federal government for stocking into the Great Lakes; 2) Walleyes for stocking into Saginaw Bay and other water bodies by the Department of Natural Resources from the various states; 3) Cyprininds by the private sector for the bait industry; 4) Yellow perch, rainbow trout and *Talapia* by the private sector for consumption; 5) ornamental fish for small fountains, aquaria and ponds; and 6) rainbow trout and other species stocked into ponds for fee-for-fishing operations. There are some 25 different species currently cultured or cultured in the past in the Great Lakes region. In addition, there is some pen culture of salmon ongoing, as well as some rainbow trout culture activities. All of these projects are small, with the exception of salmonines raised by the federal government for stocking into the Great Lakes. The data on numbers of fishes stocked by the federal government and the Great Lakes states are available in a Great Lakes sponsored database termed the Great Lakes Fish Stocking Database, which is now administered by the USFWS in Franken, WI, and can be accessed on the internet through the Great Lakes Fishery Commission website. In 2005, the eight states and Ontario (not all in the Great Lakes basin) had 183 aquaculture farms with viable operations worth 13 million dollars. However, this is barely 4% of the US total aquaculture value nation-wide and only composed 2% of Canada's production (USDA NASS, 2005). Both the Great Lakes Fishery Commission and many of the state's natural resource agencies are against cage culture of fishes in the Great Lakes.

Exotic Aquatic Species

There were 34 species of fishes introduced into the Great Lakes basin between 1819 and 1974 (Emery, 1985). A more recent summary was provided by Mills et al. (1993a). Species with major impacts entered the Great Lakes through canals built for shipping and include sea lamprey, alewife, white perch, round and tubenose gobies (*Proterorhinus semilunaris*) and ruffe (*Gymnocephalus cernuus*).

The lower food web has been damaged by exotic zooplankton and dreissenid introductions, since algae and native zooplankton have been diverted into invasive species that may be less palatable or an ecological dead end to higher trophic levels. Contaminant pathways are being modified and new trophic levels introduced (Ng et al., 2008). Zooplankton are important food for planktivores and obligatory prey for larval fish and are likely to be strongly linked to fish recruitment. *Bythotrephes longimanus* and *Cercopagis pengoi* are recently introduced zooplankers that feed on other zooplankton (Evans, 1988) and in turn are fed on by fish (Warren and Lehman, 1988). *Cercopagis* and *Bythotrephes* are expected to affect Great Lakes food webs through competition with larval fish and fish planktivores for small zooplankton. However, the most influential recent intruders are the zebra and quagga mussels, which have spread through all the Great Lakes and far outside the basin (New York Sea Grant, 1993). Zebra mussels produce faeces and pseudofaeces, which provide nutrients for algae, macrophytes, benthos and the microbial food web. This process is the major mechanism whereby mussels shift energy from the pelagic to the benthic zone (Figure 18.5). Zebra mussels selectively feed on diatoms and green algae, promoting blue-green algae, including toxic *Microcystis* blooms during summer (Vanderploeg et al., 2001). They also directly remove micro zooplankton, such as protozoa and rotifers (Lavrentyev et al., 1995; MacIsaac et al., 1995), and have an important role in nutrient cycling (Arnott and Vanni, 1996; Johengen et al., 1995). Zebra mussels modify and create new habitats for microbes, invertebrates and fishes (Ricciardi et al., 1997; Jones et al., 1997), and are linked to killing native mussels (Schloesser and Nalepa, 1994).

Quagga mussels tolerate colder water temperatures, can colonise soft substrates and occur in deeper waters than zebra mussels (Mills et al., 1993b), and as a result have dominated former habitats occupied by zebra mussels and extended the range of dreissenids from nearshore water (Dermott et al., 1998a; Mills et al., 1999; Berkman et al., 1998) to the deep abyss of the lakes. They have high fecundity and produce planktonic, veliger larvae that can be transported long distances (Garton and Haag, 1992). These species foul water intakes (Stanczykowska, 1978), and have had detrimental impacts on the two major groups that feed on algae: zooplankton and *Diporeia* (Nalepa et al., 1998; Figure 18.6). As a result there has been a trophic cascade in lakes Huron and Michigan resulting in declining prey fish and the salmon fishery in Lake Huron (Fahnenstiel et al., 2010). They have also fouled rocky spawning reefs and made habitat there more desirable for another exotic species, the round goby which eats mussels. As evidence of their ability to shift energy from the pelagic to the benthic zone, Leach (1993) demonstrated that chlorophyll-*a* declined by 54% in the western basin between 1988 and 1990. In Saginaw Bay, chlorophyll-*a* declined by 59%, total phosphorus by 42% and phytoplankton production by 38%, while water transparency increased by 60% (Nalepa and Fahnenstiel, 1995) and zooplankton declined (Bridgeman et al., 1995). Macrophytes expanded (Skubinna et al., 1995) and there was increased production of benthic algae (Lowe and Pillsbury, 1995). In Lake Erie, walleyes moved farther offshore to avoid light (Ryder, 1977).

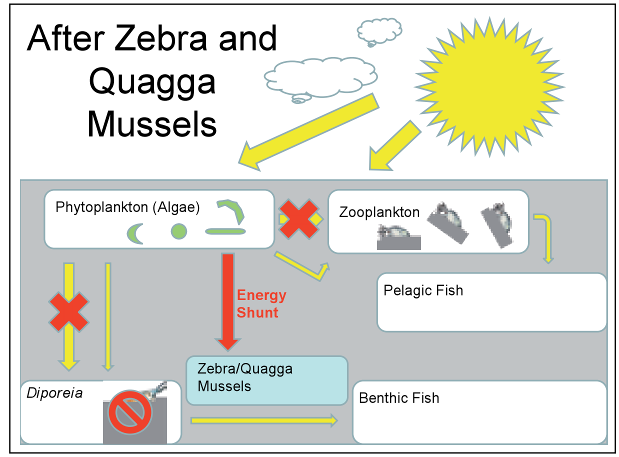

Figure 18.5. Diagram showing Great Lakes food web prior to zebra mussels when energy flowed through algae to *Diporeia* and zooplankton and then into pelagic and benthic fishes. Zebra and quagga mussels short circuit this food web by filtering the algae and shunting energy to ecological dead end mussels. A trophic cascade has resulted, wherein declines in zooplankton and *Diporeia* have led to declines in forage fishes, and in Lake Huron, a collapsing salmon fishery.

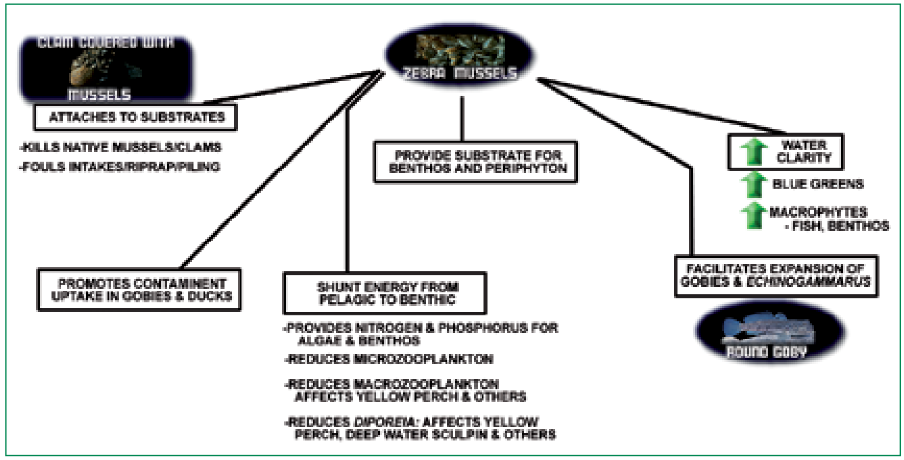

Figure 18.6. Diagram of the various impacts that zebra and quagga mussels have on the physical, chemical, and biological characteristics of the Great Lakes ecosystem. Invasive mussels foul intakes, shunt energy to the benthic zone, facilitate expansion of other exotic species, change physical habitat, increase water clarity, and kill native mussels and severely depress other benthic organisms, such as zooplankton and *Diporeia*. N=nitrogen, P=phosphorus, YP=yellow perch. Source: Jason Jude.

Non-indigenous Fishes

Non-indigenous species have been introduced at every level of the food chain, and have extirpated some native species, disrupted fish communities and reduced the abundance of others. Invaders have exploited trophic opportunities and colonised new habitats because of their species-specific attributes, environmental conditions that were similar to their native habitat, anthropogenic perturbations that allowed access to food and habitat, or modifications of the environment (invasional meltdown) by other invaders (Ricciardi, 2001). They have shifted energy from the pelagic to the benthic zone and large benthic organisms have been replaced with r-strategist species. Benthos and fishes associated with aquatic plants have flourished, while contaminant pathways have been modified, new trophic levels added and routes for botulism uptake established. An estimated 4,500 non-indigenous species have been introduced into the United States, with 162 introductions into the Great Lakes (Mills et al., 1994; Ricciardi, 2001). Introduced species are arguably the most serious threat to the ecological health of most of the Great Lakes today and Great Lakes global warming will exacerbate present effects by decreasing coldwater forms, allowing southern species to move northward (Mandrak, 1989), reducing annual primary productivity, and promoting the spread of exotic species, including diseases and parasites (Meisner et al., 1987).

Sea lampreys were in Lake Ontario as early as 1819 (Emery, 1985) and gained access to the rest of the Great Lakes in the 1920s-40s. They prey on many species, but overfishing, which reduced the mean size of the lake trout, caused them to become extinct in all the lakes except Superior and some remnant stocks in Georgian Bay, Lake Huron (Lawrie, 1970). Sea lampreys can kill up to 18 kg of prey in their life (Jude and Leach, 1999). Sea lampreys are still a serious threat, despite good control of their ammocetes in tributary spawning streams using lampricides. There still remains a large population in the St. Marys River, which is too large to be economically treatable.

With the top predators lake trout and burbot extinct or severely reduced and planktivores and other competitors also diminished to extremely low levels, marine alewives entered and quickly proliferated throughout the Great Lakes, reaching peak numbers in Lake Michigan during 1966-67 (Brown, 1972). Alewives did well in only those lakes with adequate cold refuges, but did not colonise Lakes Erie and Superior to any large degree (Smith, 1972a; Christie, 1974; Bronte et al., 1991). Alewives are predators on the larvae of native pelagic fish species, reduce zooplankton size spectra (Wells, 1970), and are prey for stocked salmonines (Jude et al., 1987; Madenjian et al., 2002). In Lake Michigan, salmonines reduced the abundance of alewives, which reduced the impact on large cladocerans. The resulting trophic cascade (Scavia et al., 1986) produced increases in water clarity due to the more efficient filtering of algae from the water by *Daphnia*. The choice of maintaining or eliminating alewives presents a conundrum for management agencies, which must decide between having large alewife populations to support a salmonine fishery worth billions of dollars, or reducing alewife populations to allow increased survival of native species such as yellow perch. In the 2000s, alewives have been reduced to extremely low levels in Lake Huron and are at a long-term low, but are still successfully reproducing in Lakes Michigan and Ontario. The near-elimination of alewives in Lake Huron has resulted in dramatic rebound of walleye in Saginaw Bay (Fielder et al., 2007) and of emerald shiners in the main lake (Schaeffer et al., 2008), as their larvae were released from predation. Fewer alewives resulted in improved recruitment of native species (e.g. yellow perch, deepwater sculpin, bloaters) and increased the abundance of the larger cladocerans (Figure 18.7). Since *Daphnia* are more efficient at filtering algae than smaller zooplankton, water clarity has improved.

White perch was first found in the Great Lakes in Lake Ontario in 1950 (Mills et al., 1993). They have colonised all the Great Lakes, but became extremely abundant in Lake Erie and are suffering die-offs in Saginaw Bay each winter. They are also negatively affecting yellow perch in Green Bay.

The round and tubenose gobies were discovered in 1990 in the St. Clair River (Jude et al., 1992) and rapidly moved throughout all the Great Lakes, with their dispersal mediated by uptake of larval forms in the ballast water of freighters (Hensler and Jude, 2007). Round gobies are robust, since they have adaptations similar to cavefishes, feed in the dark (Jude et al., 1995), and survive low dissolved oxygen conditions. In addition, they can respire through the skin (Moyle and Cech, 1988). Round goby females can spawn every 20 days up to six times during the year (Jude, 2001), guard their eggs and have high survival. Large individuals mostly eat zebra mussels (Ghedotti et al., 1995), but also fish (French and Jude, 2001), and because they are so abundant in many areas, are prey of most of the top predators in the system (Jude, 2001). They have caused extirpation of mottled sculpin (*Cottus bairdi*) in areas of overlap (Janssen and Jude, 2001; Lauer et al., 2003), have diet overlap with native benthic species in the St. Clair River (French and Jude, 2001), and have been linked to the decline of darters and logperch

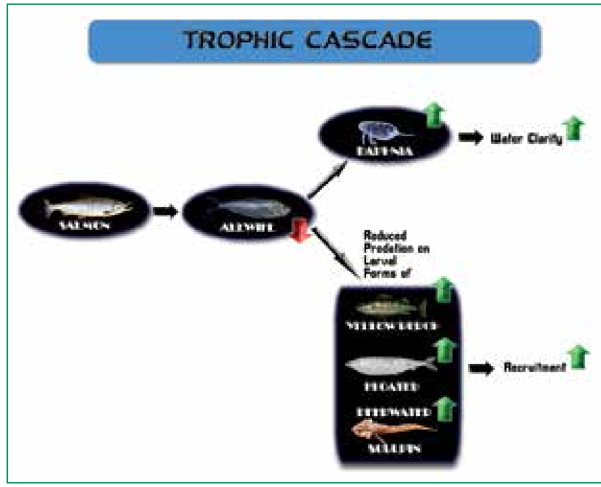

Figure 18.7. Trophic cascade involving reduction of alewives by stocked salmonines. Fewer alewives resulted in improved recruitment of native species (e.g., yellow perch, deepwater sculpin, bloaters) and increased abundance of larger cladocerans. Since *Daphnia* are more efficient at filtering algae than smaller zooplankton, water clarity improved. Source: Jason Jude.

in rocky reefs in Lake Erie (Jude and Thoma, personal communication). They have also reduced survival of larvae from smallmouth bass nests (Steinhart et al., 2003) and reduced benthic populations in rivers (Carman et al., 2006) and Lake Michigan (Kuhns and Berg, 1999). They may also change toxic substance pathways, causing increased bioaccumulation through food chains from zebra mussels to smallmouth bass (Morrison et al., 1999; Ng et al., 2008) and by ducks eating zebra mussels (Petrie and Knapton, 1999). In lakes Erie and Michigan, round gobies are also implicated in the deaths of many organisms, such as gulls, loons (*Gavia immer*), and lake sturgeon that ate round gobies contaminated with botulism-bearing zebra mussels (Domske and Obert, 2001). They consume lake trout and lake sturgeon eggs and so are a threat to restoration of these species in Great Lakes nearshore reefs. Once nearshore, round gobies are now expanding their range to offshore waters in Lakes Ontario, Huron and Michigan (Walsh et al., 2007; Schaeffer et al., 2005; Jude, unpublished data). Burbot are now consuming high percentages of round gobies on offshore reefs with the potential of changing trophic position and contaminant uptake pathways (Henser and Jude, 2008). Lake whitefish, in response to the *Diporeia* declines, have shifted their diets to round gobies and quagga mussels (Jude, pers. observations).

The Eurasian ruffe (*Gymnocephalus cernuus*), another ballast water transplant, was first found in 1987 in western Lake Superior at Duluth Harbor (Pratt et al., 1992). It rapidly attained high abundance within the Duluth Harbor (Busiahn, 1996), then spread from tributary to tributary along Lake Superior, then probably via ballast water, and was found in Lake Huron (Thunder Bay), then Green Bay, Lake Michigan, in 2002. It is expected to change fish dynamics in Saginaw Bay and Lake Erie, but has not been recorded there yet. Ruffe are small percids with spiny dorsal fins that discourage predators (Ogle et al., 1996), are very fecund (up to 20,000 eggs), are multiple spawners, and feed at night (Janssen, 1997a). Anglers find them unattractive because of their small size (Ogle, 1998). There is diet overlap and documented laboratory detrimental effects of ruffe on yellow perch (Fullerton et al., 1998). However, few changes were noted in Lake Superior native fish populations that could be attributed to impact of the ruffe (Gunderson et al., 1998; Bronte et al., 1998).

There are a number of other fish species that have threatened to enter or entered the Great Lakes. The common carp was introduced early in the 1800s (Mills et al., 1993), has colonized all the Great Lakes, especially eutrophic bays, and has had negative impacts on shallow habitats by destroying vegetation and increasing turbidity (Emery, 1985). Another cyprinid, the goldfish (*Carassius auratus*) is also common in some shallow habitats and probably originated from pets being dumped in local habitats. More threatening, however, may be several other carp species, e.g., grass carp (*Ctenopharyngodon idella*), black carp (*Mylopharygodon piceus*), bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*Hypophthalmichthys molitrix*), which escaped into the Mississippi River from aquaculture operations and are slowly migrating toward the Great Lakes. The silver carp, which has a reputation for jumping from the water and injuring people and damaging equipment, has recently been found in the Illinois River in large numbers and is displacing native fish species (USFWS, 2002). An electrical barrier was installed in the Chicago Sanitary and Ship Canal (CSSC), which connects the Illinois River with Lake Michigan, first to stop round gobies and now Asian carp (Moy, 1999). A silver carp was observed during electrofishing during August 2009 near the confluence of the Des Plaines River and the CSSC. Poisoning of the CSSC with rotenone in December 2009 showed that one bighead carp had bypassed the barrier, whereas intensive efforts by the US Fish and Wildlife Service in late 2009 and early 2010 have produced no evidence of silver or bighead carp in warm-water discharges below the barrier. Silver carp grow very large, eating zooplankton as juveniles, later switching to phytoplankton, zooplankton, algae and benthos by filter-feeding. Asian carps require warm water for growth (Nico et al., 2005), so it is likely their impact would be limited to warmer, more productive waters of the basin (Barberio et al., 2009). The Asian carps face many bottlenecks to successful survival and reproduction in the Great Lakes: 1. Water temperatures in most areas are too cold for growth, 2. Plankton populations are too sparse to support growth, 3. Successful reproduction requires large rivers with turbulence, high velocities, long reaches up to 100 km to keep eggs suspended and appropriate nursery areas (Nico et al., 2005).

Rainbow smelt (*Osmerus mordax*) is another species that was stocked into Crystal Lake in 1912 to provide forage, then escaped into Lake Michigan and then into all the Great Lakes. It is believed to suppress recruitment of native coregonines either through food competition (Anderson and Smith, 1971) or predation on fry (Selgeby et al., 1978). It is commercialy fished in Lake Erie and has attained high abundance in Lake Superior. It has become an important sport fish where it is abundant. Several exotic salmon and trout (rainbow trout), brown trout, coho salmon (*Oncorhynchus kisutch*), Chinook salmon have been stocked into the Great Lakes at various times, but they attained high abundance in the 1970s to present times, when they were stocked to establish a sport fishery. A spectacular sport fishery resulted and these species have stabilised the ecosystem, reduced alewife biomass, and more recently have presented a dilemma for fishery managers, as the alewife population on which they depend has declined to almost undetectable abundance in Lake Huron and is reduced in Lakes Michigan and Ontario.

Prevention and Research Needs

Mills et al. (1993) documented the modes of entry of non-indigenous species into the Great Lakes and identified ship ballast water as the main vector of dispersal. Once here, Great Lakes freighter ballast water dumping within the lakes and bait-bucket transfer have been responsible for their spread throughout the Great Lakes and into inland streams and lakes as well. Outreach efforts to inform the public about spreading invasive species through contaminated boats and bait buckets has been effective in some cases, but more needs to be done. We are aware of one individual who stocked zebra mussels in a Michigan inland lake because he wanted to clear up the water; he was completely unaware of the ecological consequences of his action. Attempts to control ballast water have been slow, but the US Coast Guard has required incoming freighters to exchange ballast water in the open ocean, which has proven somewhat effective. However, the flaw in these procedures is that freighters without ballast on board (NOBOBs) are allowed to enter the Great Lakes. These ships with NOBOB have residual water, sediment and organisms in their ballast tanks, which are often mixed with Great Lakes water and at some point during picking up or unloading cargo, they dump this material at a port. In addition, many organisms may be present on the outside of ships attached to the ship's hull, ropes or other structures. In order to control the spread of ruffe from Duluth, Minnesota harbours, where they were first found, freighters were asked to exchange ballast water in deep areas of Lake Superior. Ruffe still made it to lakes Huron and Michigan. Research efforts to control ballast water have involved the use of chlorine, gluteraldehyde, on-board fine-screen pumps, and suggestions to make ships deposit all ballast water into onshore sewage treatment plants. Based on their research of round goby larvae diel vertical migrations at night, Hensler and Jude (2007) suggested that ballast water uptake by freighters should only occur during the day, when larvae are almost exclusively on the bottom. Some states, such as Michigan, have passed stringent recommendations for more effective control of ballast water. The urgency to enact better control measures is based on what we know about organisms that arrived here on ocean-going ships, such as the dreissenids and round gobies. We know considerably less of lower trophic levels. Similar destructive scenarios may have occurred with bacteria, viruses and algae. Cholera has been found in the ballast water of ships and the recent outbreaks of botulism type E and viral hemorrhagic septicemia are also believed to be of ballast-water origin. Few studies of viruses and pathogens have been conducted and this information is required for identifying potential problems, planning, prevention and mustering public support for tougher regulations. Humans often have the opportunity to be in or catch fish from water that could be contaminated with ballast water, which could lead to localised catastrophes. Another scenario, which has already been and continues to occur, is zebra mussels filtering pathogens (e.g. botulism), which in turn are passed up food chains. Monitoring water and dreissenids in high-risk areas for pathogens should be a new research priority. Anthropogenic activities often cause major perturbations in ecosystems, which may facilitate the entry of non-indigenous species (Elton, 1958; Ehrlich, 1989). Most of the Great Lakes have modified or destroyed nearshore habitats and water quality has been degraded in many areas (Sonzogni et al., 1983). Global warming has already increased temperatures in the Great Lakes and is expected to cause major changes in fish species diversity and abundance (Magnuson et al., 1997). Warming will also promote expansion of aquatic nuisance species, parasites, and diseases that flourish at higher temperatures. Hence, efforts should be directed toward increased ecosystem health, balanced fish populations, and improved water quality to deter the successful entry of exotic species. We need a basic understanding of food web processes, especially those of the lower food web. The microbial food chain, viruses, pathogens, algae, zooplankton and benthos require more study in order to identify new organisms and predict exotic species effects, especially in the most disturbed, nearshore areas. Algal communities, critical food for zooplankton and *Diporeia*, have been degraded since human arrival in the 1800s, yet we know little of their population dynamics. The second major conduit for transfer of organisms to the Great Lakes from the Mississippi River is the Chicago Sanitary Canal in southern Michigan. Round gobies have entered the Mississippi River from Lake Michigan and Asian carp are threatening to enter the Great Lakes from the Illinois River. This artificial connection needs to be eliminated.

Conclusions

The Great Lakes have proceeded through four phases in their short 10,000-year history: pristine times, the apocalypse, the resurrection, and oligotrophication. Eutrophication has been controlled through improvement of sewage treatment plants and subsequent reduction in phosphorus loading. Many toxic substances have been banned (PCBs, DDT), leading to dramatic increases in sea gulls, loons, eagles and cormorants. Habitat destruction continues, but efforts are ongoing to restore wetlands and improve habitat quality and quantity. Overfishing has been curtailed by banning gill nets and shifting harvest to sport fisheries. We currently are in the age of oligotrophication, as the combined effects of reduced phosphorus loading and dreissenids have driven down the chlorophyll-*a* values in lakes Huron and Michigan to < 1, increased water clarity, and changed zooplankton species composition. Lake Huron has lower chlorophyll-*a* values than Lake Superior and these changes in the lower food web are being propagated much faster than expected, as we have also seen a dramatic decline in the forage fish biomass in Lake Huron and less so in Lake Michigan. Despite these improvements, many difficulties remain (Table 18.1). With the decline in alewife populations in lakes Ontario, Michigan, and especially Huron, there has

Table 18.1. List of existing and future management issues/problems in the Great Lakes.

Problem	Brief description
Ballast water discharge	Major source of invasive species; needs to be controlled
Zebra/quagga mussels	Continued expansion: detrimental effects on benthos
Lack of *Hexagenia* in Saginaw Bay	Need restoration to improve perch growth
Xenophobic toxic substances	Unknown chemicals with unknown impact
Depauperate abyss/loss of energy transfer	No fish in deepwater abyss of many lakes
Areas of concern	Impaired habitats need restoration/rehabilitation
Zooplankton herniations	Unknown agent negatively affecting zooplankton
Diporeia loss	Dramatic decline has negative effect on fishes
Oligotrophication	Lowered nutrients causing decreased productivity
Non-indigenous species	Continuing serious problem; ballast water control
Sea lamprey	Maintain control; problem with St. Marys River
Global warming	Reduce coldwater species, spread exotic species
Bacterial Kidney Disease	Periodic problem when salmonines are stressed
Early Mortality Syndrome	High salmonine egg mortality from eating alewife
Viral haemorrhagic	Introduced virus that kills many species of fishes
Type E botulism	Has killed many loons, gulls, sturgeon, walleyes
Lake trout rehabilitation	Still stocked in 4 Gr Lakes: established in Superior
Asian carp	Threatening to enter the Great Lakes; devastating
Round gobies	Expansion to deepwater, effects on reefs, botulism
Yellow perch decline in Lk Michigan	Lack of recruitment- overfishing/dreissenids
Salmonine stocking strategy	With alewife decline: to stock or not to stock
Wetland losses	Continuing; requires more restoration activities
Alewife decline	Continue stocking salmonines: rebound of natives
Cormorants	Population explosion: impact on fish; control
Native American gill netting	Affects non-target species
Lake Erie dead zone	Increasing areas of anoxia in recent years
Asian Tapeworm	Infects predatory fish; widespread
Hemimysis anomala	Exotic invertebrate: eats zooplankton; prey for fish

been an upsurge in recruitment in walleyes in Saginaw Bay, but managers must now decide whether to promote large alewife populations to support a salmonine fishery worth billions of dollars, or continue the trends of reducing alewife populations to allow increased survival of native species such as yellow perch, lake herring, walleye and lake trout. Early mortality syndrome continues to plague hatcheries using top predators that eat alewives, which reduces thiamine in eggs and causes mortality (Fitzsimons, 1995). A further assault on stocked salmon was an outbreak of bacterial kidney disease (BKD) within the Chinook salmon population (Kabre, 1993), which decimated their numbers in the early 1990s. Round gobies and ruffe have not yet reached peak abundance, and many ecological interactions and implications for toxic substance mobilisation and botulism facilitation remain. In addition, round gobies are now overrunning offshore reefs and penetrating coldwater trout streams and we have no information on their impact on fish communities and reproduction of salmonines in these environments. The deepwater abyss of Lake Ontario, the eastern basin of Lake Erie and the central basin of Michigan all have extinct or severely reduced populations of fishes. This has led to a loss in energy transfer from lower to higher trophic levels and efforts to restore key forage and predator species. Targets have been lake herring in Lake Huron, bloater, deepwater sculpin and Atlantic salmon in Lake Ontario, and lake trout where sanctuaries have been established to promote their abundance. For lake trout, some four decades after the first were stocked to restore populations, only Lake Superior, with a remnant stock of natives, has been fully restored. Climate change has already increased the mean temperature in the Great Lakes and prolonged the spring warming trend, enhancing yellow perch recruitment. The growth and habitat of fishes has been changed by the increase in water temperature, which is enhancing the growth of warm water species by increasing their normal habitat, and decreasing the growth of coldwater species. Non-indigenous species are expected to expand their ranges as a result. Zebra and quagga mussels have filtered the water of algae and caused massive changes to the chemical, physical and biological characteristics of the habitat they colonise. Major impacts have been documented in reductions of *Diporeia* and zooplankton, with detrimental effects on fish recruitment and reduced energy transfer to many species which depended on this lipid-rich food source. Quaggas have expanded their range throughout lakes Michigan and Huron, further exacerbating a serious problem in the Great Lakes. There is still a continuing problem with sea lampreys in the St. Marys River and they have detrimentally affected salmonines and burbot in the more northern lakes (Huron and Michigan). Viral hemorrhagic septicemia, an exotic ballast water virus, has killed thousands of important species such as muskellunge in the St. Clair River, and is also spreading throughout the Great Lakes and into inland lakes. Introduced species have at one time or another been the dominant species in all the Great Lakes. These include the sea lamprey and Pacific salmon as top predators in lakes Ontario, Michigan and Huron, and white perch, alewives, rainbow smelt and round gobies in Lake Erie. However, recent invaders continue to assault the Great Lakes (e.g., Asian tapeworm, *Hemimysis*) and some threaten (silver carp), showing that we have not established well integrated fish communities, which would resist expansion of non-indigenous species.

Many of the Great Lakes, especially their main basins, will continue to be highly prone to invasions (Jude et al., 2006). First, the Great Lakes are a species-depauperate system (similar to the eastern basin of Lake Erie and Lake Ontario) due to a short evolutionary history and isolation from invasion because of Niagara Falls, and second, because the nearshore areas tend to be disturbed. Main basin invaders will be different from those of bays and estuaries. Main basin invaders are expected to be of marine origin, but tolerant of freshwater, while bay and estuary invaders will come primarily from freshwater (Jude et al., 2006). We need more understanding of basic scientific processes in the Great Lakes, including regular monitoring of the lower food web, especially in nearshore areas. We need better long-term data sets on how food webs function, since exotic species are poised to influence food-web processes in Great Lakes ecosystems now and in the future.

Fisheries of the St Lawrence River, Estuary and Gulf

19

Daniel Martineau
University of Montreal, Saint-Hyacinthe, QC, Canada

Before the construction of the Saint Lawrence Seaway (SLS), basic data on fish biodiversity in the SLR were lacking. However, it has become obvious that the SLS has reduced fish diversity by hampering or making impossible fish migration, and by destroying swift-water spawning grounds. Raising water levels has destroyed flood plain habitat and spawning grounds for other species (pike). In fluvial Lake St François (Figure 19.1), damming has increased the aquatic plant biomass, probably by increasing sedimentation, and thus augmenting the amount of nutrients available to plants. In turn, increased plant biomass has probably elevated friction (resistance to water flow), which has further slowed down water flow (Morin et al., 2000).

Like the Baltic Sea, the Upper Estuary is home to a relatively small number of fish species, about 60 in the Estuary and 100 in the Baltic Sea, because of the physiological stress imposed on both freshwater and marine species by the estuarine environment, e.g. low salinity, low water temperature and ice cover in winter. Because of the small number of species, both systems are less resilient to various stressors such as invasion by foreign species, increased temperature and lower salinity (the latter two changes will probably occur due to global warming). Systems with higher number of species have higher resilience because there is a higher probability that among a large number of species, some will have overlapping ecological functions. Consequently, if a species is affected by an environmental change, the chances are that another species with similar functions will be resistant to that change. The number of individuals belonging to the species that has resisted the change will increase, filling up the gap left by the reduced number of organisms of the sensitive species. As a result, the system as a whole will be less affected than a system with fewer species.

In the SLE, only two species, the Atlantic sturgeon and the American eel, are exploited commercially (total catch less than $1 million in 1995). The SL is the northernmost river inhabited by the Atlantic sturgeon. This fish, the largest freshwater fish in Quebec and on the eastern seaboard of North America, spends most of its life in sea water, but migrates to freshwater to breed. Prior to 1967, the annual catch of Atlantic surgeon in the SLE was 20 to 40 tonnes. Between 1967 and 1975 no sturgeon was caught. Afterwards, the commercial catch progressively increased up to 120 t in 1993. Thirty-five commercial fishermen still catch 6,000 sturgeons a year nowadays in the SLE.

Various causes were cited to explain the 1967-1975 apparent extinction: dredging, deposition of dredging material in the spawning zones, chemical pollution, especially pesticides, over-exploitation of the population, or a combination of these factors (Caron and Tremblay, 1999). Before 1998, nothing was known about the sturgeons'

Figure 19.1. Lake St Francis (Saint-François). This lake receives its water almost exlusively from Lake Ontario. Upstream, its water flow is regulated by three large dams, Moses-Saunders (which produced hydroelectricity, Long-Sault et Iroquois. Downstream, Lake St Francis outflow is regulated by Beauharnois and des Cèdres hydroelectric dams. provided by Nathali Gratton and Denise Séguin, Environment Canada. Source: Pelletier, M. 2002.

spawning grounds except that these fish must spawn somewhere in the SL or its tributaries, yet no spawners (breeding fish) had been ever observed in the SLR. Only in 1998 were spawning grounds finally discovered by combining netting and radio-tagging spawning sturgeons. Biologists were very surprised by what they found: sturgeons not only spawn along the heavily dredged and transformed shoreline bordering Quebec City, the second major city of Quebec province, but they also feed and probably spawn right in the middle of Quebec City harbour (Hatin et al., 2002).

Many of the other fish species living in the SLR are threatened because of degradation, reduction of spawning grounds and probably contamination. American eels are heavily contaminated by their sojourn in the GL; they are found in the Estuary only in the autumn, when they migrate out of the GL to the Sargasso Sea[1]. The spawning grounds of rainbow smelt have been degraded by agriculture; Atlantic herring have been overfished in the Gulf in the 1970s and Atlantic tomcod populations, which spawn in winter in SL tributary rivers, have dwindled. They have been found with severe oral lesions, which may have contributed to this decline (Lair et al., 1997).

[1] SLE beluga prey on eels during this passage and become contaminated with mirex, a pesticide found only in the GL because originally used there (Hodson et al., 1994).

The Gulf of St Lawrence is home to Atlantic cod (*Gadus morhua*) whose populations, which have been historically and economically very important, diminished alarmingly in the 1990s. The cod fishery contributed to the establishment and survival of the first Europeans settlers in Atlantic Canada. In addition, cod from the North Atlantic and SL Gulf has been a very popular food in Catholic Europe (France, Spain, and Portugal) since the 16th century. Between the 19th century and the Second World War, French, Spanish and English vessels caught up to 600,000 tonnes a year, mostly in Canadian waters. The use of trawls, which had been banned until the late 1940s, and the arrival of foreign fishing fleets which often included factory ships, resulted in an annual catch of up to 1.4 million tonnes in the late 1960s, which explains the dramatic 1990 decline, down to yearly catches of 40,000 tonnes. In 1992, this situation led to a moratorium, still in place despite strong opposition from the fishing industry. Even the limited ‘sentinel’ fisheries that are still allowed probably hamper recovery (Shelton and Sinclair, 2008). While overfishing has been the most important factor in this decline, other factors such as hypoxia (associated with the presence of deep, O_2-poor warmer water), lower air temperature in the 1990s, higher number of cod-eating grey seals (which have replaced cod as top predators), diseases and contaminants (including biotoxins) may also have played a role. However, causes other than overfishing are marred with many uncertainties whereas in contrast, the negative impact of overfishing is only blurred by probable underestimation: cods are bycaught in other fisheries: shrimp, halibut and plaice; in addition, sub-legal sized cods are thrown back in water (and die), and lost nets are responsible for 'ghost fishing' of cod and other species (Savenkoff et al., 2004). The role of seal predation on cod populations is controversial: models have determined that the reduction in cod populations is due to a higher mortality of large (adult) cods, not of juveniles. This is an important consideration because grey seals do not feed on adult cod but rather feed on juveniles. In addition, the total amount of cod eaten by grey seals (which may be underestimated) does not seem to account for the estimated increase in cod mortality. It has been proposed that grey seals even contribute to cod survival because they also eat herring and mackerel, which feed on cod larvae and juveniles.

The role of lower water temperature in cod decline is equally fraught with doubts: the decline in cod populations started in the late 1970s and early 1980s, much earlier than when lower temperatures were first detected, in the late 80s (Chouinard et al., 2005).

The fact that cod is a bottom dwelling fish has at least two implications on its decline: 1) the negative impact of trawls on cod habitat should be given a prominent importance, yet only vague measures have been proposed to mitigate the impact of trawl, for instance 'limiting activities that could jeopardise habitats', applying techniques to reduce the negative impact of harvesting on fishing grounds and habitats, 'closing particularly sensitive areas to any potentially harmful activities' (DFO 2005); 2) Mathematical models of population dynamics have detected increased natural mortality in cod. Researchers have rejected any role for viral infections or biotoxins based on the lack of evidence supporting these factors (Chouinard et al., 2005). Yet, this author's opinion is that it is unlikely that mortality from chronic (and even acute) infectious diseases or toxins would have been detected. Indeed, the detection of mortality in free ranging fish populations (and in wildlife in general) would require the bodies of dead bottom-dwelling fish to be recovered and submitted to diagnostic laboratories, an improbable event in the case of cod. Again, 'absence of evidence' is not synonymous with 'evidence of absence', especially where causes of mortality in wildlife are concerned (Wobeser, 1994).

The fishing industry still opposes a complete moratorium on cod fisheries because it would immobilise highly capitalised (e.g. very expensive) fishing vessels and gear, and of course would deprive fisherman of their income. Yet the overcapacity of the fishing fleet has been built gradually over decades, based on a seemingly inexhaustible resource. Continuously and increasingly investing in heavy equipment is inherent to industries that are based on the commercial exploitation of free-ranging animals, terrestrial or aquatic. This trend is based on the tenuous idea that resource can grow infinitely. This leads to a self-amplifying problem. When a given animal population or species begins to be harvested commercially, it first copes successfully with growing harvest by increasing reproduction through different mechanisms. However the population invariably ends up declining because the biological limits on reproduction cannot match the industrial harvest, the growth of which is limited only by available capital. As the population declines, the industry struggles, and invariably its representatives require governments to increase catch quota normally imposed on fisherman, precisely when those should be lowered. More often than not, the government gives in. This vicious circle accelerates population declines, which may go beyond recovery threshold (Stephane Lair, U de Montréal, pers. comm.). Another reason for the opposition of fishermen to the moratorium, probably justified, is that censuses are not sufficiently frequent (only once a year), and have been carried out on various, antiquated vessels instead of modern, standardised well-equipped vessels, which is necessary to ensure consistency in censuses (Vanderzwaag and Hutchings, 2005).

The weaknesses of Canadian fisheries management that have led to the collapse of cod populations have been summarised harshly: 'failures to heed scientific advice because of socio-economic and political pressures; adoption of a top-down approach by regulators contributing to social pressures to 'beat the system,' for example through misreporting catches; exclusion of the public and NGOs from fishery decision making; allowance of too many fishers for too few fish; commitment to single species quota management as the fundamental management;

approach without recognizing all the limitations, such as lack of adequate scientific information on biomasses of fish stocks and enforcement complications; faith in linear mathematical models for setting reference points for fisheries; hesitation to base fisheries management on prudent principles; dominance of sociocultural and economic metaphors over a biological/ecological metaphor' (Vanderzwaag and Hutchings, 2005). Perhaps an inevitable conclusion to draw from the collapse of cod populations is that cod fisheries should be replaced by aquaculture because the exploitation of wildlife on an industrial scale, fish in particular, leads almost certainly to the exhaustion of the resource.

Thiamine Deficiency in Aquatic Food Chains

The Cumulative Result of Ecosystem Disruption by Clupeids?

20

John D. Fitzsimons
Great Lakes Laboratory for Fisheries and Aquatic Sciences, Burlington, Canada
Martha Wolgamood
Michigan Department of Natural Resources, Mattawan, MI, USA
Charles P. Madenjian and David B. Bunnell
United States Geological Survey, Ann Arbor, MI, USA

Clupeid fish such as herring (*Clupea* spp), shad (Dorosoma spp), and alewives (*Alosa pseudoharengus*) have long been recognised as a source of thiaminase activity in aquatic food chains throughout North America and the Baltic Sea (Deutsch and Hasler, 1943; Neilands 1947; Gnaedinger and Krzeczkowski, 1966; Ji and Adelman, 1998; Tillitt et al., 2005; Honeyfield et al. 2007; Wistbacka and Bylund, 2008). Thiaminase is an enzyme, possibly of bacterial origin (Honeyfield et al., 2002; Tillitt et al., 2005), which is thought to be involved in the destruction of thiamine in the gut of predators that consume prey fish containing thiaminase. This leads to reduced uptake of thiamine by clupeid predators and eventually thiamine deficiency if thiaminase activity is high enough and clupeids comprise a high enough proportion of the diet (Honeyfield et al., 2005). When fish containing high levels of thiaminase are fed to captive foxes or mink, this results in Chastek paralysis (Green et al., 1937; Deutsch and Hasler, 1943). Similarly on fish farms, salmonines fed fish containing high levels of thiaminase also develop thiamine deficiency (Allan, 1958; Saunders and Henderson, 1969). It should be noted, however, that in spite of their high thiaminase content, species like alewives appear to contain adequate thiamine levels to support fish nutrition (Saunders and Henderson, 1974; Honeyfield et al., 2005; Fitzsimons et al., 1998, 2005b; Tillitt et al., 2005).

Clupeids contain some of the highest thiaminase levels reported in aquatic organisms (Tillitt et al., 2005; Honeyfield et al., 2005, 2007; Wistbacka and Bylund, 2008). It is only recently, however, that the process by which clupeids exert aquatic ecosystem disruption and cause thiamine deficiency with this enzyme has become evident. In the laboratory, it is possible to cause thiamine deficiency in captive fish fed a diet consisting exclusively of clupeids (Saunders and Henderson, 1974; Honeyfield et al., 2005). In the wild, this was thought to be unlikely, however. Diets in the wild consisting predominantly of prey from a diverse prey community, either lacking or having low levels of thiaminase, were believed to reduce the overall influence of individual prey fish species containing high levels of thiaminase. As a result, having thiaminase-rich prey in the diet was thought to be tolerable for a limited period of time (see Saunders and Henderson, 1974). Predators of clupeids including Atlantic cod (*Gadus morhua*) from the Gulf of St Lawrence have apparently tolerated a diet comprised of Atlantic herring (*Clupea harengus*) when that diet also included euphausiids, shrimp, crab, mysids, capelin and polychaetes (Hanson and Chouinard, 2002). Similarly, Baltic salmon (*Salmo salar*) apparently tolerated a diet of sprat (*Sprattus sprattus*) and Baltic herring (*Clupea harengus membrus*) that also included other prey such as three-spine stickleback (*Gasterosteus aculeatus*), perch (*Perca fluviatilis* L.), smelt (*Osmerus eperlanus* L.), sandeel (*Ammodytes* sp.), bream (*Abramis brama* L.) and eelpout (*Zoarces viviparus* L.) for millennia in the Baltic Sea (Hansson et al., 2001; Wistbacka and Bylund, 2008).

In contrast to the conditions described above where predators appeared to co-exist with clupeids containing high levels of thiaminase in their diet, such is not the case for salmonines in the North American Great Lakes, as is best typified by lake trout (*Salvelinus namaycush*) (Fitzsimons et al., 2003). As a result of aggressive stocking programs, the abundance of adult lake trout in the Great Lakes has increased markedly from its former depressed state that was the result of overexploitation and sea lamprey (*Petromyzon marinus*) parasitism (Hatch et al., 1981; Holey et al., 1995; Rutherford, 1997; Hansen, 1999: Bronte et al., 2007). Despite these efforts, many of these lake trout stocks, especially in Lakes Ontario and Michigan but less so in Lake Huron, show little evidence of significant natural reproduction (Madenjian and DeSorcie, 1999; Madenjian et al., 2004; Fitzsimons et al., 2003). All of these stocks are dependent on alewives (Lantry, 2001; Madenjian et al., 1998, 2006). Predation by alewives on larval lake trout has been proposed as a major contributor to the lack of natural reproduction by lake trout in the Great Lakes, based on work conducted in Lake Ontario but the results are ambiguous, being confounded by simultaneous changes in lake trout egg deposition indices and the potential for thiamine deficiency to affect vulnerability of larval lake trout to alewife predation (Krueger et al., 1995, O'Gorman et al. 1998, Fitzsimons et al. 2009a, unpublished data). A shift in the springtime distribution of alewives to further offshore in Lake Ontario (O'Gorman et al., 2000) and away from the majority of lake trout spawning reefs in this lake was associated with a significant increase in lake trout natural reproduction (Fitzsimons, 1995; Perkins and Krueger, 1995; unpublished data). Although proof that alewife predation was exerting a negative effect, the absence of a major and sustained increase in the amount of natural reproduction by lake trout suggests that other factors in addition to alewife predation may be limiting lake trout reproduction In Lake Ontario (Fitzsimons et al., 2003; B. Lantry pers. comm.). Thiamine deficiency effects are implicated since lake trout egg thiamine concentrations remained unchanged between 1994 and 2004 (Fitzsimons et al., 2007), but the effects of the deficiency in Lake Ontario will remain of uncertain importance until there is a major change in lake trout diets and its effect on natural reproduction can be assessed. For Lake Huron, Fitzsimons et al. (2009b) reported that relief from the thiamine-reducing effects of alewives made a greater contribution than relief from alewife predation to increased natural reproduction by lake trout following the collapse of alewives in this lake. As a result of their high thiaminase content, the potential for alewives to have unique depensatory effects on predators like lake trout in the North American Great Lakes may be widespread (Walters and Kitchell, 2001). An understanding of the mechanisms leading up to this depensation has remained incomplete until only recently as the implications of the dominance of a thiaminase-rich prey fish such as alewives in the diets of top predators have become more fully understood.

There are several recent, although indirect, examples from both marine and freshwater environments that when clupeids become abundant enough in the diet of a predator, they can negatively affect recruitment and this may be related to their high thiaminase content. For example the most productive period of Atlantic salmon in the north-east Atlantic Ocean was associated with the collapse of Norwegian spring-spawning herring (*C. harengus*), a species that was formerly their major food source and is known to be thiaminase-rich (Wistbacka et al., 2002; Haugland et al., 2006). Similarly, a recent increase in recruitment of lake trout occurred in Lake Huron following the collapse of alewife in this lake (Riley et al., 2007). Prior to their collapse, alewives had been one of the most important diet items of lake trout in Lake Huron (Madenjian et al., 2006). The situation in Lake Champlain mirrors that in Lake Huron in that the recent addition to and subsequent expansion of alewives in Lake Champlain reduced egg thiamine in lake trout (unpublished data) and resulted in high levels of Cayuga Syndrome (Fisher et al. 1996), a thiamine deficiency mortality syndrome, in the progeny of resident Atlantic salmon (K. Kesley, VDNR, Grand Isle, VT, pers. comm.). Prior to alewives occurring in Lake Champlain, there had been no evidence of Cayuga Syndrome in resident Atlantic salmon. Collectively, this suggests that high consumption of prey fish containing high levels of thiaminase, e.g. clupeids, can under certain circumstances impose negative impacts on fish stocks. The conditions under which this occurs and the full range of biological effects have only recently been elaborated and are the subject of this review.

Effects of Thiamine Deficiency by Life Stage and Relationship to Recruitment

As a disruptor of ecosystem function, the effect of thiamine deficiency resulting from a high thiaminase diet has few parallels in nature in terms of the magnitude of potential negative effects. Thiamine deficiency appears fully capable of causing mortality and other effects at multiple life stages and as a result has the potential to block reproduction in a variety of top predators, leading to population declines and possibly even extinctions (see Ketola et al., 2000).

For Great Lakes Basin salmonines, the acute mortality effects of thiamine deficiency affecting embryonic stages have been the most obvious and hence the most studied effects (Fitzsimons, 1995; Fisher et al., 1996; Marcquenski and Brown, 1997; Brown et al., 1998). These effects are referred to as Early Mortality Syndrome or EMS in Great Lakes salmonines or Cayuga Syndrome in Finger Lakes Atlantic salmon. The involvement of thiamine in their aetiology has been confirmed in that the syndromes can be both reduced by thiamine prophylaxis (Fitzsimons, 1995; Fitzsimons et al., 2001a; Brown et al., 2005a) and induced by thiamine antagonists (Amcoff et al., 1999; Fitzsimons et al., 2001 a, b). Their presence has been linked to reduced egg thiamine levels, which have been directly related to the amount of alewives in maternal diets (Fitzsimons et al., 1999; Fitzsimons and Brown, 1998; Brown et al., 2005b, c; Fitzsimons et al., 2007; Fisher et al., 1996; Honeyfield et al., 2005).

Mortality alone seems insufficient to explain the recruitment failure of lake trout associated with consumption of alewives. Acute mortality resulting from EMS, which ranges from approximately to 20 to 30% (Fitzsimons et al., 1999), but can be upwards of 70% (Fitzsimons et al., 2007), is generally insufficient to totally block reproduction by lake trout. Nevertheless ongoing recruitment failure in stocks exhibiting thiamine deficiency (Fitzsimons and Brown, 1998), coupled with a lack of other plausible explanations (Fitzsimons et al., 2003), suggests thiamine deficiency may be imposing effects in addition to acute mortality that can lead to recruitment failure. This notion prompted investigation into the effects of thiamine deficiency on the larval growth, foraging and predator avoidance of lake trout, because these early life history attributes have well-established links with early survival. This culminated in research reported by Fitzsimons et al. (2009a), who found a direct relationship between larval lake trout growth, foraging rate, and predator avoidance, and egg thiamine concentration. Of these effects, growth impairment was apparently the most sensitive effect of thiamine deficiency. The threshold egg thiamine concentration established by these authors for a 50% decline in growth (5.1 nmol/g) was approximately three-fold higher than what they had previously reported for 50% EMS (1.6 nmol/g) (Fitzsimons et al., 2007). By relating their growth effect threshold to the current distribution of egg thiamine levels in lake trout, these authors estimated that for Lakes Erie, Ontario and Michigan, 46, 77, and 97% of lake trout families, respectively, would spawn eggs with at least a 50% reduction in growth. This was compared with 0, 46 and 38% of lake trout families from Lakes Erie, Michigan and Ontario, respectively, having egg thiamine concentrations resulting in 50% or greater EMS. While it might be argued that reduced growth in and of itself is not directly lethal, reduced growth may contribute to mortality and ultimately recruitment failure. Growth directly affects size and size is directly related to swimming speed (Blaxter, 1986; Webb and Weihs, 1986). Reduced swimming speed makes larval fish more susceptible to predation (Blaxter, 1986; Werner and Gilliam, 1984; Zaret, 1980; Brooking et al., 1998) while reducing foraging efficiency (Blaxer, 1986; Webb and Weihs, 1986), both of which can contribute to mortality in the wild (Miller et al., 1988; Houde, 1997). In support of the notion that sublethal effects may be a significant but unrecognised impact of thiamine deficiency, Fitzsimons et al. (2009b) found that a fry emergence index (FEI) for larval lake trout in Lake Huron was directly related to egg thiamine concentration. The occurrence of EMS in their study was relatively low (e.g. <15%). They attributed the thiamine effect on FEI to a greater susceptibility of thiamine-deficient fry to predation by crayfish, which may have been exacerbated by the altered behaviour of thiamine-deficient fry making them both more readily detectable (e.g. hyperexcitability) and attacked (e.g. lethargy, loss of equilibrium).

Thiamine levels of adult salmonines feeding heavily on alewives appear to be negatively affected by such a diet based on comparisons of tissue thiamine concentrations between populations having a high proportion of alewives in their diet and populations having a much smaller proportion of alewives or no alewives at all (Brown et al., 2005d). Negative effects resulting from such depressed levels have been reported for adults of several species of salmonines. Effects include altered behaviour (Brown et al., 2005d), impaired migration (Ketola et al., 2005), impaired spawning habitat use (Ketola et al., 2009) and mortality (Fitzsimons et al., 2005a).

Mortality in the wild of adults affected with thiamine deficiency may be quite high. Fitzsimons et al. (2005a) discussed at least two instances of massive die-offs of Lake Michigan coho salmon (*Oncorhynchus kisutch*) during 2001 in Platte Bay prior to fish entering the Platte River on their spawning run in 2001. In contrast, females entering the lower part of the river in the same year as the die-offs occurred exhibited few signs of thiamine deficiency (e.g. lethargy) and had muscle thiamine levels averaging 0.97 nmol/g, which was above the 0.59 nmol/g level associated with lethargy (Brown et al., 2005d)., The upstream spawning migration of coho salmon on the Platter River affected thiamine levels since the signs of thiamine deficiency (e.g. lethargy) increased to approximately 20% in fish reaching a hatchery 15 km upstream of where fish first entered the Platte River (Fitzsimons et al., 2005a). Thiamine levels in fish exhibiting thiamine deficiency signs at the hatchery were less than half those of fish first entering the river. Associated with increased signs of thiamine deficiency was co-occurring mortality, which was significantly reduced in coho salmon given a thiamine injection upon entry into the river. Similarly for Lake Ontario Chinook salmon (O. *tshawytscha*) ascending the Salmon River in 2004-2005, there was upwards of 60% pre-spawning mortality (Everitt, 2006). Gradient and flow on the Salmon River during the time that Everitt (2006) conducted his studies was much higher than on the Platte River when Fitzsimons et al. (2005a) conducted their studies (Ketola et al., 2009). Although part of the Chinook salmon mortality occurring on the Salmon River may have been related to the effects of angling (Bendock and Alexandersdottir, 1993) or temperature stress (Richter and Kolmes, 2005), mortality was still higher than expected. Subsequent work with Lake Ontario Chinook salmon (Fitzsimons et al. 2011a; Figure 20.1) has revealed extremely low muscle thiamine concentrations in Lake Ontario Chinook salmon which are at or below levels associated with lethargy in other salmonines (Brown et al., 2005d; unpublished data). Thiamine deficiency, because it causes increased plasma concentrations of lactate (Combs, 1992), would have exacerbated the negative effects of angling and temperature stress that lead to the build-up of lactic acid in the blood and that have been associated with mortality (Wood et al., 1983). In addition, thiamine plays a central role in the production of ATP equivalents that support metabolism. As a result, during periods of vigorous activity such as that caused by angling or high temperature there may be additional demands on thiamine reserves and if these are already depleted by a thiaminase containing diet mortality may result (Fitzsimons et al., 2005a).

Relatively little is known about the effects of thiamine deficiency at the juvenile stage, even though they may be feeding on much the same prey as adults, including alewives (Madenjian et al., 1998). Juveniles appear to be more sensitive to the effects of thiamine deficiency than adults (Morito et al., 1986; Ketola et al. 2008). Recent work in Lake Ontario on the ontogeny of thiamine deficiency in lake trout suggests there is a strong potential for effects to occur throughout almost the entire period of piscivory including the juvenile period. Alewives are the most significant prey species in the diet of salmonines in Lake Ontario (Lantry, 2001) and both lake trout (Fitzsimons et al. 2011b; Figure 20.1) and Chinook salmon (Figure 20.2) show evidence of ontogenetic declines in muscle thiamine relative to reference populations. Such declines are presumed to represent a diet containing a high proportion of alewives. This is based on the knowledge that as well as being dominant in the diet of salmonines, alewives are the major prey species in Lake Ontario (Owens et al., 2003).

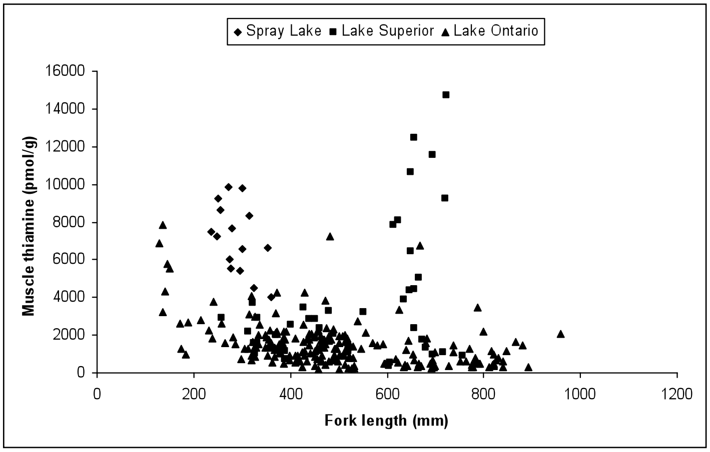

Figure 20.1. Relationship between muscle thiamine concentration (pmol/g) and fork length (mm) for lake trout from Lakes Ontario (alewife diet) and Superior (low alewife diet) and Spray Lake (non alewife diet). From Fitzsimons et al., 2011a.

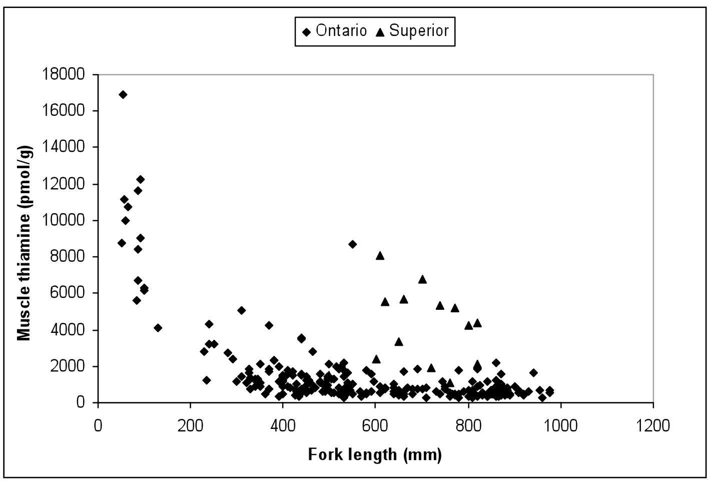

Figure 20.2. Relationship between muscle thiamine concentration (pmol/g) and fork length (mm) for Chinook salmon from Lakes Ontario (alewife diet) and Superior (low alewife diet). Source: Fitzsimons et al., 2011b.

Analysis of stomach contents and stable isotopes confirms the fact that alewives are the dominant diet item for both lake trout and Chinook salmon (Fitzsimons et al. 2011a, b). Reference populations used to establish the degree of thiamine deficiency in Lake Ontario lake trout had either lesser amounts of thiaminase-containing prey (e.g. rainbow smelt (*Osmerus mordax*) and alewives; Harvey and Kitchell, 2000) (e.g. Lake Superior) or no thiaminase-containing prey in their diet (e.g. Spray Lake). A small proportion of juvenile lake trout in Lake Ontario had muscle thiamine levels similar to those associated with acute mortality in captive adult lake trout (Brown et al., 2005d), whereas a larger proportion had levels at or below those associated with altered behaviour (Brown et al. 2005d) that could increase susceptibility to predation (Mesa, 1994; Carlson et al., 1998; Conte, 2004; Ketola et al., 2009). As a result, the effects of thiamine deficiency on juveniles may be extensive. It is noteworthy that following a decline in slimy sculpins in Lake Ontario (Owens et al., 2003), there was a coincident drop in the survival index for three-year-old lake trout their major predator (B. Lantry, USGS, pers. comm.; Owens and Bergstedt, 1994). Sculpins essentially lack thiaminase (Tillitt et al., 2005), and historically comprised the bulk of the diet of juvenile lake trout in Lake Ontario (Elrod and O'Gorman, 1991). At the time of the sculpin decline, alewives were the next most abundant prey fish in Lake Ontario and thus most likely replaced sculpins in lake trout diets. Other potential prey fish were either less abundant or declining (e.g smelt), or of such low abundance (e.g. emerald shiner (*Notropis atherinoides*), three-spine stickleback) that they were unlikely to make a contribution to lake trout diets (Owens et al., 2003). Therefore it seems reasonable to conclude that when slimy sculpins declined, juvenile lake trout switched their diet from one that was thiaminase-poor (e.g. sculpin) to one that was thiaminase-rich (e.g. alewife). As a result, lake trout may have suffered direct and indirect mortality from thiamine deficiency effects, but more work is required to confirm this.

Ascendancy of Clupeids in Aquatic Food Chains and Trophic Cascade Effects

Having demonstrated that significant negative impacts can result from thiamine deficiency in Great Lakes salmonines, potentially blocking natural reproduction for lake trout (Fitzsimons et al., 2003; Bronte et al., 2003) and Atlantic salmon (Ketola et al., 2000), emphasis has switched to understanding causes and how to manage the problem. This includes the particular reasons why clupeids such as alewives should reach such high levels of abundance to nearly dominate prey communities and as a result dominate predator diets. It is also of interest to know why clupeids of all fish should have such high thiaminase levels and what factors regulate thiaminase activity, and finally what, if anything can be done to mitigate effects.

Large dominant top predators that are at the top of food chains are successful in part because of 'cultivation' effects where they crop down forage species that are potential competitors or predators (Walters and Kitchell, 2001). Due to their relatively high fecundity and tolerance of a wide range of environmental conditions (Scott and Crossman, 1973), and inate capacity to recover after catastropic declines, clupeids have the potential to readily exploit a reduction in predation pressure from such top predators, allowing them to achieve rapid population expansion and reach high levels of abundance (Smith 1970; O'Gorman and Schneider 1986; Hutchings and Reynolds, 2004). This high abundance when coupled with diverse feeding habits (filter feeding, particle feeding; Janssen, 1976) and competition with and predation on other prey fish, allows clupeids such as alewives to suppress other prey fish (Madenjian et al., 2008; Bunnell et al., 2006). Such was the case for alewives in the Great Lakes where they have been strongly implicated in the suppression of several native species although documentation of effects on cisco (*Coregonus artedii*) until redently were largely circumstantial and thought to be without merit (Crowder, 1980, 1986; Madenjian et al., 2008; Bunnell et al., 2006). Cisco was historically one of the most important prey fish in all five of the North American Great Lakes (Fitzsimons and O'Gorman 2006). As early as the 1800s it was speculated that the expansion of alewives in Lake Ontario following the loss of Atlantic salmon the top predator (Smith 1892), had resulted in dramatic declines in cisco (Mather, 1881), considered to be one of the most abundant species prior to the invasion of the lake by alewives in 1873 (Rathbun and Wakeham, 1897). However, Madenjian et al. (2008) presented evidence that cisco are only minimally affected by alewife because the time window for alewife to predate on larval cisco is small as a result of limited overlap in habitat. Nevertheless, Dunlop et al. (2010) provided evidence for Lake Huron of increased abundance of cisco following the collapse of alewives in this lake. Their observations when combined with those of others finding increased cisco abundance following the decline of alewives (Dobiesz et al., 2005; Warner et al., 2009) stongly implicate alewives in supressing cisco populations. Although the shift from alewife to cisco in Lake Huron suggests relief from either predation or competition, the nature of the negative interaction between alewives and ciscoes has not been established. It may involve changes in the abundance of large zooplankters such as *Limnocalanus,* since alewife planktivory is highly size-selective and has been implicated in regulation of *Limnocalanus* in Lake Michigan (O'Gorman et al., 1991, Barbiero et al., 2009). Alternatively a negative interaction may be the result of reduced predation by alewives on young of the year but not egg and larval stages of cisco, since the seasonal depth distribution of alewives does not overlap the spawning habitat of cisco (Wells, 1968).

As a result of their potential to attain high population abundance while suppressing alternate prey species, clupeids can in turn dominate the diets of predators. This would be more likely to occur when the population control formerly provided by a predator was interrupted for a sufficient period of time to allow clupeid populations to build up to a high level and become dominant. For example, evidence of large alewife populations, usually evident as major die-offs (Brown, 1972; O'Gorman and Schneider, 1986), were not documented until the 1960s and 1970s even though alewives first entered the Great Lakes in the 1800s (Ketola et al., 2000) and by the 1950s were found in all of the Great Lakes. In Lake Michigan, proliferation of alewives did not occur until approximately one decade after the crash of the lake trout population in this lake, which was due to a combination of overfishing and sea lamprey parasitism (Hatch et al., 1981; Holey et al., 1995; Rutherford, 1997). This pattern was repeated throughout the Great Lakes (Smith, 1995). High clupeid population abundance can be sustained even if predator abundance returns to near normal, as is evident for the Great Lakes (Madenjian et al., 2002; Mills et al., 2003). Despite the restoration of lake trout stocks in Lake Michigan by stocking, first begun in the mid-1960s and which continues today (Holey et al., 1995), large populations of alewives persist in the face of large adult stocks of lake trout that have resulted from over four decades of stocking, and which consume primarily alewives (Madenjian et al., 1998, 2002; Bronte et al., 2007).

The pattern of ecosystem change associated with a build-up in clupeid abundance and development of thiamine deficiency that follows declines in the abundance of the top predator, appears to be similar across ecosystems and has been attributed to multi-level trophic cascades (Harman et al., 2002; Casini et al., 2008, 2009). Recent changes in the Baltic Sea show a similar chronology to the Great Lakes, of a build-up in clupeid abundance, in this case sprat and Baltic herring, following the loss of the top predator, in this case Baltic cod. The loss of cod has been related to a period where saltwater intrusions from the North Sea did not occur. The ensuing stagnation of the Baltic Sea resulted in a significant decline in Baltic cod as a result of embryonic mortality caused by low dissolved oxygen (Mackenzie et al., 1996). The decline in cod was followed by a dramatic increase in the abundance of sprat and Baltic herring (Sparholt, 1994). Despite partial recovery of dissolved oxygen levels following a major inflow to the Baltic Sea from the North Sea in 1993 (Matthaus and Lass, 1995), cod recruitment remained low, suggesting that other processes were limiting reproductive success (Bagge, 1994). Documentation of extensive cod egg predation by sprat and herring (Koster and Mollmann, 1997) and the resulting development of cod recruitment models (Koster et al., 2001 a, b) revealed that egg predation had become a strong, albeit not the only regulator, of cod recruitment in the Baltic Sea.

The relative influence of clupeid egg predation on cod as a depensatory factor of cod in the Baltic Sea seemed much stronger than possible thiamine deficiency effects. High consumption of clupeids by cod appeared to be without consequence of thiamine deficiency induced embryonic effects usually associated with high clupeid consumption. Instead cod egg thiamine status was not affected (Amcoff et al., 1999), nor was larval survival responsive to thiamine prophylaxis suggestive that cod were able to maintain adequate thiamine nutrition (Mellargaard, 1996; Nissling and Vallin, 1996). Thiamine deficiency effects were in fact more evident in other apparently more sensitive species such as Baltic salmon and sea trout (S. *trutta*) that presumably have a greater proportion of clupeids in their diet now than before cod declined, as evidenced by reduced egg thiamine concentrations (Amcoff et al., 1999). The build-up in clupeid bimass, elevated occurrence in the diet of salmonines and resulting thiamine deficiency is very similar to the situation for salmonine top predators in the Great Lakes.

Factors Affecting Thiaminase Activity in Clupeids and Their Influence on Thiamine Deficiency

The particular reasons why clupeids consistently have some of the highest thiaminase activity observed amongst telosts remains unclear (Nielands, 1947; Greig and Gnaedinger, 1971; Tillitt et al., 2005; Honeyfield et al., 2007; Wistbacka and Bylund, 1998). In a review of phylogenetic and ecological characteristics associated with thiaminase activity in Noth American Great Lakes fishes, Riley and Evans (2008) found, based on diverse measures of thiaminase activity, that taxonomically more ancestral species (Anguilliformes, Clupeiformes, Cypriniformes, Siluriformes) were more likely to show thiaminase activity than the more derived species (protoacanthopterygians and neoteleosts). In addition, these authors found that species that fed at lower trophic levels and occupied benthic habitats appeared more likely to show thiaminase activity.

Within alewives, where most work has been done to understand thiaminase dynamics, there is evidence that thiaminase activity varies among seasons, within and among lakes, and is related to the size of alewife (Tillitt et al., 2005; Fitzsimons et al., 2005b) suggesting that a variety of factors may be involved in modulation of thiaminase activity. Whether such factors are related to variation in diet, habitat, or some other factor or factors, is unclear. Fitzsimons et al. (2005b) found that thiaminase activity of Finger Lakes alewives was inversely correlated to lipid level and that lipid level was correlated with lake chlorophyll-*a*, a measure of lake productivity. As such, lake productivity through its influence on lipid content may influence thiaminase activity. Effects of lake productivity on alewife lipid content is consistent with the timing of a decline in energy density for Lake Ontario alewives between 1978 and 1990 (Rand et al., 1994). For alewives, energy density is directly related to lipid content (Rottiers and Tucker, 1982). Rand et al. (1994) attributed the decline in energy density of alewives, especially during the latter part of the period 1978 to 1990, to a decline in lake productivity, most likely due in part to a decline in zooplankton density (O'Gorman et al., 1997).

Fish lipid content is known to be affected by the lipid content of the diet and feeding rate (Madenjian et al., 2000). Hence the higher thiaminase activity noted by Tillitt et al. (2005) in Lake Michigan alewives in spring relative to summer and autumn may be a reflection of either the lower lipid levels observed in alewives during the spring relative to the summer and autumn (Flath and Diana, 1985), or the lower feeding activity noted for alewives in the spring relative to the summer and autumn (Stewart and Binkowski, 1986), or some combination of these. Thiaminase activity of alewives collected in the winter, the period of lowest feeding activity (Stewart and Binkowski, 1986), was twice that of alewives collected during the summer, when feeding activity is expected to be much higher (unpublished data).

The apparent relationship between lipid and thiaminase activity may be as a result of the effect of lipids on the immune system and health of an organism and possible regulation of thiaminolytic bacteria which have been isolated from alewives (Honeyfield et al., 2002). Fluctuations in alewife thiaminase activity resulting from fluctuations in feeding activity could in turn lead to fluctuations in the expression of thiamine deficiency in alewife predators. Undernutrition due to insufficient intake of energy and/or macronutrients due to deficiencies in specific micronutrients can impair the immune system, suppressing immune functions that are fundamental to host protection (Marcos et al., 2003). Sheldon and Blazer (1991) reported that bactericidal activity in channel catfish (*Ictalurus punctatus*) was positively correlated with dietary highly unsaturated fatty acids (HUFAs). Eicosanoids, which control inflammation and immunity, consist of prostaglandins, prostacyclins, thromboxanes and leukotrienes and are primarily derived from the fatty acids arachidonic and linoleic acids (M. Arts, Environment Canada, pers. comm.). A relationship between health and thiaminase activity has been confirmed for common carp (*Cyprinus carpio*), which exhibited elevated muscle thiaminase activity after injection with live *Aeromonas salmonicida* (Wistbacka et al., 2009).

Tillitt et al. (2005) felt that there was strong evidence for a bacterial origin for the thiaminase found in fishes. They based this on the thiaminase gene having been cloned in *Paenabacillus thiamimolyticus* (Abe et al., 1987), these bacteria having been isolated and cultured from the alewife digestive tract (Honeyfield et al., 2002), and the fact that the distribution of thiaminase in fishes is consistent with areas where bacteria are concentrated in fishes (Fujita, 1954; Zajicek et al., 2005). Further, these authors stated that the biochemical characteristics of thiaminase, including temperature and pH optima in fishes, particularly Lake Michigan alewives, were consistent with a bacterial source (see Zajicek et al., 2005). Given the possible role of lipids in immune function, high lipid levels may be involved in down-regulation of thiaminase-producing bacteria, whereas low lipids may be involved in the up-regulation of thiaminase-producing bacteria. This in turn may influence the occurrence of thiamine deficiency in predators, since fluctuations in the thiaminase activity of a particular prey species may lead to fluctuations in the amount of thiamine assimilated from prey species by a predator. Honeyfield et al. (2005) found that thiamine levels in the eggs of lake trout fed mixtures of alewives, which are thiaminase-rich, and bloaters, which are thiaminase-poor, were directly related to the average amount of thiaminase activity in the diet.

There appears to be support for the notion that lipid can influence thiaminase activity in clupeids in the wild and affect their potential to cause thiamine deficiency effects. The importance of diet and specifically its influence on lipid stores and resulting thiaminase activity was suggested in work by Wistbacka and Bylund (2008). These authors found, albeit indirectly, that the thiaminase activity of Baltic herring appeared to be inversely related to their lipid content. During the period 1979-1991 when the M74 prevalence in Baltic salmon was low, the fat content of Baltic herring in the main basin of the Baltic Sea was almost twice as high as in the most serious M74 period 1991-2002 (Wistbacka and Bylund, 2008). Wistbacka et al. (2002) had earlier indicated that Baltic herring, with their high thiaminase activity, were the immediate causal factor for the M74 syndrome in Baltic salmon, and this was the case for both time periods.

Variation in prey fish lipid stores and its effect on thiaminase activity may also be responsible for variation in EMS in coho salmon. In Lake Michigan, there was a dramatic upward shift in the occurrence of EMS in coho salmon after 1990, after dreissenids had invaded the lake (Brown et al., 2005b). Coho salmon in Lake Michigan eat alewives almost exclusively, feeding almost exclusively on large alewives in the second year of life prior to spawning (Madenjian et al., 1998a, b). The lipid content of large (≥ 120 mm) alewives dropped by upwards of 50% following the invasion of Lake Michigan by dreissenids in the late 1980s (Hondorp et al., 2005; Madenjian et al., 2006b). Madenjian et al. (2006) attributed the drop in lipid content to the decreased importance of *Diporeia* in the diet, most likely the result of a decline in *Diporeia* in Lake Michigan (Nalepa et al., 2005, 2006). *Diporeia* is relatively high in lipid content (Hondrop et al., 2005) compared with other invertebrates and a decrease in its importance in the diet of alewives would be expected to lead to decreases in both lipid content and energy density of alewives, since the two are correlated (Rottiers and Tucker, 1982; Madenjian et al., 2000).

Diet may also be a direct source of thiaminase for fish. Increased thiaminase activity was measured in fish exposed to toxic blue-green algae containing thiaminase although the mode of uptake was not described (Arsan, 1970; Arsan and Malyareevskaya, 1969). Zajicek et al. (2005) measured thiaminase activity albeit at low levels in net plankton, *Mysis* and *Diporeia* from lakes Michigan and Superior. All of these biota are potential prey of alewives (Hondorp et al., 2005) More comprehensive sampling in Lake Michigan (unpublished data) found thiaminase activity in net plankton and *Mysis* of up to 20% of the average measured in alewives from this lake (Tillitt et al., 2005). Furthermore, while thiaminase levels were similar in net plankton and *Mysis*, they were threefold higher than in *Diporeia* (unpublished data), suggesting that if diet was an important vector for thiaminase activity in alewives, the composition of the diet could be quite important.

Attempts at experimentally determining the role of different factors in controlling thiaminase activity have had little success but have provided some surprising results in terms of the dynamic nature of thiaminase activity in clupeids. Experimentally stressing Seneca Lake alewives in the laboratory by either varying the salt content of holding water or food limitation in order to induce stress and affect circulating white blood cells (Pickering, 1984; Barton et al., 1987) did not affect their thiaminase activity (Lepak et al., 2008). However, Lepak et al. (2008) found that the thiaminase activity of alewives held in the laboratory and fed a thiaminase free diet prior to experimentation was over two-fold higher than the thiaminase activity of alewives immediately after collection from Seneca Lake. The authors were unable to provide an explanation for this difference. They felt it was unrelated to stress as alewives grew well in the laboratory with little mortality, which they would not have expected had the fish been under stress. Similar effects were seen with Baltic herring held in captivity for 25 days, where there was an elevation in thiaminase activity compared with that of herring immediately after capture (Wistbacka and Bylund, 2008). Whether such changes involve effects on immune function, possibly by modulation of eicosanoid biosynthesis mediated by stress or trauma, remains to be determined. Lack of identification of a factor or factors associated with captivity that causes elevation in thiaminase activity is a major impediment to experimentation and to further understanding of the importance of the regulation of thiaminase activity in alewives.

Relationship between Alewife Abundance and Thiamine Deficiency

Given the ability of alewives to cause thiamine deficiency in proportion to their importance in the diet of laboratory fish a similar relationship would be expected among wild populations although such a relationship if it exists may have changed over time (Honeyfield et al. 2005). Fitzsimons et al. (1999) correlated a measure of alewife abundance with EMS, an indicator of thiamine deficiency; for Lake Michigan these authors reported a significant but negative and weak correlation(r^2=0.14) between trawl catch abundance of adult alewife and the occurrence of EMS in coho salmon that ranged from 0 to close to 100%. Of the salmonines found in the Great Lakes coho salmon show the highest consumption of alewives so are likely to be the most responsive to changes in alewife abundance (Jude et al. 1987; Madenjian et al. 1998b; Lantry 2001). These authors used EMS as their measure of thiamine deficiency because of its strong relationship with egg thiamine (Hornung et al., 1998; Honeyfield et al., 1998) and the lack of historical data on thiamine levels at the time. Fitzsimons et al. (1999) were unable to find similar negative correlations between alewife abundance and the occurrence of EMS in Lake Michigan lake trout, which they attributed to the greater diet diversity of lake trout compared to coho salmon (Jude et al. 1987; Madenjian et al., 1998). The results of Fitzsimons et al. (1999) however, seem to run counter to the association established between alewives in the diet and reductions in egg thiamine (Fitzsimons and Brown, 1998; Honeyfield et al., 2005). Moreover the diet of salmonines seems to be opportunistic (Warner et al. 2008) such that salmonines should make the greatest usage of the most abundant food source. Hence during years of high alewife abundance, salmonines should make the greatest usage of alewives and this in turn should result in high EMS, not low EMS, unless other factors are at play. Its noteworthy however, that the data of Fitzsimons et al (1999) was based on all sizes of alewives and it has been established that in the their second year of life, coho salmon fed almost exclusively on large (>120 mm) alewives although smaller amounts of small alewives are also consumed (Madenjian et al. 1998). Moreover the lipid content of large alewives (Madenjian et al 2006b) declined midway through the the time series used by Fitzsimons et al. (1999) and this may have affected alewife thiaminase activity and altered their ability to cause thiamine deficiency (see Fitzsimons et al. 2005b).

Due to the ongoing uncertainty as to the relationship between alewives and the occurrence of EMS in feral fish we reevaluated the relationship between alewife abundance and the occurrence of EMS in families of coho salmon for Lake Michigan using new information. For this we focused on the biomass of large alewives, ages one to three, as they accounted for almost 90% of the biomass of large alewives in Lake Michigan between the ages of one and six during the period 1999 to 2007 (Warner et al. 2008). Alewife biomass for a given year and age was determined from abundance at age three when alewives are fully recruited to bottom-trawl gear using the methods described in Madenjian et al. (2005). We used trawl data collected by the Great Lakes Science Center (see Madenjian et al., 2005). Data on the proportion of coho salmon families with EMS were derived from the records of the Michigan Department of Natural Resources, Fish Health Laboratory (see Honeyfield et al., 1998), which has maintained long-term observations of the occurrence of EMS in families of coho salmon ascending the Platte River. We focused on coho salmon ascending a single river since Wolgamood et al. (2005) had documented among river variation in EMS for Lake Michigan coho salmon. For the purposes of this analysis

Table 20.1. Summary of number of Lake Michigan (Platte River) female coho salmon sampled, proportion of families developing EMS, and mean EMS in affected families for the period 1997 to 2007 (M. Wolgamood, Michigan Department of Natural Resources, Mattawan, Michigan, unpublished data).

Year	N	Percentage of families affected	Mean EMS (%) in affected families
1999	30	96	100
2000	30	100	84
2001	22	100	100
2002	25	8	32
2003	20	5	53
2004	21	33	57
2005	27	4	22
2006	30	60	73
2007	24	42	88

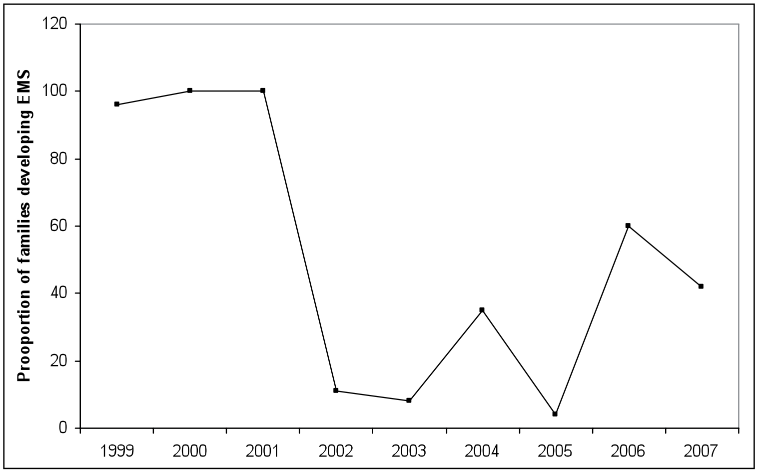

Figure 20.3. Annual variation in the proportion of coho salmon families with EMS ascending the Platte River for the period 1999 to 2007 (Source: M. Wolgamood, Michigan Department of Natural Resources, Mattawan, Michigan, unpublished data).

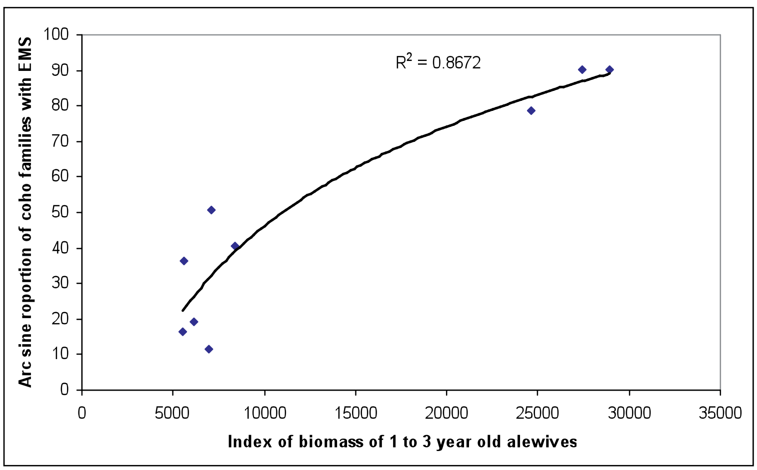

Figure 20.4. Relationship between the arc-sin of the annual proportion of coho salmon families ascending the Platte River with EMS and estimated biomass of alewives ages one to three based on bottom trawling for the period 1999 to 2007 (Source: USGS, Ann Arbor, Michigan, unpublished data).

we defined EMS as any family having a hatch to feeding fry mortality greater than 20% (Brown et al., 2005c). EMS data are summarised in Table 20.1 and Figure 20.3.

Using a restricted range of ages for alewives (ages one to three) that better reflects those consumed by coho salmon (Madenjian et al. 1998) and constitutes a major portion of the biomass for this species in Lake Michigan (Warner et al. 2009) we found that the occurrence of EMS in coho salmon families was related to alewife biomass by an exponential relationship (F=45.4, p<001; Figure 20.4). The use of a restricted age range and the greater range in biomass of alewives for this time series relative to historic (Figure 20.5), may have increased the likehood of finding a positive relationship between the occurrence of EMS and alewife biomass that was expected based on earlier observations linking elevated EMS and reduced egg thiamine with alewife consumption (Fitzsimons and Brown, 1998, Honeyfield et al., 2005). The relationship here suggests that during periods of extreme alewife abundance that from the historical record appear to be rare, most coho salmon families may show some level of EMS with corresponding effects on reproduction although these are uncertain since the amount of EMS in a given family does not appear correlated to the proportion of families exhibiting EMS (Wolgamood et al., 2005). During periods of lower alewife abundance it's evident

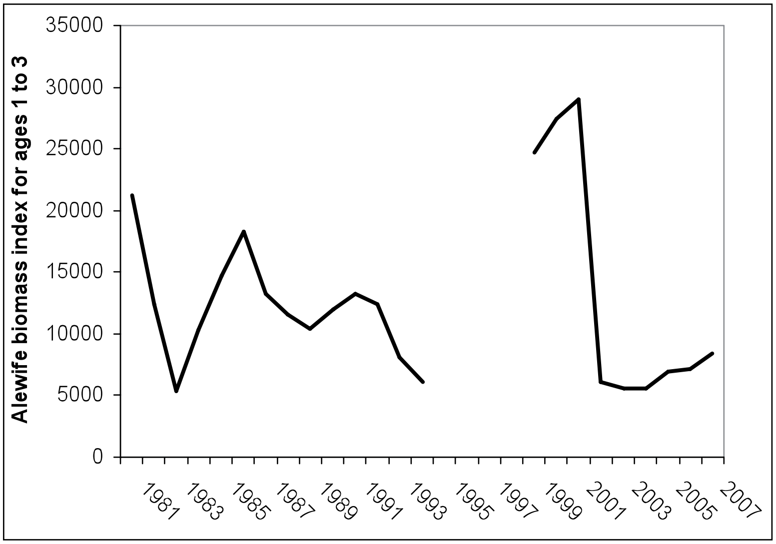

Figure 20.5. Biomass of alewives ages one to three based on bottom-trawling in Lake Michigan during the period 1981-2007 (Source: USGS, Ann Arbor, Michigan, unpublished data).

that a greater proportion of coho salmon families would not be expected to show EMS although thiamine deficiency effects on other endpoints for coho salmon are uncertain. Fitzsimons et al. (2009a) noted for larval lake trout that growth was a more sensitive endpoint to the effects of thiamine deficiency than EMS. Nevertheless natural reproduction by coho salmon in Lake Michigan although limited, has been reported for some time (Madenjian et al., 2002). Although the mechanisms involved in limiting the occurrence of EMS among coho salmon families during periods of reduced alewife biomass remain unclear they may involve greater diet diversity as suggested by Brown et al (2005c). Increased utilization of prey fish in the diet having lower (e.g. rainbow smelt) or no thiaminase (e.g. yellow perch) (Jude et al., 1987; Madenjian et al., 1998b; Tillitt et al., 2005) would conceivably result in a greater uptake of thiamine from the diet.

Future Directions

Attempts at experimentally determining the factors controlling thiaminase activity in alewives have yielded few tangible results on the factors controlling thiaminase activity in alewives (Lepak et al., 2008; unpublished data). An understanding of controlling factors could potentially be used to develop management actions (e.g. increase lake productivity to affect alewife lipid level) to regulate alewife thiaminase activity to a level consistent with natural reproduction by their predators (see Fitzsimons et al., 2007). That such a level exists is suggested by self-sustaining lake trout populations co-existing with alewives in at least two Finger Lakes, although additional mitigating factors may also be involved (Fitzsimons et al., 2007). As a result, other actions may be more appropriate for affecting thiaminase intake by predators until a better understanding of factors controlling thiaminase activity of particular prey species like alewives becomes available. One such action may be to restore prey diversity to the extent that the relative consumption of thiaminase-containing prey fish is considerably reduced and hence does not lead to thiamine deficiency effects in their predators. In some Great Lakes this appears be occurring naturally with the invasion and rapid population increase in the aquatic invasive species round goby (*Neogobius melanostoma*) (Schaeffer et al., 2005a). This species apparently has low thiaminase activity based on collections in Lake Michigan (Tillitt et al., 2005). In western Lake Ontario after a ten year period during which lake trout fed almost exclusively on alewives, as many as 20% of lake trout now appear to be consuming gobies based on stomach content analysis (Fitzsimons et al., 2009c). This appears to be sufficient to affect lake trout egg thiamine levels, since average egg thiamine concentrations of lake trout collected during 2007 were almost two-fold higher than in 2003, when gobies were probably absent from the diet (unpublished information). Similarly in Lake Huron, egg thiamine levels in lake trout stocks from the Thunder Bay area that fed almost exclusively on gobies were over twice that of offshore stocks of lake trout that fed almost exclusively on alewives (J. Johnson, MI DNR, and D. Honeyfield, USGS, pers. comm.).

A passive rather than a directed approach to the restoration of the prey fish community to eliminate thiamine deficiency effects may, however, have unintended consequences on the reproductive success of predators. This seems to be the case for gobies and lake trout. Large populations of gobies can have negative consequences for lake trout, as gobies are significant lake trout egg and fry predators (Chotkowski and Marsden, 1999; Fitzsimons et al., 2006). Gobies have been associated with the near elimination of emergence of fry for at least one lake trout spawning reef in Lake Ontario (Fitzsimons et al., 2009c), although additional work is required to determine the spatial extent of lake trout egg and fry predation by gobies in the lake.

Restoration of native species like ciscoes, which lack thiaminase activity, has been advocated (Fitzsimons and O'Gorman, 2006), but impediments to restoration of wild stocks in the Great Lakes are not always clear but as pointed out by Dunlop et al. (2010) may well involve alewives. Accordingly the presence of alewives may make it unlikely that restoration of native prey species like ciscoes would be fully successful. Lake Huron had the lowest abundance of alewives (with the exception of Superior) before alewives crashed there in 2002-2003 yet the abundance of cisco remained low (Dobiesz et al. 2005). It was only after the alewife population crashed in Lake Huron that there has been consistent evidence of spatially and temporally extensive recruitment of cisco in this lake. In contrast, for Lakes Ontario and Michigan, where alewife abundance is much higher, cisco stocks are spatially restricted and show only limited recruitment (J. Hoyle, ON MNR, R. Claramunt, MI DNR, pers. comm.).

Although the complete elimination of alewives from the diet of salmonines may seem like an extreme management action, such a step may be necessary in order to restore normal thiamine nutrition for top predators like lake trout which appear to be more sensitive to the thiamine lowering effects of an alewife diet than Chinook salmon (Fitzsimons et al., 2007). It is not clear how effective restoration of essentially thiaminase-free prey fish would be in restoring thiamine levels in predators if alewives were still present and still being consumed. During the 1980s and 1990s, the prey community of Lake Michigan was dominated by bloater chub (Madenjian et al., 2002) yet lake trout continued to feed as heavily on alewives then as in the 1970s, when bloaters were scarce. This dietary preference was reflected in egg thiamine levels as well (Fitzsimons and Brown, 1998; Honeyfield et al., 2005). In Lake Huron, it was only after a major collapse in alewife stocks occurred (Shaeffer et al., 2005b), forcing lake trout to adopt new diet choices that lacked alewives, that there was a measurable increase in egg thiamine levels (D. Honeyfield, pers comm.; Fitzsimons et al., 2009b). Thiamine levels are now adequate for reproduction, as indicated by increased, spatially extensive and sustained natural reproduction by lake trout in Lake Huron (Riley et al., 2007). Although other factors such as relief from predation of alewives on larval lake trout may also have been involved (see Krueger et al., 1995), Fitzsimons et al. (2009b) concluded that for Lake Huron there was little evidence to support relief from an alewife predation effect as an explanatory variable. These authors found a general lack of overlap between the timing of larval lake trout emergence and onshore migration of alewives. Although changes in lake trout spawner abundance coincident with the decline in alewives may have also contributed to increased lake trout reproduction, these authors found that the spatial and temporal patterns of natural recruits were not consistent with the spatial and temporal patterns of spawner abundance. To support their claim that relief from thiamine deficiency rather than predation may have been a major contributor to the increase in lake trout natural reproduction, Fitzsimons et al. (2009b) documented increases in egg thiamine levels following the crash in the alewife stock and showed that a lake trout fry emergence index (FEI) was positively

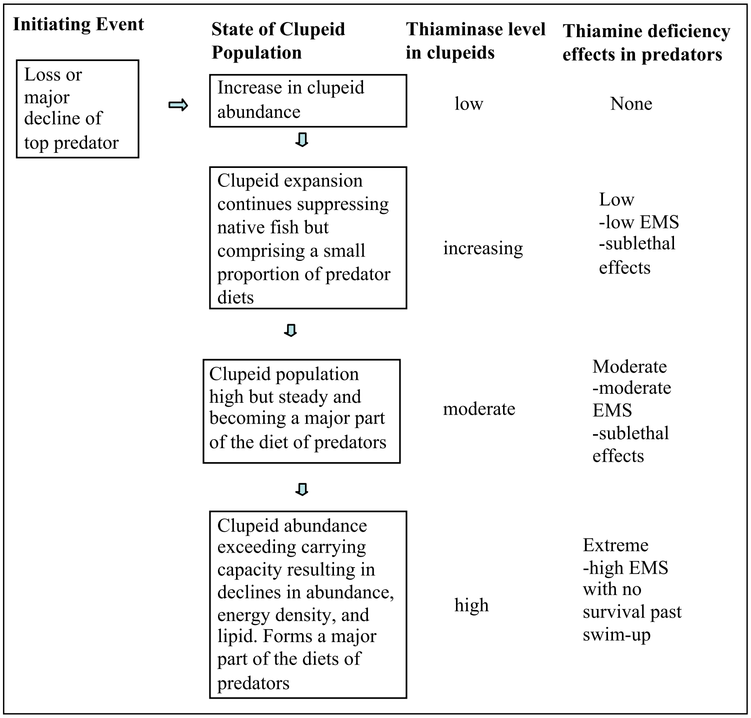

Figure 20.6. Schematic of proposed cascade of events in the expansion of clupeids, resulting thiaminase dynamics, and thiamine deficiency effects following the loss of a top predator (Source: Authors).

correlated with egg thiamine levels, while the occurrence of EMS was negatively correlated with FEI.

Conclusions

To conclude, the loss of predator control on clupeids can set off a cascade of events wherein clupeid populations expand, suppress other prey fish species, and eventually become the dominate prey fish (Figure 20.6). In this position, clupeids can dominate predator diets, especially if the abundance of predators has already been depressed by other factors. Depending on their thiaminase activity, which may be under the control of ecosystem factors such as productivity and possibly mediated through the immune system, clupeids can exert a variety of negative impacts at multiple life stages of a predator by the ensuing thiamine deficiency. These negative impacts can culminate in their most extreme case in the virtual elimination of natural reproduction in the predator, possibly leading to population extirpation as appears to have been the case for Lake Ontario and Finger Lakes Atlantic salmon (Ketola et al., 2000). Alternatively a variety of other less extreme scenarios not leading to population extirpation are also possible, depending on the severity of the thiamine deficiency and associated effects although the sustainability of these scenarios remains to be demonstrated

Acknowledgements
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Part E

Infectious Diseases at the Wildlife-livestock Interface

Authors: Val Beasley, Thomas Gillespie, Richard Lampmann, Ulf Magnusson and Torsten Mörner

Coordinating Author: Ulf Magnusson

Overview of Infectious Diseases and the Wildlife-Livestock Interface

21

Ulf Magnusson
Swedish University of Agricultural Sciences, Uppsala, Sweden

A Global Issue

The current global spread of the highly pathogenic avian H5N1-influenza (HPAI) has brought the insight that diseases can be transmitted between wild and domestic animals and to humans and a broader public. Scientifically, this insight has boosted the interest in the 'one medicine' concept, where human and veterinary medicine meet. About 58% of the infectious diseases of humans are estimated to be zoonoses (diseases that can be transmitted from vertebrates to humans) and they comprise almost three-quarters of emerging infectious diseases (Woolhouse, 2006; Jones et al., 2008).

A group of international organisations such as the WHO, FAO and OIE has proposed the framework 'One world, One health' in this context to emphasise the global dimension of some zoonoses (FAO et al., 2008a). These organisations stress the importance of a holistic view, taking into account human, livestock and ecosystem health. Ecosystem and livestock health generally have the weakest knowledge base in this zoonotic triangle, despite their utmost importance. Livestock, besides being affected *per se*, can serve as reservoirs or vectors and ecosystem and environmental variables tend to govern the intensity and spatial and temporal patterns of outbreaks. Some examples of pathogens that can affect humans, livestock and wildlife in temperate climates are shown in Table 21.1.

Epidemiology

The epidemiology of infections and diseases is highly dependant on several factors within, or in the interfaces between, human, livestock or wildlife populations. For

Table 21.1. Examples of recent reports on zoonotic pathogens in temperate climates that affect livestock and wildlife.

Pathogen	Wildlife	Livestock	Zoonosis	Reference
Rabies virus	Foxes, racoons, dogs	Cattle	Yes	Johnsson et al., 2008
Avian influenza, , H5N1	Waterfowl	Chicken, ducks	Yes	Ward et al., 2009
Leptospira sp.	Wild boar, rodents,	Pigs	Yes	Ebani et al., 2003
Brucella sp.	Wildboar, elk, hares, bison,	Pigs, cattle, sheep, goats	Yes	Godfroid et al., 2005
Mycobacterium bovis	Badgers, deer	Cattle	Yes	Boehm et al., 2009
Trichinellosis	Wild boar, bear	Pigs	Yes	Blaga et al., 2009

instance, the transmission of infections between humans is more likely to happen in areas with high population density, while transmission from livestock to humans is more likely in areas with high human and herd/farm density, especially where humans and livestock live in close proximity, as is often the case in developing countries. Furthermore, the spread of infections in livestock populations is facilitated by high herd/farm density with poor biosecurity, e.g. frequent movement of animals between herds and communal pastures. Finally, transmission between livestock and wildlife is more likely to occur if the animal population density is high and if livestock and wildlife are allowed to come into contact, as in free-range systems. Besides these and other population-related factors in the interfaces between humans, livestock and the ecosystem, there are several factors in habitat structure, farming traditions and practices, ecosystem changes etc. that influence the epidemiology of zoonoses.

The Role of Farming Practices and Habitat Destruction

Well-known examples of how farming practices, some with a strong basis in tradition or other human behaviour, can influence the spread or emergence of diseases are the Nipah virus outbreak in Malaysia 1998-1998 and the recent HPAI in Vietnam.

The novel paramyxovirus Nipah virus emerged in northern parts of Peninsular Malaysia in 1998 and caused severe febrile encephalitis in humans, with a high mortality rate (Chua, 2003). It was found that fruitbats were the natural reservoir hosts, infecting pigs which in turn infected humans coming into contact with the pigs. Notably, in pigs the virus caused encephalitis and respiratory diseases, but with a relatively low mortality rate. Therefore, infected pigs could be asymptomatic and were moved throughout the peninsula, thereby contributing to the spread of the disease. It is believed that the close association of piggeries with orchards and the design of pigsties, combined with destruction of natural habitats for the bats, increased the interface between bats and pigs and thereby contributed to the interspecies transmission of the virus.

In southern Vietnam there is a farming tradition that duck keepers move their flocks between rice paddies for feeding after the harvest. These movements of ducks can sometimes be quite far reaching, even between provinces. Ducks often do not show as clear symptoms of HPAI as chickens, but are very potent carriers of the virus, thus this farming tradition is regarded as contributing to a silent and efficient spread of the H5N1 virus in the country. When it comes to the role of wild birds in the global spread of H5N1, this seems to be limited to just a few cases, especially to, or in, Europe. Legal or illegal trade in poultry seems instead to be the major route for the virus moving between countries or continents (FAO et al., 2008b).

The Role of Wildlife

In wildlife, the occurrence and localisation of disease are determined by a variety of factors including some that relate to the host, some that relate to the causative agent and some that are considered environmental factors (Wobeser, 1994). Common environmental factors are climate, topography, soil, water and biotic features including other fauna and flora. Characterisation of the environmental conditions associated with disease and disease outbreaks is an important part to the understanding for the epidemiology of disease in wildlife. It is impossible to prepare a comprehensive list of all factors that should be taken into consideration during a disease investigation, but it is important to develop standard protocols that incorporate the complex biological, chemical and abiotic interactions of pathogens within the ecosystem (Wobeser, 1994). Cooperrider et al. (1986) and Wobeser (1994) provide excellent information about various environmental factors in relation to wild animals.

The Role of Arthropod Vectors

Pathogens in tropical areas of the Eastern Hemisphere appear more likely to have arisen from wildlife sources and to involve invertebrate vectors than those of the temperate zone, which tend to be more closely associated with diseases of domesticated animals (Wolfe et al., 2007). Crowd diseases emerged in conjunction with the development of agriculture and the concurrent increase in density of human and domestic animal populations (Diamond, 1999). The geographical dispersal of zoonoses across Europe and North America accompanied the major cultural and technological transitions from small local agrarian groups to large Eurasian nation states and from European intercontinental exploration and conquest to our present status of almost global interconnection (Weiss and McMichael, 2004). Similar to what has been found for free-living organisms, there is a negative latitudinal gradient in the species richness of human pathogens (Guernier et al., 2004). Thus, the greatest burden of vector-borne diseases is in the tropics, but the majority of hotspots for outbreaks tend to be in western Europe, northeastern United States, Japan and southeastern Australia (Jones et al., 2008).

Conclusions

There are a number of bacterial, viral and parasitic diseases present at the livestock-wildlife interface. Quite a few of these are zoonotic and are also emerging globally. To handle these diseases – for the sake of public, livestock and wildlife health – a holistic approach beyond conventional human and veterinary medicine must be taken. This approach must include ecosystem health as well as social/cultural aspects. Overall, there are considerable knowledge gaps among professional health workers about those additional aspects.

In the following sections we therefore discuss the above mentioned crucial elements in the understanding of the interface between livestock and wildlife: disease monitoring in wildlife (chapter 22); the role of arthropod vectors in disease transmission (chapter 23); and infections in relation to habitat fragmentation and species barriers (chapter 24). The control and management of these diseases is discussed in Part F (Prevention of infectious diseases in livestock and wildlife)

Monitoring for Diseases in Wildlife Populations

22

Torsten Mörner
National Veterinary Institute, Uppsala, Sweden
Val Beasley
University of Illinois at Urbana-Champaign, IL, USA

Introduction

Diseases of wildlife occur in many different forms in a wide range of animal species and populations. Diseases, when expressed in free-ranging animals, can have a significant effect on wildlife ecology. Whilst some diseases exist as symptomless, subclinical infections without obvious ecological impact and of no consequence for domestic animals or humans, occasionally there are dramatic epizootic outbreaks characterised by high morbidity and mortality (Wobeser, 1994). In addition, wild animals can be reservoirs for highly contagious and severe diseases as listed by the World Organisation for Animal Health (OIE), including many that infect domestic animals or humans. Active surveillance for known diseases of economic or public importance amongst wildlife is an increasingly well recognized need at the national and international level. Reports of illnesses or deaths involving many animals from a free-living population may represent the initial alert to the likely presence of a new disease agent. Early intervention and investigation of such unusual or unexpected disease events is essential to the goal of determining the cause and significance of the outbreak. Such reports may be the first indication of the introduction of an exotic disease agent.

Many national disease monitoring programmes include free-ranging or farmed wildlife. These programmes are usually proactive measures aimed at generally supporting national domestic animal and wildlife health, international trade in animals and animal products and protecting public health. Part of any national or international strategy for monitoring wildlife disease should include the capability to investigate mass mortality or morbidity events, investigate new disease syndromes, identify and categorise new pathogens, and monitor the status of known diseases within wildlife populations.

A disease occurring in time between some constant limits, above and below an average, is called an endemic situation (enzootic is the specific word employed to specify that the population is composed of animals). Morbidity breaking out in a place where it was previously unnoticed is called an epidemic (or epizootic for animals). Finally, when events occur in an unpredictable manner, they are called sporadic.

Investigating and Sampling for Wildlife Diseases

For many reasons, it is more difficult to monitor diseases in wildlife than in domestic animals. Wild animals are not constrained by boundaries and can range over large distances. This is particularly so for migratory birds or mammals, which seasonally move across continents or vast oceans. Opportunities to sample them may occur only briefly, at selected feeding or breeding locations.

Wild animals inhabit natural environments in various ways. Apart from colonial and herding species, which may number in the tens of thousands, wild animals also exist as smaller groups such as family cohorts, or occupy territory as solitary animals and breeding pairs.

Among the earliest surveillance programmes for wildlife disease were those established in Denmark in the early 1930s (Christiansen, 1935) and in Sweden in the 1940s (Borg, 1975). These programs were based on examinations of dead animals submitted to national veterinary laboratories. In the early 1950s, the programme in Sweden revealed problems with mercury poisoning of wildlife (Borg, 1966), and it was after this discovery that a systematic health monitoring programme of the Swedish environment was established (Mörner, 1991). Similar health monitoring programmes of wildlife, based on examinations of dead wildlife, are today in action in some other European countries such as Norway, Finland, Austria and France (Leighton, 1994), but at the moment not in all European countries. Diseases in wildlife occur in most countries. In North America, there are several regional co-operative studies on wildlife diseases, e.g. the Southeastern Cooperative Wildlife Disease Study in Athens, Georgia, USA, the USGS National Wildlife Health Center in Madison, Wisconsin, USA, and the Canadian Cooperative Wildlife Health Centre in Saskatchewan, Saskatoon, Canada. Other wildlife disease surveillance programmes exist worldwide, but the majority of these programmes are designed primarily to protect domestic animal health and trade, and only part of the work deals with wildlife health *per se*. Australia is in the preliminary stages of developing a national wildlife health network designed to report, investigate and discuss unusual mortality or disease incidents as they occur.

Morbidity (Number of Animals with a Disease)

In contrast to humans and domestic animals, in wild animals the expression of clinical signs is extremely difficult to observe and quantify. Practical difficulties can exist in determining the morbidity and mortality rates as a proportion of the population at risk. Nearly identical clinical manifestations or lesions may be caused by many different types of pathogens and their occurrence in wildlife can be extremely difficult to detect. Furthermore, wild animals normally do not show any clinical signs when observed, and thus apparently healthy individuals may be carriers of various pathogens. Therefore, disease surveillance for pathogens and morbidity studies for a particular disease cannot be based on the collection of clinical data. In order to determine the morbidity for a certain disease, many animals or specimens from a large number of animals need to be evaluated. The migration or host range movement of some wild animals that occurs before a disease outbreak is discovered sometimes makes assessment of the rates of morbidity or mortality over time almost meaningless (Artois et al., 2001).

Mortality (Number of Animals Dying from a Disease)

Most wildlife disease monitoring programmes worldwide are based on examining dead animals found in the field. These monitoring programmes provide information about the kinds of diseases that occur within a certain region and the diseases that are of greatest importance. Such programmes often detect unexplained mortalities through the collection and analysis of dead wild animals. The carcasses of animals that have died from a trauma injury or other external factors can also be used to screen for toxic, infectious or parasitic agents, even where the carcass presents no macroscopically visible lesions.

The spatial and longitudinal analysis of wildlife mortality statistics and the results of the associated systematic screening provide a reliable source for analysing health risks posed by/to wildlife. In order to determine the mortality of a certain disease, carcases of animals that died in the field must be discovered. However, for many different reasons it can be difficult to find and count sick and dead wild animals. Stutzenbacher et al. (1986) reported that only 6% of marked duck carcasses were detected by searchers in a Texas marsh. Similarly, less than 27% of the deer carcasses present on an area in Montana after a disease outbreak were detected by hunters (Swenson, 1979). A way to improve the ability to find sick and/or dead animals is to use trained dogs. In one study, dogs found 92% of passerine bird carcasses placed on plots, compared with 45% for human searchers (Homan et al., 2001).

Passive Surveillance

Disease and mortality in wildlife can appear in many different ways, depending on the morbidity and mortality rate. In many programmes worldwide, passive surveillance is mainly based on investigations of dead animals submitted for necropsy and laboratory examinations.

Mass mortality events involving wildlife may often occur unpredictably, and opportunities to investigate such events may be short-lived. Examples of this include recovery of dead marine mammals, fish or seabirds from beaches or coastal waterways, discovery of dead birds or mammals in forests, agricultural or urban areas, and mass mortality events within national parks or nature conservation reserves. More commonly, dead wildlife are submitted as single accessions to animal health laboratories by landowners, hunters or the public. Such passive collection may represent the most frequent opportunity to identify various pathologies in association with disease-causing agents. In isolation, such wildlife accessions may merely represent a disease record to include in the laboratory database. Depending on the accompanying history and the consistency of diagnostic finding, such passively acquired accessions may give insight into the occurrence of important disease processes in wild animal populations. The significance of these accessions may only become apparent over time. To diagnose a wide variety of diseases occurring in wildlife, the necropsy of a wild animal must be carried out by a pathologist with certified specialist training. Also, the examination must be conducted exhaustively in accordance with a standardised procedure, regardless of the size and state of preservation of the carcass (Woodford et al., 2000).

Active Surveillance

For significant diseases in wildlife, an active surveillance programme may be the preferred approach. Such programmes aim to collect a certain number of samples from a target population (either live and/or dead animals) to determine the point prevalence of certain pathogens using antigen or specific antibody techniques. Once an infectious pathogen has been identified, serological surveys supported by accurate species-specific tests are the most commonly used means to actively assess the extent of previous infections within selected free-ranging populations.

Meat Inspection

Abattoir inspection of game meat can be another way to monitor for some important infections, e.g. tuberculosis. The status of this mycobacterial infection can be extremely difficult to monitor purely by field observations of clinical disease. Moose (*Alce alces*) and deer in the Nordic countries and wild boars (*Sus scrofa*) and deer in central and southern Europe are currently the species more frequently examined in this way. National legislation also requires inspection of game meat in many other parts of the world and this procedure should be part of the national monitoring programme for wildlife diseases.

New Diseases

One of the major functions of monitoring programmes is to detect new diseases. For different reasons, detection of new diseases is a difficult task. First, a concept definition of 'new' disease is needed. This would include alterations caused by known disease agents in new (different than known) host species, and also completely new causes of disease, including both single agents and multifactorial causes. Once again, detection probability will depend on disease prevalence, disease transmission and disease-caused mortality, and even on disease relevance. For example, disease agents that are likely to spill over to humans are likely to be more surveyed than others.

Detection of new diseases requires sound knowledge of the current disease status of a given list of host species in a given area, and the systematic investigation of those clinical cases where the aetiology is unclear or possibly new. This in turn is linked with a proper monitoring scheme (including population monitoring, active disease surveillance), and particularly careful passive or scanning surveillance.

After some years, a monitoring programme based on regular investigations of diseases and post-mortem examination of dead animals will achieve basic knowledge about the kinds of diseases occurring within certain geographical regions or in certain animal species and populations (Williams, 2002). Archiving of laboratory cases associated with stored serum and tissue samples is invaluable for retrospective investigations of new or recently emerged diseases. If new diseases occur, they are probably most often discovered through passive programmes based on laboratory accessions and post-mortem examinations. An example of this is European brown hare syndrome (EBHS), which was first observed in European brown hares (*Lepus europaeus*) in Scandinavia in the early 1980s (Gavier-Widén and Mörner, 1991). This disease syndrome was characterised as a primary liver pathology and the etiological agent was assumed to be either an infectious agent, most likely a virus, or a toxic chemical, such as a pesticide. It was not until 1987, when colleagues from Northern Europe met to discuss the epidemiology of EBHS, that it became clear that an infectious agent was most probably causing this syndrome. The etiological agent, a calicivirus, was later described by Lavazza and Vecci (1989) and serological retrospective studies demonstrated that the virus had been present in Europe and other countries since as early as 1971 (Moussa et al., 1992).

There are other instances where significant disease-causing agents have been discovered in free-ranging wildlife as a result of routine submissions of wildlife to animal diagnostic laboratories. In addition, the search for a potential wildlife host may result from the diagnosis of infectious diseases affecting humans, as in the case of Lyssavirus and Nipah viruses (Hooper et al., 1997; Chua et al., 2000; Johara et al., 2001) or domestic animals, as with Hendra and Menangle viruses (McColl et al., 2000).

The Impact of Wildlife Diseases on Wildlife Populations

Many of the preliminary investigations of natural mortality events in wildlife comprised somewhat non-statistical and non-random sampling. They represent a collection of different diseases and causes of death, perhaps associated with some distributional information. Such investigations tend to provide only limited insight into the relevant epizootiology. The reason or motivation for submitting wildlife samples also needs to be considered, as increased effort to recover specimens may follow from increased public awareness. If the public or media perceives a disease outbreak in wildlife as new and important, the public response increases, and with it the number of samples submitted.

To assess the significance of a mortality event caused by a disease in a wild animal population, it may be necessary to attempt to measure the death rate. This can be a difficult task. Recording all dead animals and estimating even the local population at risk may be fraught with error. In the case of some investigations of wildlife diseases, clinically affected or diseased animals have been marked and monitored over time. By their nature, such studies tend to be restricted to defined host ranges and populations and conducted over limited durations. Another way to overcome this problem is to use radio-telemetry and satellite-tracking techniques to monitor the survival or otherwise of tagged animals. This has proved useful in a study on epizootic haemorrhagic disease in deer (Beringer et al., 2000) and in rabies in skunks (*Mephitis mephitis*) (Greenwood et al., 1997).

The ability to prepare for, and respond to, unusual mortality or morbidity events can only be based on previous knowledge and awareness. Access to and awareness of infectious disease must be diagnostic expertise and should be provided to the public and to persons with responsibility for wildlife and environmental stewardship. Sampling during index outbreaks may be minimal, opportunistic and selective, but after the preliminary evaluation of laboratory findings, it is likely that where there is a recurrence, subsequent sampling can be more comprehensive. In remote areas, it is likely that a layperson or a field biologist will make the discovery of an unusual morbidity, mortality or clinical disease event in wildlife.

Where non-specialist personnel are asked to conduct initial investigations, contact with specialists and the relaying of instructions about appropriate sampling and storage of specimens will be necessary. In the long term, the preparation of specific contingency plans and procedure manuals, supported by training, will improve the capability of field biologists and wildlife researchers to respond to such incidents. The development of national wildlife disease networks and training modules for wildlife investigators will also be useful.

Monitoring for the presence of diseases and conducting wildlife health evaluations are normally based on the premise that any pathological or microbiological data collected from individual animals that make up a population will be informative of the host-agent relationship within a given population and environment.

The host-parasite relationship differs widely among different infectious diseases. Some disease agents only infect one or few animal species, such as calicivirus infections of lagomorphs (Lenghaus et al., 2000) or myxomatosis (Woods, 2000). By contrast, viral diseases such as rabies (Rupprecht et al., 2001), bacterial diseases such as anthrax (Gates et al., 2000) and tularaemia (Mörner and Addison, 2000) or parasitic diseases such as sarcoptic mange (Bornstein et al., 2000) are found in a large number of different species.

To assess whether a mortality or morbidity event is due to a disease-causing agent, it is important to collect as much relevant data relating to the incident as possible. Although not all sick or dead animals may be available for investigation, attempts should be made to estimate or count their number and relate those affected or dead to the total population that is potentially exposed or at risk. It is also important to relate the occurrence of a disease to other factors in the environment, which may predispose to the expression of overt disease, and to prepare a time sequence (or timeline).

Conclusions

Surveillance and monitoring programmes are the first steps toward providing an appropriate awareness of the health status of wildlife populations. Justification for developing and maintaining such a capability includes the need for knowledge to: i) manage wildlife populations and their habitats, ii) limit risks related to animal export trade and translocation of animals, and iii) protect natural biodiversity values and safeguard public health. Wildlife disease monitoring programmes integrated within existing national animal health surveillance infrastructures are essential to adequately respond to unusual wildlife mortality events and to efficiently investigate the epizootiology of new diseases found in wildlife.

Emerging Vector-borne Diseases of Public Health in Europe and North America

23

Richard Lampman

University of Illinois at Urbana-Champaign, Champaign, IL, USA

The main objective of this review is to provide a brief introduction to the components of transmission cycles for common vector-borne diseases in the temperate Northern Hemisphere (Table 23.1) and the potential mechanisms regulating vector-pathogen-host interactions (Table 23.2).

Transmission Cycle Components

For vector-borne diseases, the greatest burden is from tropical protozoans, based on either mortality or disability-adjusted life years (Hill et al., 2005). For example, malaria causes 1.1 million deaths/year, leishmaniasis 51,000 deaths/year, trypanosomiasis 48,000 deaths/year, and Chagas disease 14,000 deaths/year. In contrast, bacteria and viruses usually have a lower mortality toll (e.g. *Borrelia*, *Rickettsia* and flaviviruses) (Dobson and Foufopoulos, 2001; Morens et al., 2004).

In the United States, the primary re-emerging and emerging vector-borne diseases are Lyme disease and West Nile virus (WNV), although occasional focal outbreaks of equine encephalitis, California group encephalitis, babesiosis, ehrlichioisis, anaplasmosis and dengue do occur (Gubler, 1998). The clinical cases of Lyme disease outnumber those of WNV, but the number of people at risk to exposure and the case fatality rates are heavily weighted toward the flavivirus (Hayes et al., 2005). The 2002 and 2003 outbreaks of WNV in the United States were each the largest reported epidemic of neuroinvasive

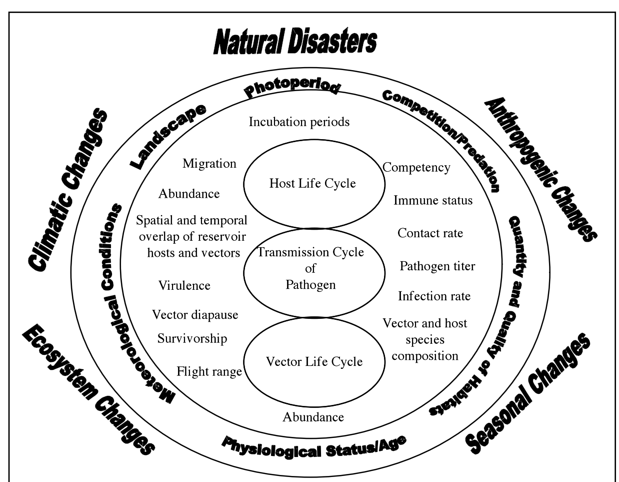

Figure 23.1. Diagrammatic representation of factors influencing the temporal dynamics and intensity of vector-borne pathogen transmission. Source: Author.

Table 23.1. Emerging vector-borne diseases in North America and Europe. Source: Author.

Disease/Pathogen Classification	Vector	Primary Vertebrate Reservoir
Bunyavirus		
LaCrosse, California Group (*Orthobunyavirus*)	*Aedes triseriatus*	Squirrels, chipmunks
Tahyna (*Orthobunyavirus*)	*Aedes cinerus, Aedes vexans*	Hares, rodents, small mammals
Rift Valley fever (*Phlebovirus*)	*Aedes* spp., *Culex* spp.	Cattle, sheep, goats
Flavivirus		
West Nile virus	*Culex pipiens* complex, *Cx. tarsalis, Cx. univittatus, Cx. modestus, Culex* spp.	Birds (corvids with high mortality in USA to WNV99), reservoirs include passerines (robins, cardinals, sparrows)
Japanese encephalitis	*Cx. tritaeniorhnchus*, other *Culex* and *Aedes* spp.	Swine, aquatic birds
Tick-borne encephalitis, Louping Ill, Central European TBE, Russian spring-summer encephalitis	*Ixodes ricinus, I. persulcatus*	Primarily rodents, but also deer, birds, medium-sized mammals
Dengue/DHF	*Aedes aegypti*, some areas *Ae. albopictus*	Human to human transmission may be a sylvatic cycle with non-human primates similar to Yellow Fever
Togavirus		
Eastern equine encephalitis	*Culiseta melanura, Coquillettidia perturbans, Aedes* spp.	Wild birds, passerines associated with cedar swamps
Ross River	*Culex annulirostris* (inland), *Aedes* spp. (coastal)	Kangaroos, macropods, cattle, horses?
Chikungunya virus	*Aedes aegypti, Ae. albopictus*	Monkeys, baboons?
Sindbis virus	*Culex* spp.	Birds, songbirds
Reovirus		
Colorado tick virus	*Dermacentor andersoni*	Ground squirrels, chipmunks, rodents?
Bacteria/Rickettsia-		
Rocky Mountain spotted fever, *Rickettsia rickettsia*	*Dermacentor variabilis, D. andersoni, Amblyomma* spp., *Rhipicephalus* spp.	Rabbits, field mice
Human ehrlichiosis, *Ehrlichia chafeensis, Ehrlichia* spp.	*Amblyomma americanum, Ixodes scapularius*	Rodents, deer
Anaplasmosis, *Anaplasma phagocytophilum*	*Ixodes scapularis* and *I. pacificus*	Domestic and wild ruminants (deer, elk), rodents
Relapsing fever, *Borrelia hermsii*, *Borrelia* spp.	*Ornithodorus hermsi, Ornithodorus* spp	Squirrels, rodents
Lyme borreliosis, *Borrelia burgdorferi*	*Ixodes scapularis* USA, *I. ricinus* in Europe	Mice, rodents (deer not competent)
Protozoa		
Human babesiosis, *Babesia microti* (USA)/*B. divergens* (Europe)	*Ixodes scapularis, Ixodes ricinus*	Rodents cattle, horses, deer, dogs, cats, rodents
Chagas disease, *Trypanosoma cruzi*	Triatominae bugs (conenose or assassin bugs)	Rodents
Malaria, *Plasmodium falciparum, P. vivax, P. ovale* and *P. malariae*	*Anopheles* spp., varies between countries	Humans; great apes or birds?

Table 23.2. Components of vector-borne transmission cycles, biological and environmental determinants, and comments. Source: Author.

Component of Transmission Cycle	Potential Biological and Environmental Drivers	Example or Possible Consequence
Change in composition of pathogens and vectors --Introduction of novel pathogen by infected vector or host --Introduction of competent vector --Range expansion of pathogen and/or vector --Ecological succession	Legal and illegal global travel and trade; movement of livestock, humans, pets and arthropods on planes and ships; hitchhikers in trade goods (e.g. tyres) or containers, tarps, ship ballast, etc. Climate change Meteorological change	Malaria, West Nile virus, Yellow fever virus, Dengue Aedes albopictus, Aedes aegypti, Aedes (Ochlerotatus) japonicus, Anopheles gambiae
Invaders that become vectors of indigenous pathogens	Arthropods on planes and ships; hitchhikers in trade goods (e.g. tyres) or containers, tarps, ship ballast, etc.	Aedes albopictus, Aedes aegypti, Aedes (Ochlerotatus) japonicus, Anopheles gambiae
Abundance of vector and hosts	Appropriate habitats (e.g. aquatic for mosquito larvae); presence of food; nesting, and resting sites; meteorological conditions, ecological changes (landscape preferences), irrigation, breakdown sanitation or control measures (natural and human), water management	Aquatic habitats for mosquitoes, vegetation for questing ticks, animal host availability
Infection rate Contact rate between vector and reservoir hosts	Interaction of reservoir host and vector species; vector feeding pattern; fidelity of vector and hosts to an area	Host preference and host availability; infection rates vary geographically, seasonally and between vector species of mosquitoes; exceedingly high in some areas
Transmission or incidence rates Contact rate between vector and incidental hosts	Human demographics and behaviour; mosquito management; vector feeding pattern	Outdoor activities, personal protection
Feeding pattern on hosts	Seasonal change, host availability, landscape ecology	Shift to mammals in some Culex spp. with host composition change; multiple feeding per cycle; co-feeding infection between infectious and uninfected feeding co-feeding vectors; change with age of vector, landscape ecology (search near larval habitats)
Species composition Vectors and hosts	Climate and meteorological conditions, landscape ecology, photoperiod	Cx. restuans early season, Cx. pipiens, 'dilution effect' with hosts of different competency
Vector developmental times, survivorship, flight/search activity	Temperature, precipitation, windspeed, moonlight and electric light, predators/parasites, host cues (attractants)	Faster development, shorter longevity; greater flight range, greater dispersal; biotic mortality and morbidity factors
Competency (vector and host); pathogen titres	Temperature, animal physiological status, genetics	Increasing temperature may increase titre and shorten incubation periods; lower amount needed in viraemic host to infect vector
Susceptibility; immune status; reservoir and incidental hosts	Movement of hosts causing decline in viraemic animals or increase in susceptibles, acquired immunity, health/ physiological status	Sensitive species such as corvids and equines; 'herd immunity' or change in probability of finding susceptibles, especially as season progresses; first-year hosts may provide naïve source
Spatial and temporal overlap of vector, pathogen and host	Demographics, urbanisation, irrigation, deforestation and reforestation, climate change, weather patterns	Association of humans with vectors based on work, recreation and home exposure
Virulence and resistance	Genetic diversity; exposure to selection agents (antibiotics and chemoprophylactics); changes during animal passage, adaptation to new vectors and hosts	Ratio of Plasmodium falciparum to P. vivax; chloroquine, mefloquine, etc. antimalarial resistance; shift in virulence from one host species to another
Geographical distribution	Climate, landscape, habitats for vectors and host	Temperature ranges and patterns of precipitation; biotic and physical patterns necessary for life cycle
Winter survival/maintenance	Host or vector hibernacula, transstadial transmission, ability to survive seasonal temperature/wet-dry periods	Urban-created hibernacula (stormwater tunnels), leaf debris, animal burrows, vertebrate infection (?)
Vertebrate migration/movement	Local and long distance movement of pathogen, timed to photoperiod, temperature and availability of food	Chronic infections, animals shedding virus in exudates, movement may disrupt or initiate cycles

arboviral disease in the Western Hemisphere and the largest worldwide epidemic of neuroinvasive WNV.

A recent assessment of potential rodent- and arthropod-borne diseases with the greatest risk to humans in Europe included Crimean-Congo haemorrhagic fever, dengue, chikungunya, tick-borne encephalitis, leishmaniasis and hantavirus (Senior, 2008). A search of the Medline database for zoonotic pathogens in Europe found 15 emerging pathogens, of which nine were potentially arthropod-borne (i.e. *Rickettsiae* spp., *Anaplasma phagocytophilum*, *Borrelia burgdorferi, Bartonella spp*., Crimean Congo haemorrhagic fever virus, Toscana virus, tick-borne encephalitis virus group, West Nile virus and Sindbis virus) (Vorou et al., 2007). Southern Europe continues to see a large number of malaria cases, whereas tick-borne borreliosis and encephalitis are more common in wooded areas of northern Europe (Matuschka et al., 1998; Hubálek and Halouzka, 1999; Wielinga et al., 2006; Gratz, 2006; Greer et al., 2008).

The primary arthropod vectors, based on the number of humans and animals at risk worldwide from pathogens they transmit, are mosquitoes (e.g. *Anopheles*, *Culex* and *Aedes*, including *Ochlerotatus*) and hard ticks (e.g. *Hyalomma* and *Haemaphysalis*, *Ixodes*, *Amblyomma*, *Rhipicephalus* and *Dermacentor*) (Gratz, 1999). Approximately 10% of the described tick species in Europe are zoonotic vectors and many, like *Ixodes ricinus*, are infected with more than one pathogen (Jongejan and Uilenberg, 2004). Globally, mosquitoes are the main group of human vectors in terms of cases and mortality, but in North America and Eurasia, tick species historically infect more humans and domestic animals with pathogens (Jensenius et al., 2006). Although the incidence of many tropical vector-borne diseases (e.g. malaria) has been greatly reduced in parts of northern and eastern Europe and North America, re-emergence remains possible because competent vectors (e.g. *Anopheles* spp.) and susceptible hosts (e.g. humans) still exist and international travel and trade virtually guarantee re-introduction through an infected vector or host (Childs et al., 1999; Mellor and Leake, 2000).

Hard ticks (Ixodidae) are an important group of zoonotic vectors (e.g. Lyme disease, various rickettsia diseases and tick-borne encephalitis), although some soft ticks (Argasidae) are also important to humans (e.g. tick-borne relapsing fever). Many of the unique differences in the life history of ticks and mosquitoes, particularly their mobility and longevity, shape the characteristics of transmission dynamics.

Other insect vectors that may be involved with regional resurgences of pathogens include tsetse flies (*Glossina* spp.), sandflies (*Phlebotomus*), black flies (*Simulium*), biting midges (*Culicoides*), lice (*Pediculus humanus*), fleas (*Pulex*, *Xenopsylla*, *Ctenocephalides*, *Tunga*), and true bugs (*Triatoma*). This review focuses on mosquitoes and ticks.

Drivers or Determinants of Outbreaks

The causal factors or drivers of emerging infectious diseases (EIDs) have been widely discussed (Morse, 1994; Gratz, 1999; Sutherst, 2004; Jones et al., 2008) and can be arbitrarily divided into three broad categories: biological (i.e. changes in host susceptibility or immunity, geographical and seasonal variability in competency of vectors and hosts, and inherent variation in the virulence/pathogenicity of etiological agents); anthropogenic (i.e. human activities that modify either the abundance, distribution or probability of interaction of pathogens with vectors or hosts); and abiotic (i.e. perturbations in the environment, particularly those related to climate and meteorological conditions). Outbreaks of vector-borne diseases may be due to any factor or combination of factors that increases the likelihood of a pathogen being transmitted to a human host or a change in the intensity of response to the pathogen (Daszak et al., 2000; Harrus and Baneth, 2005). Global trade and transportation of humans and livestock have been major factors in the distribution of exotic pathogens and vectors. For example, *Aedes albopictus* and *Aedes japonicus* were introduced to Europe and North America via trade in tyres with Asian markets (Lounibos, 2002). The control of malaria and other mosquito-borne diseases in some northern areas by synthetic pesticides such as DDT was often used as justification for shifting funds away from vector biology and vector management expertise, which in some cases facilitated the re-emergence of malaria (Novak and Lampman, 2001; Shiff, 2002; Gubler, 2002). Furthermore, in some stable endemic conditions, changes in vector control may affect the benefits of herd immunity or alter the resistance of human populations (Bodker et al., 2006).

Common Arthropod-borne Diseases in North America

A partial list of arthropod-borne diseases reported from North America and Europe is presented in Table 23.1 with their pathogens, vectors and common reservoir hosts (adapted from Novak and Lampman, 2001; van der Weijden et al., 2007). The ecological similarity in pathogens from the temperate zones of Eastern and Western Hemispheres is well recognised (Childs et al., 1999). For the majority of this review, the discussion centres on mosquito- and tick-borne pathogens (arthropods), the major vectors of pathogens to humans and animals (Kalluri et al., 2007). Most arthropod-borne zoonotic diseases have an obligatory transmission cycle between an arthropod and an animal reservoir (definitive) host. Humans are generally incidental hosts, not important for maintaining transmission; however, for malaria, dengue and yellow fever, humans can serve as a definitive reservoir host during outbreaks.

The major pathogens transmitted by mosquitoes in the US are arboviruses that cause encephalitis (i.e. inflammation of the brain). This includes viruses in three virus families (Togaviridae, Flaviviridae and Bunyaviridae) (Calisher, 1994). Flaviridae are positive-sense, single-stranded RNA viruses that include West Nile virus (WNV), St. Louis encephalitis virus (SLEV), tick-borne encephalitis virus (TBEV) and dengue virus (DENV). The Togaviridae (genus *Alphavirus*) are positive-sense, single-stranded RNA viruses that include Eastern equine (EEEV) and Western equine encephalitis (WEEV) virus. Within the Bunyaviridae, there are three genera associated with vector-borne pathogens of animals (*Phlebovirus*, *Orthobunyavirus* and *Nairovirus)*. The genome of this family consists of three segments of negative-sense RNA, except for the genus *Phlebovirus* (e.g. Rift Valley Fever Virus), which has a segment that is ambisense. The California sero-group viruses (e.g. LaCrosse encephalitis virus, California encephalitis, Snowshoe hare virus) are in the genus *Orthobunyavirus* (=*Bunyavirus) (http://www.ncbi.nlm.nih.gov/ICTVdb/index.htm)*. The sandfly fever group of bunyaviruses (*Phlebovirus*) are transmitted by phlebotomines (sandflies), mosquitoes or ceratopogonids in the genus *Culicoides*. *Phlebovirus* also has a subgroup of viruses transmitted by ticks, which includes tick-borne Crimean-Congo haemorrhagic fever virus. Equine vaccines are available for EEEV, WEEV and WNV, but human vaccines are largely restricted to yellow fever, Japanese encephalitis virus and TBEV, although availability differs by country (Nalca et al., 2003).

It has been argued that Lyme disease and human babesiosis in North America shifted from enzootic rodent-tick cycles to zoonotic transmission because of the rapid increase in white-tailed deer populations and the encroachment of humans and housing into typical tick habitats (Spielman et al., 1993). Lyme disease is the most common vector-borne disease in the United States, although its distribution is primarily in the eastern forest biome (http://www.cdc.gov/ncidod/dvbid/Lyme/). Over 93% of the 64,382 cases between 2003-2005 occurred in the northeastern, mid-Atlantic and north-central states (i.e. Connecticut, Delaware, Maryland, Massachusetts, Minnesota, New Jersey, New York, Pennsylvania, Rhode Island and Wisconsin). These areas also tend to have a high incidence of babesiosis, anaplasmosis and ehrlichiosis (McQuiston et al., 1999; Demma et al., 2005; Chapman et al., 2006). *Ixodes scapularis* (the black-legged tick or deer tick) can transmit *Borrelia burgdorfeii*, *Anaplasma phagocytophila* and *Babesia microti*, and multiple pathogens have been isolated from an individual tick (Magnarelli et al., 1995, Herwalt et al., 2003). In 2005, there were 786 cases of anaplasmosis and 506 cases of ehrlichiosis reported to the CDC.

Common Arthropod-borne Diseases in Europe

The endemic arboviruses transmitted by mosquitoes in Europe include Sindbis virus (*Alphavirus*, Togaviridae), West Nile virus (*Flavivirus*, Flaviviridae), and several California and Bunyamwera sero-group pathogens (*Orthobunyavirus* = *Bunyavirus*, Bunyaviridae) including

Tahyna virus, Snowshoe hare virus, Inkoo virus and Batai virus (http://www.umweltbundesamt.de/gesundheit-e/veranstaltungen/vector-borne-diseases/06_%20Hubalek.pdf; WHO, 2004). Sindbis virus caused outbreaks of fever, rash and arthralgia in northern Europe during 1981-1982, 1988 and 1995 seasons. West Nile virus is widely distributed in southern and central Europe, with relatively few outbreaks until the late 1990s when Romania had 500 cases with a 9% case fatality rate (Hubalek and Halouska, 1999). Tahyna virus is most common in central Europe, Snowshoe hare virus occurs in northern Europe and Batai virus of the Bunyamwera group primarily in central Europe. Many of these viruses do occur over most of Europe and the potential for outbreaks exists. A relatively high seroprevalence among reservoir hosts in Great Britain suggests that enzootic activity may occur for West Nile, Sindbis and Tahyna virus, but transmission to humans may be inhibited by either herd immunity in hosts or the absence of a human biting vector (Gould et al., 2006; Medlock et al., 2007).

Imported viruses detected in Europe include chikungunya (*Alphavirus*, Togarviridae) and dengue and yellow fever (*Flavivirus*, Flaviviridae). These viruses are transmitted by *Aedes aegypti* and *Ae*. *albopictus* in human-mosquito-human cycles during outbreaks. Humans develop viraemia for 2-5 days, which is sufficient to infect the mosquito vector. Chikungunya is typically in Africa, India and Asia, but an outbreak occurred in Western Europe in 2006-2007 (Enserink, 2007). Dengue and yellow fever are tropical diseases that are largely restricted to areas where *Ae*. *aegypti* or *Ae*. *albopictus* are established, which can vary seasonally and may change dramatically with climate change and human transportation of vectors (Monath, 2007).

Malaria in northern and southern Europe is a complicated system of imported, autochthonous and established endemic cases (Jelinek, 2008). In many areas of southern Europe, endemic malaria was essentially eradicated by pesticide use and source reduction of vector habitats as in North America (de Zulueta, 1973; Hay et al., 2004). However, several resurgences have occurred due to shifts in human movements and the public health focus (Martens and Hall, 2000).

Ticks are currently considered the main vectors of human infectious diseases in Europe and North America (Spach et al., 1993; Parola and Raoult, 2001; Brouqui et al., 2004; Vorou et al., 2007). In Europe, these include borrelioses such as Lyme and tick-borne relapsing fever, Mediterranean spotted fever (MSF) due to *Rickettsia conorii*, other spotted fever rickettsioses, human anaplasmosis by *Anaplasma phagocytophilum*, (=human granulocytic ehrlichiosis or HGE) and tick-borne encephalitis (TBE) virus, also known as Central European encephalitis or Russian spring-summer encephalitis. Control of ticks by habitat modification (landscape management), exclusion of vertebrate hosts, host-targeted acaricides and space sprays around homes are effective for reducing exposure, but sylvatic reservoirs remain and ticks readily reinvade (Stafford, 2007).

Tick-borne encephalitis virus (TBEV) is endemic in temperate regions of Europe and Asia and is found in discontinuous woodland areas and along the ecotone of trees and trails in wooded areas and parks. The European subtype is transmitted by *Ixodes ricinus* and the Asian (Far Eastern and Siberian) subtypes by *I. persulcatus*. Infections with the European subtype by *I. ricinus* usually occur in early fall and with the Eastern subtypes by *I. persulcatus* in spring (http://wwwn.cdc.gov/travel/yellowBookCh4-Tickborne.aspx). Although partially due to improved surveillance, TBEV has increased in incidence by over 400% in the past 30 years, with over 10,000 cases each year in Europe (Vorou et al., 2007). Tick-borne encephalitis is the most widespread arbovirus in Europe and cases increased 2- to 30-fold in Central and Eastern European countries about the same time as the end of the former Soviet rule (1992-1993) (Randolph, 2001). For example, TBEV infection rates in *I. ricinus* in central Europe vary depending on geographical location and time of year from less than 0.1% to approximately 5%, while rates of up to 40% have been reported in *I. persulcatus* in Siberia (http://wwwn.cdc.gov/travel/yellowBookCh4-Tickborne.aspx). The virus is increasing in incidence and distribution, extending into western Europe, Scandinavia and across the Russian Federation to the Pacific. Overall, 9,000 to 12,000 TBEV cases are reported annually from the European countries and the Russian Federation (WHO, 2004). Small rodents (e.g. field-mice and voles) are the primary vertebrate hosts, ungulates are amplifiers of adult ticks and migratory birds may disperse infected ticks (Waldenström et al., 2007). In southern regions of

Table 23.3. General factors reported to increase the risk of disease transmission among people, wildlife and domestic animals in fragmented habitats in the Baltic and Great Lakes watersheds. Source: Author.

Human factors	
	Direct agonistic interactions with wildlife, such as removal of 'pests'
	Forest clearing, extractive forestry and encroachment into wildlife habitats
	Utilising water sources located within wildlife home ranges
Wildlife factors	
	Direct encounters with people and domestic animals as a result of home range overlap
	Crop raiding and incursions into human settlements
	Movement across landscapes frequented by livestock
Domestic animal factors	
	Hunting of wildlife (dogs)
	Grazing at the edges of, between or in habitat fragments
	Contamination of physical environment with environmentally persistent pathogens

the Czech Republic, infection rates vary between 0.2 and 1.3% in nymphs and between 5.9 and 11.1% in adult ticks (Danielova et al., 2002). The risk of transmission is related to tick habitat, so it is negligible for individuals who remain in urban or unforested areas, except for a slight chance of infection through unpasteurised dairy products. Although there are no TBEV vaccines available in the United States, inactivated TBEV vaccines are available in Europe.

Lyme borreliosis is the most commonly reported tick-borne infection in Europe and North America. The spirochete bacteria, *B. burgdorferi s.l.*, can be divided into several species, but only three, *Borrelia burgdorferi s.s.*, *B. garinii* and *B. afzelii*, have been associated with clinical cases of disease in Europe (Brouqui et al., 2004). In 1995, the incidence rate (per 100,000 head of population) and annual number of cases were estimated to be low in the United Kingdom (0.3 and 200, respectively), but many orders of magnitude higher in Slovenia (120 and 2,000, respectively) and Austria (130 and 14,000, respectively) (WHO, 2004). Increases in the incidence of borreliosis from 2001-2005 have been noted for Poland, eastern Germany, Slovenia, Bulgaria, Norway, Finland, Belgium, Britain and the Netherlands (Smith and Takkinen, 2006). *Ixodes persulcatus* and *I. ricinus* are the main vectors and *Borrelia* prevalence is typically higher in *I. persulcatus*. In some areas, the incidence of infection in tick populations is extremely high (e.g. 45% of ticks in Croatia tested positive for *B. burgdorferi*). The transmission cycle is similar to that previously described for Lyme borreliosis in North America. Small rodents such as the house mouse are the most common reservoirs of *Borrelia* spp., amplifying infection rates in younger ticks, while larger animals serve as hosts for amplifying the abundance of adult ticks. Ungulates are not reservoir hosts, but may amplify the number of ticks. In Europe, migrating birds may contribute to the spread of tick vectors infected with *Borrelia* or *Ehrlichia* (Olsen et al., 1995; Bjoersdorff et al., 2001). Early detection in humans is crucial, as the earlier a person is treated with antibiotics the less severe the later stages of the disease tend to be.

About 10% of the tick species in Europe are zoonotic vectors, often infected with more than one pathogen. In North America and Eurasia, tick species historically transmit pathogens to more humans and domestic animals than other arthropods (Jongejan and Uilenberg, 2004; Jensenius et al., 2006). However, recent outbreaks of *Culex*-borne West Nile virus in the US and parts of Europe demonstrate the potential rapid dispersal of a mosquito-bird transmission cycle into a naïve host population. Furthermore, the reduced incidence of tropical vector-borne diseases (e.g. malaria) in parts of Europe and North America is seldom due to the eradication of competent vectors (e.g. *Anopheles* spp.) and susceptible hosts (e.g. vertebrates), and thus re-emergence remains a possibility. The more that is known about the components of the transmission cycle and the impact of environmental drivers for a particular vector-borne disease, the better interventions can be designed to target disruption of the cycle.

Conclusions

The greatest burden of vector-borne diseases is in the tropics, but the majority of hotspots for outbreaks tend to be in western Europe, northeastern United States, Japan and southeastern Australia. The major emerging/re-emerging pathogens are viruses, protozoa and bacteria/rickettsia, and about 30% of these are vector-borne. For every emerging or re-emerging pathogen detected, it is likely that even larger numbers of pathogens go undetected because transmission cycles are unable to become established from one season to the next or because the diseases arising remain hidden in foci with little human involvement.

Habitat Fragmentation and Species Barriers

24

Thomas Gillespie
Emory University, Atlanta, GA, USA

Habitat Fragmentation

It is well established that habitat fragmentation reduces overall species diversity and alters species abundance (Laurance and Bierregaard, 1997; Ferraz et al., 2003), often with cascading effects on ecological processes and community structure (Crooks and Soule, 1999; Cordeiro and Howe, 2003). An important aspect of this is how habitat fragmentation alters the way in which hosts and pathogens interact, and how this affects the ability of the host to survive and prosper.

Habitat loss is one of the most important threats to global biodiversity. Modifications to habitat result in both reduction in size and fragmentation. Understanding the ecological importance and conservation value of fragmented landscapes is vital for wise management. Management of small patches of habitat is an opportunity to make important conservation gains, particularly for species with occupancy areas that do not encompass a protected area. For the aforementioned reasons, the study of habitat fragmentation is an active field of inquiry in conservation biology (Laurance and Cochrane, 2001). Studies include investigations quantifying changes in the physical environment (Kapos et al., 1997), experiments ranging from the micro to landscape scale (Debinski and Holt, 2001; Laurance et al., 2002), metapopulation approaches (Hanski and Gilpin, 1997; Lawes et al., 2000), and field studies in fragmented habitats (Laurance and Bierregaard, 1997). These studies have yielded valuable insights into the importance of fragment size, shape and isolation on ecological processes and species survival probabilities. Much of the previous work on fragmented habitats has involved fragments protected from human use (Lovejoy et al., 1986; Tutin et al., 1997; Tutin, 1999). In reality, most fragments are not protected and are characterised by open access to private citizens, who depend on them for fuelwood, medicinals or bushmeat. Thus, fragments change in structure and composition as landowners use the forest for grazing or to extract timber or fuelwood or allow fallow land to regenerate. Although studies in protected reserves have provided us with many insights, they may have biased our perception of the long-term value of fragments.

Emerging infections pose a threat to global human health that is equal to the threat they pose for wildlife conservation. Novel infectious diseases are emerging today in human populations at an accelerated rate worldwide, and the trend shows no signs of abating. Microbes thought to be on the brink of extinction decades ago remain tenaciously endemic, both because of gaps in surveillance and because the pathogens themselves have shown a surprising ability to evolve. Pathogens such as HIV, West Nile virus, SARS coronavirus and influenza virus emerge and re-emerge with disquieting regularity, in some cases causing epidemic or pandemic mortality. Globalisation, climate change and increased contact with reservoir species through agricultural intensification and natural resource exploitation all drive this trend (Daszak et al., 2000; Daszak et al., 2001; Woolhouse and Gowtage-Sequeria, 2005).

Although humans have always shared habitats with wildlife, the dynamics of human-wildlife interactions have changed dramatically in the recent past. Within the last few decades, humans have altered wildlife habitats irrevocably, disturbing ecosystems as the material and economic needs of expanding human populations grow. Today, wildlife lives in habitat mosaics of farmland, human settlements and forest/grassland fragments, and in isolated protected areas such as national parks. Human influences in the form of roads, hunting and climate change are reaching even into the last remaining 'strongholds' of biodiversity. Infectious disease emergence is an unfortunate and unanticipated consequence of these ecological changes.

Species Barriers

Indeed, a full 75% of emerging human infectious diseases are zoonotic or have recent zoonotic origins, with diverse wildlife taxa, livestock and domestic carnivores serving as common sources of infection (Taylor et al., 2001). Comparative epidemiological analyses indicate that an ability to cross any species barriers actually enhances the probability that a pathogen will be classified as 'emerging' (Cleaveland et al., 2001; Taylor et al., 2001; Woolhouse and Gowtage-Sequeria, 2005). This realisation, combined with a sense of urgency about anthropogenic environmental change, has spawned a series of new disciplines bearing such names as 'conservation medicine' or 'ecosystem health', complete with dedicated societies, journals and international meetings (Daszak et al., 2004).

The process by which pathogens cross species barriers and eventually cause persistent health problems involves a complicated series of steps, each with its own (usually low) probability (Wolfe et al., 2007). For example, diseases that find their way into new species do not always possess the ability to spread within that new species, and diseases that can spread within a new species sometimes fail to perpetuate. Nevertheless, the initial 'jump' from one species to another is the critical step, since interrupting the process of transmission between species eliminates the possibility of any 'downstream effects'. Domestic animals can play a critical role in enhancing wildlife-human disease transmission. Dogs, for example, may serve as intermediate hosts for the transmission of blood-borne viruses and parasites to the humans who own them.

The Baltic and Great Lakes areas share a similar propensity for mesopredator release. Crop raiding is another 'risky' behaviour that may increase infectious disease transmission. To raid crops, animals must often cross pastures, dodge chained dogs and packs of roving dogs, and avoid being injured by farmers or their children who guard crops actively. Importantly, results to date indicate that direct contact between species is not necessary for interspecific disease transmission. Indeed, most transmission of gastrointestinal pathogens between people and wildlife is probably indirect and environmental. Pathogens such as *Cryptosporidium*, *Giardia* and *E. coli* readily contaminate water and soil and may persist in wet areas. Human, wildlife and domestic animal contact with common environmental sources of infection may explain many of the trends.

Conclusions

If human behaviour is indeed a strong force influencing the transmission of pathogens between wildlife and people, then targeted interventions should be possible. Making people aware of the disease-related risks of their activities, and providing alternatives, could go far towards reducing interspecies disease transmission and improving human health, animal health and wildlife conservation. Only with a detailed ecological understanding of how human behaviour alters the dynamics of disease transmission among wildlife, people and domestic animals can we design rational intervention strategies that contribute efficiently and effectively to animal and public health and conservation.

Part F

Prevention of Infectious Diseases in Livestock and Wildlife

Authors: Rodney B Baker, Björn Bengtsson, Joanne C. Chee-Sanford, Marja-Liisa Hänninen, Ari Hörman, Ivan J. Krapac, Roderick I. Mackie, James E Pettigrew, Mati Roasto, Martin Wierup and Anthony C. Yannarell

Coordinating Author: Martin Wierup

Principles and Strategies for the Prevention and Control of Infectious Diseases in Livestock and Wildlife

25

Martin Wierup

Swedish University of Agricultural Sciences, Uppsala, Sweden

Introduction

Infectious animal diseases cause major losses to livestock production. A significant number of them can easily be spread to humans e.g. as food-borne infections, and to wildlife. Infectious diseases are also a major cause of poor animal welfare. Within livestock production, it is too late to undertake actions only after clinical signs of disease have developed. Instead, a continuous focus on disease prevention is needed, which also minimises the need for use of antimicrobials and any subsequent risks of problems with antimicrobial resistance. The importance of prevention is emphasised in the new EU Community Animal Health Strategy, which has been named 'Prevention is better than cure' (DG SANCO, 2007). This chapter summarises the elements involved in the prevention and control of infectious diseases and principles and strategies for their use.

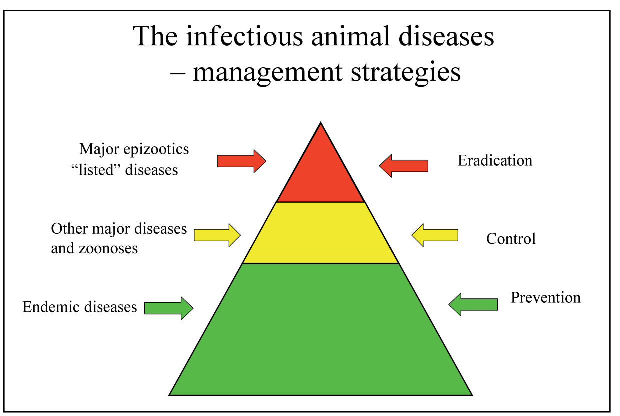

Figure 25.1. Classification and major principles for the control of infectious animal diseases, with the pyramid indicating the number of diseases involved. Source: Wierup.

Categorisation of Diseases

For a structured approach, the wide panorama of existing diseases can be split into three major categories as shown in Figure 25.1. Figure 25.1 also indicates the major principles globally applied for their prevention and control.

The diseases belonging to each of the three categories (Figure 25.1) vary by country and region, but categorisation is traditionally based on the economic importance, prevalence and historical experiences of different diseases. The importance of the three groups of diseases is also reflected in the global and national legislation applied for their prevention and control (Figure 25.2).

International and national legislation and policies focus on the major epizootic diseases and increasingly also

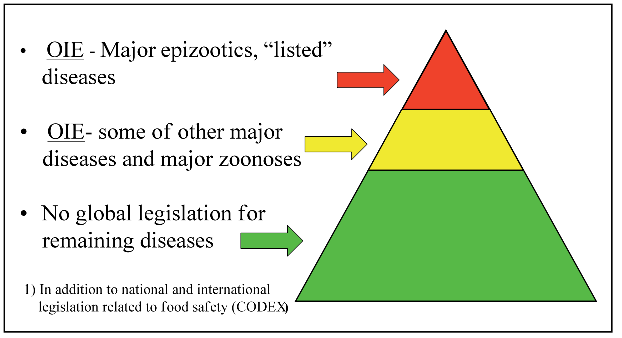

Figure 25.2. Global legislation applied for the control of animal infectious diseases. The pyramid indicates the number of diseases involved and a top-down approach in order of given priority in current global legislation. Source: Wierup.

on the major food-borne zoonotic diseases. According to the legislation, the public compensation for outbreak control and surveillance is generally limited to specific epizootic diseases (e.g. Foot and Mouth Disease (FMD) and Classical Swine Fever (CSF) often referred to as 'listed diseases'. The legislation and categorisation of the diseases is of crucial importance in determining whether there is public or private responsibility for interventions regarding specific diseases, as highlighted in the recent assessment of the EU animal health policy (DG SANCO, 2006). In general, producers are fully responsible for the prevention and control of the endemic diseases, as well as for the economic burden caused by these diseases. A concern for producers is that they are usually also given the responsibility for the control of the food-borne zoonoses (e.g. Salmonella and Campylobacter), despite those diseases seldom causing any significant clinical disorders or economic losses in animals (IAASTD, 2009). It should also be understood that in the developed countries most of the listed diseases have been eradicated or brought under strict control.

In view of the fact that there is no difference in principle between the methods available for the control of the three categories of diseases (Figure 25.1), experiences gained from the control of the listed diseases can successfully be applied e.g. to the endemic diseases and vice versa. In many developed countries a number of the endemic diseases have also been successfully eradicated or controlled by applying methods used for the listed diseases. Such programmes have been found to be very cost-effective (e.g. Valle et al., 2005). The increasing focus on animal welfare and antibiotic resistance further emphasises the importance of control of the endemic diseases (Angulo et al., 2004; OIE, 2005). The global burden of the whole panorama of infectious animal diseases, including the public health cost of human infections, is dominated by the endemic diseases, further emphasising their importance (Figure 25.3).

Basic Requirements for Prevention and Control

Correct Diagnosis and Appropriate Scientific Skill

All actions should be based on correct diagnosis. Basic clinical training and skill as well as access to qualified diagnostic laboratory services are therefore essential. In addition, the outline of a suitable strategy for disease-preventing health control and its implementation is specialist work that requires deep insights into veterinary pathology and microbiology and an understanding of the epidemiology and pathogenesis of the disease and of the situation associated socio- economic situation. These qualifications can be summarised under the term "epizootiology" or with "preventive veterinary medicine" which more includes also possible public health involvement.

Recording of Disease Occurrence

Control and prevention of animal diseases also require insights into disease occurrence. A system for the monitoring and surveillance of disease occurrence should therefore be established (Doherr and Audigé, 2001; Stärk et al., 2006). In health control at herd level, disease-recording systems can be based on observational results such as the recording of lesions at slaughter or production data such as weight gain, pregnancy and farrowing rate. Such data, despite often being of a non-aetiological type, are valuable tools as a basis for further evaluation of possible involvement of specific infectious diseases.

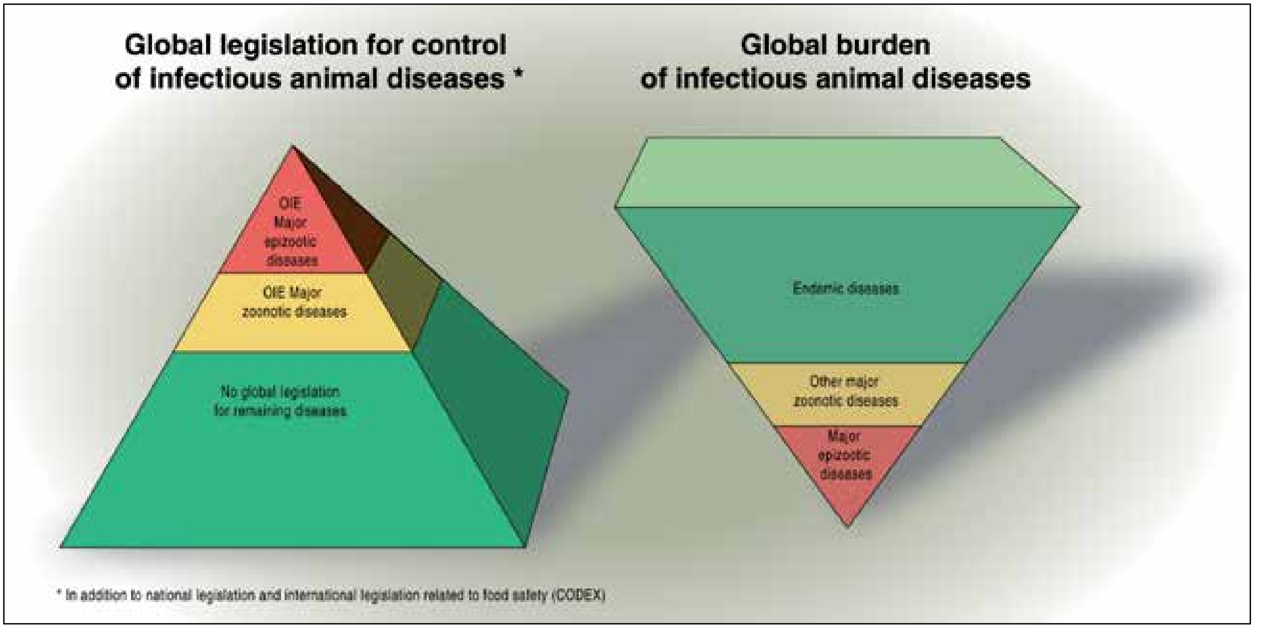

Figure 25.3. Estimated global burden of animal infectious diseases indicating the relative magnitude of the burden (losses of production and labour and costs of control and medical care). Source: IAASTD 2009, http://www.agassessment.org/docs/10505 _HumanHealth.pdf.

Concept for Prevention

Figure 25.4 shows the factors influencing the establishment of infections, which are used here for an analysis of available concepts for disease prevention and control.

Microbial Exposure

Prevention of infections can simply be achieved by protecting the target animal from exposure to infectious doses of the pathogenic microbe.

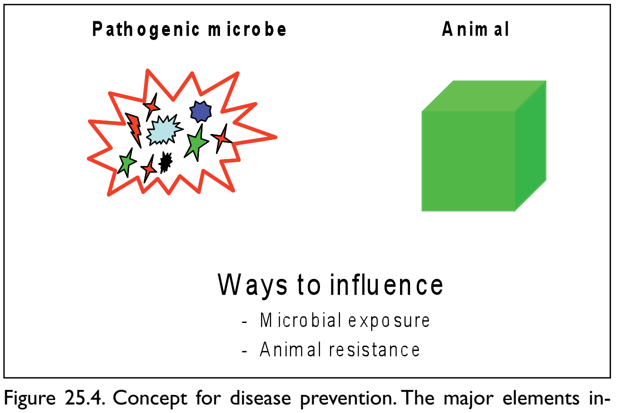

Figure 25.4. Concept for disease prevention. The major elements involved in the prevention of infectious diseases. Source: Wierup, 2008.

1. Total Exclusion from Exposure – Eradication

The most extreme and safest way to prevent an infectious disease is by eradication of the pathogenic microbe. This is usually applied for epizootic diseases in a country or on regional level. The requirement for an eradication procedure is that the epidemiology of the infection concerned is known. An eradication procedure has to be based upon good monitoring and surveillance systems and associated reliable diagnostic techniques and capacity that allow the identification of infected animals or holdings. Access to management and technical infrastructure is necessary to stop the spread of the infection and eliminating its source is essential. Of particular importance is consideration of the requirements for preventing re-introduction of the infection once eradication status has been achieved.

The diseases to be covered by an eradication policy are specified on both national and international levels on the basis of their economic importance and degree of contagiousness (Figures 25.1 and 26.2). Examples are FMD, CSF and Newcastle disease. The control and eradication procedures for this group of diseases have been worked out based upon the epidemiology and the properties of the causal agent for each of the diseases involved. The methods are today well-known both for outbreak control and for the eradication process, as well as prevention of re-introduction of the agent into an area that has achieved disease-free status. The importance of alertness is exemplified in the EU, where all member states must have detailed contingency plans in place against the epizootic diseases.

Scientifically, it is today possible and economically justifiable to control several non-listed diseases with the aim of eradication, even though they are currently not covered by legislation and associated reimbursement of associated costs. Examples of such diseases are Aujeszky's disease in swine and Infectious Bovine Rhinotracheitis (IBR) in cattle, and also diseases with a more complicated epidemiology such as Bovine Virus Diarrhoea (BVD) in cattle. When monitoring demonstrates that herds or even regions or countries are free from such diseases, it is consequently justifiable to ensure that this status is maintained.

It must be noted that measures aiming at controlling and eradicating diseases are in a challenging conflict with the concept of free trade. Demands, e.g. pre-movement testing to prevent the spread of infections, are often considered as trade barriers, which therefore are continually being discussed e.g. within the EU and at OIE, WTO and CODEX on the international level. If these problems are not appropriately addressed, free international trade risks resulting in a situation where borders are unconditionally closed for certain trade or ambitions to improve animal health are discouraged. We may then end up in a situation where the lowest disease status of a participating country will be considered as the standard.

If eradication status cannot be achieved on a country level, it can perhaps be achieved on a regional or herd basis. As part of an organised health control scheme or quality control programme, individual herds and specified types of herds can achieve and maintain disease- free status for certain diseases. This is usually officially regulated for herds of special importance such as boar and bull stations, as well as hatcheries in poultry production. The reason is that infections in those holdings would easily result in a dramatic and fast spread of infections to a large number of herds.

SPF production in swine is a special form of production where total freedom from certain pathogens exists. The status is primarily obtained in animals delivered by caesarean section. These animals are then bred and housed in ordinary farms but under strict biosecurity. The system operates on a large-scale basis in e.g. Denmark and to a limited extent in other countries. The economic significance of infections excluded by this model is well exemplified by the growth rate in such pigs. When raised under SPF conditions, the slaughter weight of 110 kg is reached within approximately 150 days, compared with at least 180 days for conventionally reared pigs (Wallgren, 1994).

2. Partial Exclusion from Exposure – Prevention

This method should always be applied in animal production, at least for endemic infections. The concept is to minimise the microbial exposure to a level below the infective dose, or decrease it to such an extent that immunity is induced in exposed animals but no clinical disease develops. This allows possible further spread of the infectious agent from the primarily exposed animals to be limited to such an extent that secondary clinical outbreaks in contact animals are prevented. Methods empirically and/ or scientifically found to promote this concept are:

Optimising Hygiene

The application of basic concepts of hygiene, which can be defined as measures taken to prevent sources of pathogens building up and methods applied for preventing their exposure to possible target animals. In animal production, this should primarily focus on preventing exposure to the manure. For example, Holmgren and Lundeheim (1994) found that piglets separated from their manure by e.g. cleaning, a drainable or slatted floor, or through well-handled straw bedding, were less susceptible to infections, with subsequent less need for antibiotic therapy. This is typically also found for poultry production.

Isolating Sick Animals

The isolation of sick animals is a basic measure for limiting the exposure of excreted pathogens to neighbouring animals in a herd. Sick animals excrete large amounts of infectious doses. For example, a Salmonella Dublin-infected calf is reported to excrete up to 100 (LD 25) infectious doses per gram of faeces (Wierup, 1983). It should then easily be understood that the isolation of such an animal may well be the threshold that prevents an isolated outbreak of salmonellosis in one or a few animals ending up in an enzootic spread of the infection within a herd.

Avoiding Introducing Sick Animals into a Herd

For the same reason as sick animals should be isolated, they should not be traded or introduced into other herds. Transport itself is known to be a factor that can trigger the exacerbation of a subclinical infection to fulminant status and associated excretion of the pathogen (Isacason et al., 1997). A simple recommendation is therefore to isolate new animals for at least an incubation period before they are introduced into a herd. This allows clinical observation and, when required, testing for specific infections. A general recommendation is that animals should be introduced into a herd only if they come from herds of the same or higher health status, usually verified through being linked to specified health controls and associated biosecurity measures.

Replacing Live Breeding Animals by Semen and Embryos

The genetic status of a herd has traditionally been improved through importation of live breeding animals. In most countries this method has historically led to the introduction of a number of economically devastating diseases. In Sweden, the importation of Friesian bulls in the 17th century resulted in the introduction of e.g. bovine tuberculosis, which subsequently reached a mean national prevalence of 30% of slaughtered animals when diagnosed only by meat inspection, before eradication efforts started. Even if the risk for introduction of epizootic diseases has decreased as a result of international cooperation, this is not the case for the endemic diseases. However, this risk is considerably decreased when the importation of live animals is replaced by semen and embryos.

Using Antimicrobials

The use of antimicrobials is usually considered the method of choice to combat and decrease the number of pathogens. However, the associated risks and problems due to antibiotic resistance have highlighted the need for a change in attitude. In different countries, official or industry-based guidelines have therefore been worked out for such a use in order to contain antibiotic resistance. It cannot be overemphasised that the use of antimicrobials should be based upon a diagnosis following a clinical, and when relevant also a bacteriological, examination. Even if this demand in the real-life situation cannot be fulfilled before therapy is started in all individual cases, different forms of disease monitoring should be performed to limit the use of antimicrobials to cases and situations when bacterial infection is the problem.

The antibiotic resistance pattern should also be checked regularly using relevant methods. It is necessary to determine that the bacteria causing the infection is sensitive to the antibiotic to be used. The dose, way of administration and duration of the therapy should be according to recommendations formulated by the manufacturer, as approved by the relevant authority. As a final step, it is important to follow up and document the clinical results of the therapy, in fact a documentation of the real-life situation.

Although not implemented in all countries, a generally accepted recommendation is that antimicrobials should only be used following a veterinary prescription. In addition, strategies and rules should be established to minimise the risk of a positive economic incentive for veterinary surgeons to prescribe antimicrobials.

The use of antimicrobials for growth promotion should be terminated, as discussed in chapter 29. The use of antimicrobials should also be monitored. This is necessary for an evaluation of the antibiotic resistance pattern in relation to the use of antimicrobials and for the formulation of associated recommendations on their use. In the absence of such data, it is not possible to verify compliance with given recommendations and regulations. Efforts should also focus on the education of producers/farmers in the proper use of antimicrobials.

Host Resistance

General Resistance

To maintain the optimal physiological resistance of a healthy animal, a series of factors have to be fulfilled. The nutritional needs of the animal have to be satisfied. Deficiencies or excesses in proteins, vitamins and trace elements, as well as imbalanced feed composition, can easily result in outbreaks of disease and a decreased capacity of the immune system.

Animal rearing and housing are also essential and ventilation and temperature are well-known factors of importance in this respect. Correction of the ventilation often considerably improves the health status in relation to respiratory diseases in e.g. fattening pigs. When antimicrobial growth promoters (AGP) were banned in Sweden, it was found that the ventilation in the broiler houses often was under-dimensioned, resulting in clinical problems with e.g. necrotising enteritis during periods with hot outside temperatures (Wierup, 1999). Temperature is another factor, and e.g. in temperate climates additional heating prevents diarrhoea both in new-born piglets and weaners, and is also a necessity for newly hatched chicks (Wathes et al., 1989).

Management routines that allow normal behaviour and wellbeing are also essential. A basic requirement is to avoid different forms of stress. This is exemplified in pig production, where the loose sow system in wrongly designed pens in relation to feeding and the social grouping of the sows may easily lead to fighting, with severe secondary injuries and under-nutrition in low-ranked animals (Edwards, 1992).

Specific Immunity

Through the use of vaccines, the immune system can be used to control specific infections and many infections can be effectively prevented in this way. The clostridial infections in ruminants are a typical example and another is vaccination against piglet diarrhoea, which has practically replaced the previous substantial use of antibiotics to control this disease (Söderlind et al., 1982). A more recent example is vaccination against vibrio infections in fish farming (Markestad and Grave, 1997).

Control and eradication of immunodeficient diseases such as BVD, Bovine leucosis in bovines, CAE in goats, Maedi Visna in sheep and Infectious Bursal Disease in poultry, as well as Aujeszky's disease in pigs, have significant health-supporting effects beyond the direct losses due to the absence of clinical disease caused by those diseases.

The disease preventive effect of maternal immunity reflects the infections that the mother has experienced. Older animals therefore generally provide better maternal immunity to their offspring than younger. This was observed at an early stage e.g. in Denmark, where infectious diseases were more frequently found in piglets from gilts and young sows compared with piglets from older sows (Nielsen et al., 1976). The age profile of a herd is therefore of importance, and too high a recruitment rate may have a negative effect on the health situation.

Combinations

The principles described above can be used as single actions. However, as a rule the production systems of animal husbandry usually combine several principles in order to optimise the disease-preventing effect. In the following, this is exemplified mainly from pig production.

The all-in, all-out concept prevents the spread of infections between consecutive groups of animals raised in the same unit. This is practised in the production of most animal species raised for meat production, such as fattening pigs and broilers. A typical violation of that strategy is to keep slow-growing animals in a batch in the same unit until they reach appropriate slaughter weight and simultaneously bring in a new batch of young animals. The latter then easily become infected by those pathogens that are usually the cause of the growth retardation in the slow-growing animals.

The all-in, all-out production system also facilitates cleaning and disinfection between batches, which is a further step to minimise the spread of pathogens from older growing animals to new and younger ones. The all-in, all-out system, with careful cleaning and disinfection, has long been essential in broiler production, aiming at control of salmonella and campylobacter (Berndtson, 1996; Wierup and Wegener, 2006), and such systems are now generally the standard routine in beef and swine production too. Violation of these procedures empirically often results in break-downs of the health status.

A further development of this concept is age-segregated batch production, in which is an all-in, all-out system with groups of animals of the same age. The groups can be kept separate on the same farm, on-site, or at a separate holding off-site. The distance between holdings is of special importance for the spread by wind of respiratory infections. The distance from an SPF swine herd to other holdings of swine is therefore recommended to be at least 0.5-1.0 km.

In pig production, sows are now frequently managed so that they all farrow within a few days. The piglets from these sows are kept together and separated from other animals until slaughter, which also includes the ambition to keep each litter together. In Sweden this type of production was found to be of basic importance to overcome clinical problems when the use of AGP was banned in 1986 (Holmgren and Lundeheim, 1994; Wierup, 2001).

Sow pool production is a special form of multi-site production in which sows are mated and housed at one production site, a pool, and then transported in batches for farrowing on separate farms (satellites). At those satellites the piglets are reared until slaughter, while the sows are transported back for breeding at the pool after weaning of their piglets (Holmgren and Gerth-Löfstedt, 1992).

The importance of the above systems was reflected early in the growth rate obtained. The daily weight gain in slaughter pigs (30 kg to slaughter) in SPF production is often > 1 kg, compared with 0.85 kg in corresponding conventional continuous production. However, in the satellites of sow pools, a 1 kg daily weight gain can often be obtained (Wallgren, 1994). The economic significance of these differences is obvious.

The significance of the measures exemplified above can be better understood when considering that most infectious diseases, although caused by one specific microbe, usually primarily have a multifactorial course. Thus all factors decreasing the risk of an infection becoming established will contribute to improving the health situation in individual animals and on a herd basis.

It is also interesting to note that the probability of infection becoming established in susceptible animals when exposed to infected animals is usually far less than 1. In relation to Aujeszky's disease, it was demonstrated in a swine herd that in spite of frequent direct contacts with ADV-infected pigs for up to 1 year, no spread of infection occurred in exposed non ADV-infected animals (Engel et al., 1995). In Denmark, it was also found that the spread of PRRS infection into herds by semen from PRRS-infected boars was recorded only in 7 out of 700 herds tested (Mortensen, 1998).

The control of salmonella in animal production in Sweden constitutes a good example of how the long-term and consistent application of the preventative measures described above has reduced the prevalence of this infection to a negligible level (Wierup, 2008).

Disease Prevention in Wildlife

The control of infectious diseases in wildlife involves substantial challenges compared with their control in domestic animals. However, when possible, fruitful efforts have been made primarily to protect human health against zoonoses in wildlife (e.g. rabies) or to prevent diseases in the wildlife from being transmitted to food-producing animals (e.g. classical swine fever and *Brucella suis*). A third aim is to protect wildlife from certain destructive diseases, e.g. bovine tuberculosis, that can threaten the existence of certain wild animal species in animal parks and zoos (e.g. Michel et al., 2003). This presentation is limited to the control of two different types of diseases (rabies and CSF) as examples of the challenges involved. Other diseases in wildlife that pose a significant threat to the livestock production are bovine tuberculosis (bTB), where e.g. infections in badgers have been found to spread to cattle, which can prevent the eradication of bTB from the cattle population in England and Ireland. The currently increasing wild boar population also poses a risk of reintroducing Brucella suis to the domestic swine production (EFSA, 2009:2). The vector-borne diseases, which are of increasing importance in relation to climate change, are dealt with in chapter 24.

Rabies

Starting 1940s, an epizootic of fox rabies spread westwards from Poland, with a 20-60 km advance per year, resulting in the infection of several European countries. The most westerly point of spread in France was reached in 1982, and a peak of more than 4,200 rabid animals was recorded in 1989. Eighty-three percent of the reported cases were in red foxes (*Vulpes vulpes*), which are the main reservoir as well as the main vector of the virus (Aubert, 1995).

Oral vaccination campaigns have resulted in a drastic decrease in the incidence of rabies in most western European countries. In Finland, which in 1988 experienced an outbreak of rabies in raccoon dogs and foxes, field vaccination using two bait-layings a year was successfully applied, and since 1991 a single bait-laying each year in autumn has sufficed (Finnegan et al., 2002).

The vaccination has to be performed in organised campaigns and the strategy, including the interval between vaccinations and the duration of the campaign, need to be modified to the regional situation and the vectors involved. In Germany, vaccination has resulted in a drastic drop in rabies incidence but severe set-backs have occurred in some areas. There is always a risk of re-infection from infected surrounding areas when vaccination is stopped and the creation of an immunological barrier to such areas is usually required.

Successful vaccination campaigns have also been performed in Canada and in the USA. Beginning in the 1990s, coordinated oral vaccination campaigns were implemented in Texas to contain and eliminate variants of rabies virus in grey fox (*Urocyon cinereoargenteus*) and coyote (*Canis latrans*), and in several eastern US States with the goal of preventing the spread of rabies in raccoon (*Procyon lotor*). The primary components of the control strategy include: enhanced rabies surveillance, coordinated vaccination, use of natural barriers, and contingency actions to treat emerging foci.

The National Rabies Management Program, which is a cooperative programme that began in 1997, has progressively grown to meet rabies control needs and currently includes oral vaccination in 16 eastern states and Texas and Arizona. Approximately 11 million baits are distributed annually over about 200,000 km^2 in strategic locations to contain and eliminate variants of the rabies virus in coyotes, grey foxes and raccoons. However, the existence of different vector animals is challenging. Skunks appear to help maintain the raccoon variant and serve to re-infect areas (Guerra et al., 2003), potentially confounding the ability to achieve long-term rabies management goals with currently available tools (Slate et al., 2005).

On the other hand, the control of rabies in bats poses other challenges which so far cannot be met by organised preventative actions. A limiting factor is that many bat species are protected. In Europe there is a cycle of insectivorous bat rabies (European bat lyssaviruses; EBLs) which is independent from the epidemiological rabies cycle that involves foxes and other terrestrial mammals.

Classical Swine Fever (CSF)

CSF has caused major outbreaks in the EU during recent decades and a major threat of new epidemic exists due to the fact that CSF virus is present in the wild boar population. The infection is spread by direct contact between wild boar and pigs and indirectly mainly due to the release of contaminated meat products in the environment (Artois et al., 2002).

Wild boar populate most European forests, even in wetlands or mountainous areas (Acevedo et al., 2006). This potential threat is increasing, since the size and range of European populations has critically increased over the past 30 years, possibly due to changes in the hunting practices, the expansion of single-crop farming and climate warming.

During 2003-2007, CSF was reported in Germany, France, Luxembourg, Belgium, Slovakia, Romania, Bulgaria and many other European countries such as the Balkan states and Russia (EFSA, 2009).

In order to prevent outbreaks of CSF in domestic animal production, measures are being undertaken to reduce the contacts between pigs and wild boar, e.g. through the education of hunters and farmers regarding swill feeding and evisceration in forests and the use of electric fences for open-air farming that will prevent physical contact between wild and domestic animals. In addition, significant attempts are being undertaken to stop the natural spread of the disease among wild populations.

Theoretically, eradication in the wild boar population can be achieved by decreasing the number of susceptible individuals in the population to under a threshold level that decreases the probability of the virus surviving. However, in the absence of tools for exhaustive culling and the vaccination of a well-controlled population, as in the domestic situation, the eradication of CSF in wild populations is a complicated issue. The available and applicable current tools are hunting and vaccination. Hunting is performed by amateurs and may negatively influence the population dynamics and, like vaccination, cannot be exhaustive or homogeneous. As it is now known that the spread of CSF in the wild boar population in practice cannot be stopped by hunting, substantial efforts have been made in the EU to control it by the use of vaccine (EFSA, 2009).

Oral vaccination of feral pigs has been used by several EU member states to control the disease. The vaccine used is attenuated C-strain in liquid form (Chenut et al., 1999) and is incorporated into strong-smelling baits that are attractive to wild boar (Kaden et al., 2000). Baits are distributed either by hand at feeding places or by aeroplane. Field trials in Germany and France have confirmed the positive effect of vaccination in controlling CSF outbreaks in wild boar populations (Kaden et al., 2005).

Besides vaccination in infected areas, immunisation has been carried out in a zone surrounding that area. The concept of this so-called 'cordon sanitaire' is to build up a vaccination barrier to a non-infected area to stop the further spread of the disease to unaffected territories.

The main limitation of oral vaccination in wild boar relates to bait consumption in the youngest age classes and in practice the direct impact of oral vaccination is restricted to animals older than 3 months. However, due to the transfer of colostral immunity, vaccination of older sows has an indirect effect on the immune status of the offspring.

The main effect of vaccination is to maintain a high level of herd immunity, which prevents the spread of the virus. Thus vaccination of an infected population can be considered a valuable tool to control and possibly eradicate the infection from an area (e.g. Von Rüden, 2008).

Conclusions

The control of infectious diseases in livestock involves numerous strategies, of which the use of antibiotics is one. The intelligent use of the wide panorama of available disease preventive measures, as exemplified above, can easily contribute to a good health situation and improve the economics of animal production without the use of antibiotics except for treatment of sick animals. On the other hand, when a bacterial infection is established, antibiotic therapy is an effective first method of choice and should be used to prevent suffering in sick animals. Antibiotics are probably the most valuable drugs in animal production and the use of antibiotics should thus be considered as an integral and usually final part of the disease prevention strategy. They should be used when all other measures have failed, and not as a replacement for them. A production system that requires the routine use of antibiotics for all animals is not sustainable and cannot be recommended due to reasons further described in the subsequent chapter 29. The control of infectious diseases in wildlife involves substantial challenges. However, fruitful efforts have been made primarily to protect human health against zoonoses in wildlife (e.g. rabies) or to prevent diseases in the wildlife from being transmitted to food-producing animals (e.g. classical swine fever and *Brucella suis*). A third aim is to protect wildlife from certain destructive diseases, e.g. bovine tuberculosis, that can threaten the existence of certain wild animal species in animal parks and zoos.

Antimicrobial Resistance
a Food Safety Perspective

26

Mati Roasto and Ari Hörman
Estonian University of Life Sciences, Tartu, Estonia
Marja-Liisa Hänninen
University of Helsinki, Finland

Many food-borne pathogens and opportunistic bacteria such as enterococci can have natural habitats in food animals. They can contaminate meat and milk products during slaughter and milking or contaminate raw vegetables when the soil is fertilised with animal manure. The molecular analysis of antibiotic resistance genes, plasmids and transposons has demonstrated that identical elements are found in animals and humans. The phenomenon applies to pathogenic, opportunistic and commensal bacteria: the use of antibiotics in veterinary medicine, similar to its use in agriculture and aquaculture, selects resistant bacteria, which are released into the environment. Specific food items, water and direct contact can spread these bacteria from animal microfloras to human microfloras (Teuber, 2004).

The development of antimicrobial resistance in pathogenic bacteria is a matter of increasing concern. There is growing scientific evidence that the use of antimicrobials in food animals leads to the development of resistant pathogenic bacteria that can reach humans through the food chain (Van Looveren et al., 2001). These human food safety concerns have been influential in prompting the European Union to ban the use of antimicrobials as growth promoters in food production and to increase their surveillance for bacterial resistance in food-borne pathogens and indicator organisms (Smith et al., 2007). In farm environments, commensal and environmental bacteria may be a reservoir for the transfer of antimicrobial resistance genes to pathogenic bacteria. Bacteria acquire most resistance genes through horizontal transfer, while conjugative genetic elements such as plasmids and transposons are common vectors for the dissemination of antimicrobial resistance genes to diverse microorganisms (Smith et al., 2007). Many scientists and public health specialists expect this resistance problem to worsen unless we act decisively.

Estonian antimicrobial susceptibility studies (Roasto et al., 2007) of *Campylobacter* strains revealed high resistance patterns for several antimicrobials. A high proportion of multidrug-resistant isolates (27.5%) was found. All of these isolates were resistant to enrofloxacin and all except one were resistant to nalidixic acid. Hakanen et al. (2003) noted that 20% of the human isolates associated with travelling were resistant to three or more antimicrobials. Multiresistant isolates reported in the Roasto et al. (2007) study consisted of a combination of all tested antimicrobials. The results showed that multidrug resistance was significantly associated with enrofloxacin and nalidixic acid resistance (correlation coefficient 0.372 and 0.310, $P<0.01$). These findings suggest that the use of fluoroquinolones may select multiresistant strains, since resistance to erythromycin, gentamicin or oxytetracycline was rare without simultaneous resistance to fluoroquinolones. A recent study on antimicrobial resistance of *Escherichia coli* at a farm where no antimicrobial treatment of the birds was performed during one year before

Table 26.1. Antimicrobial susceptibility of *C. jejuni* isolates (n = 131) from broiler chickens in Estonia, 2005-2006.

Antimicrobial agent[a]	Antimicrobial concentration range (μg/ml) VetMIC™Camp	Breakpoint (μg/ml)	No. of resistant strains (%)
Am	0.5-64	32	10 (7.6)
Ef	0.03-4	1	96 (73.3)
Em	0.12-16	16	26 (19.8)
Gm	0.25-8	8	25 (19.1)
Nal	1-128	32	99 (75.6)
Tc	0.25-32	4	42 (32.1)

a Antimicrobial agents: Am, Ampicillin; Ef, Enrofloxacin; Em, Erythromycin; Gm, Gentamicin; Nal, Nalidixic acid; Tc, Oxytetracycline.

the sampling showed that the resistance to tetracycline, gentamicin and streptomycin persisted but all isolates were susceptible to enrofloxacin (Smith et al., 2007). Thus multiresistant strains may reflect the past history of antimicrobial usage during a longer period. This phenomenon may partly explain the rather high number of multiresistant strains in the Estonian study (Roasto et al., 2007). Antimicrobial susceptibility patterns of *C. jejuni* isolates from broiler chickens in Estonia in 2005-2006 are shown in Table 26.1.

In addition, the costs of treating antimicrobial-resistant infections place a significant burden on society. For example, it has been estimated that the in-hospital costs of hospital-acquired infections caused by just six common kind of resistant bacteria were at least 1.3 billion USD (1 billion euros) in 1992 (http://www.hhs.gov/news/press/2001pres/20010118b.html; accessed 24 May 2007). Multiresistance to antimicrobials (resistance to three or more unrelated antimicrobials) is a major public health problem because it limits chemotherapeutic options. The concept of Critically Important Antimicrobials, both for humans and animals, should be used by EU member states for setting priorities in improved management of antimicrobials in animal production.

Health Management with Reduced Use of Antibiotics in Pig Production

27

CASE STUDY
USA

James E. Pettigrew
University of Illinois at Urbana-Champaign, Urbana, IL, USA
Rodney B. Baker
North Carolina State University, Raleigh, NC, USA

The world's food production system has responded magnificently to the need to dramatically increase production during the last few decades, to supply the growing human population of the world. But the challenge continues, driven by both the projected increase in population and the wonderful improvement in living standards and diet quality that will be the fortunate lot of many people in the developing world. The need to continue to increase food production with limited resources places the onus squarely on the livestock industry to increase both efficiency and production.

Protecting the health of animals in future livestock production systems will be paramount to successfully meeting the food needs of the global human populations. Disease elimination and exclusion of economically important agents have proven difficult at best, requiring adoption of a multitude of functional management opportunities. One important component of health maintenance has long been the use of antibiotics. Thus, a high level of efficiency and productivity requires healthy animals. Against this backdrop, other ethical and economic concerns also focus attention and resources on keeping animals healthy. Keeping animals healthy is a difficult challenge in any circumstances, but the difficulty is magnified by limitations on the use of such a powerful health technology as antibiotics.

The convention has been historically to consider three separate roles of antibiotics in animal production: therapeusis, prophylaxis and growth promotion. In practice, the boundary between use for prophylaxis and growth promotion has often been indistinct, so when use of antibiotics for 'growth promotion' is halted, there is often an increase in disease (McEwen et al., 2003). The reasons for reduction of antibiotic use in animal production may be strong and valid, but the implementation of such reduction has dramatic impacts on the maintenance of animal health.

We believe the appropriate reaction to such a change is to re-evaluate the entire health management system. Our focus is on the pig industry, where the health management programme, as we view it, centres on such characteristics of the production system as multi-site production, pig flow and weaning age. Other important components include biosecurity, sanitation, vaccines and dietary factors. We reject the notion that we should simply seek 'alternatives to antibiotics' as too narrow, but there is strong evi-

Table 27.1. An incomplete list of potential dietary technologies to improve pig health and productive performance.

Energy & protein sources	Additives	Feeding management
Spray-dried plasma	Immune egg products	Low-protein diets
Milk protein products	Mannan oligosaccharides	Restricted feeding
Egg products, conventional	Fructo-oligosaccharides	Fermented liquid feed
Fibre/Alternate cereals	Other oligosaccharides	
Lactose	Probiotics	
	Essential oils	
	Acids	
	Zinc oxide	
	Copper sources	
	Yeasts/yeast products	
	Bacteriocins	
	Bacteriophage	
	Enzymes	
	Nucleosides	

dence that some feed ingredients and other characteristics of the feeding system can be important components of a health management programme.

This chapter will focus first on the rich supply of feed-based health technologies now available for use by the industry, and then will consider the practical and successful application of a wide range of health technologies for pigs with reduced antibiotic use.

Feed-based Health Technologies

The group of technologies considered herein is restricted to dietary ingredients that have physiological activity beyond provision of bioavailable nutrients, and to formulation practices and feeding methods that similarly alter physiological conditions. Many of them are suggested to provide benefits through impacts on microbial populations in the digestive tract and/or influence on immunity, although other modes of action also fall within the scope of this discussion. Consideration of microbial populations in the digestive tract draws attention to growth and survival of pathogens, but also includes the potential importance of commensal bacteria.

The industry now has a rich supply of potential dietary technologies available for evaluation and use (Table 27.1). A thorough discussion of all of these potential dietary tools is beyond the scope of this paper, but some of them are discussed in varying depth below.

This discussion relies somewhat on the powerful but imperfect statistical approach of meta-analysis, a combination of experiments into a single statistical analysis. Meta-analysis achieves substantial experimental power through the amalgamation of several experiments, overcoming the fact that most single experiments have too little experimental power to detect some important treatment effects. It also provides an unusually broad inference space because of the range of conditions under which the several experiments are conducted.

The results of a meta-analysis reflect the experiments included, so it is important that those experiments are representative of conducted or potential experiments. A biased sample leads to biased results. That is a special problem when dealing with the published literature because scientists or journals often choose not to publish data that fail to show statistically significant differences among experimental treatments. That regrettable choice results in a bias in the refereed literature, which results in an unavoidable bias in the results of a meta-analysis based on the scientific literature. Note that this bias affects any review of the literature, but is most obvious when the review is formalised and quantitative as in the case of a meta-analysis. It likely overestimates the benefit of dietary factors in some of the cases considered here.

Spray-dried Plasma

Two meta-analyses of the impact of spray-dried plasma on the growth performance of young pigs were reported in 2001 (Coffey and Cromwell, 2001; van Dijk et al., 2001). Each found dramatic responses to plasma, with mean increases in growth rate of 25% and 27%. Such large increases in pig growth rate are rare, and the size of the impact has driven the widespread adoption of spray-dried plasma in diets for weaner pigs in spite of its relatively high cost.

We have recently reviewed data that were not included in the earlier reviews, and found a mean increase in growth rate of 23% (P<0.0001) when plasma was included in the diet, in good agreement with the earlier reviews. We believe the positive bias often found in the literature is unlikely to occur in this case, largely because most of the recent studies have not been conducted to evaluate the effect of plasma, but to evaluate other products as potential 'plasma replacements'. Thus, whether there is a response to plasma may be largely irrelevant to a decision about publication. The recent data indicate that the response to plasma does not diminish with increased weaning age, and does not appear to be strongly related to the protein source the plasma replaces.

There is now strong evidence that spray-dried plasma in the diet provides protection against enteric disease caused by *Escherichia coli* (e.g. Bosi et al., 2001; Owusu-Asiedu et al., 2003). This development has enormous practical importance.

We still do not understand clearly the mechanisms through which dietary spray-dried plasma improves growth performance and protects against disease. However, it is useful to remember that plasma carries many physiological signals throughout the body, so it contains functional components. The most likely mechanisms relate to increased feed intake and to protection against disease.

Protective effects of plasma may derive from its immunoglobulin content (Pierce et al., 2005), from glycoprotein glycans which may block adhesion of pathogens to intestinal binding sites (Nollet et al., 1999), or from immunomodulation. Dietary plasma appears to down-regulate the inflammatory process in healthy pigs (Touchette et al., 2002), which may contribute to increased feed intake and direction of nutrients to productive functions. However, in immune-challenged pigs, dietary plasma stimulates immune function (Touchette et al., 2002), presumably providing protection.

Immune Egg Products

Hens immunised against pig pathogens produce antibodies against those pathogens and deposit them in eggs. Those eggs or their components can then be fed to pigs to provide passive immunity to the diseases in question. Selection of the most appropriate antigen and immunisation schedule appears critical to success. In most experiments published to date, antibodies have been raised to surface antigens of *E. coli*, including the K88 and F18 antigens.

It is now clear that this technology can be enormously effective in reducing the percentage of pigs that die or show clinical signs (e.g. Yokoyama et al., 1997; Marquardt et al., 1999), although failures to provide benefits (e.g. Chernysheva et al., 2003) must also be acknowledged. In some cases, disease was largely controlled by spray-dried plasma and there was no further benefit from immune egg products (Owusu-Asiedu et al., 2003).

Acids

Addition of organic acids to pig diets has been suggested to improve growth performance. The perceived mechanisms include reducing the pH of the stomach contents, which in turn may improve nutrient digestion and change the microbial populations in digesta. A second mechanism is that acids in undissociated form can enter bacterial cells, where they dissociate and damage or kill the cell. Different acids may have different effects.

It is common to feed combinations of acids rather than individual ones, and sometimes the combinations include inorganic as well as the more common organic acids. Unfortunately, most of the published data address single acids.

A review of a substantial body of data (M.T. Che and J.E. Pettigrew, unpublished) shows impressive increases in growth rate of 12% (P<0.001) during the first 2 weeks after weaning, 6% (P<0.001) during the first 4 weeks, 4% (P = 0.01) during the growing phase and 3% (P = 0.02) during finishing when acids are added to the diet. The likely overestimation due to bias in the literature discussed above applies to these numbers to an unknown extent. The response to acids is remarkably robust, with no detectable influence in starting pigs of weaning age, presence of animal proteins in the diet, growth rate or acid inclusion level on the size of response. There also appear to be increases in dry matter digestibility of one percentage unit (P = 0.01) and in protein digestibility of 3 percentage units (P = 0.001).

Lactose

Lactose is an easily digested carbohydrate, but it may also be a prebiotic (a compound that serves as a preferred substrate for certain bacteria and therefore encourages their proliferation in the digestive tract). It has been suggested (Partanen et al., 2001) that dietary lactose may stimulate the growth of organisms such as *Lactobacilli* in the stomach, that the *Lactobacilli* ferment the lactose to lactic acid, and the net effect may be similar to feeding lactic acid. Data from our laboratory (Palacios et al., 2004) refute that suggestion, showing that dietary lactic acid has impacts on conditions in the digestive tract that are not mimicked by lactose.

Lactose improves the growth performance of young pigs (Tokach et al., 1989) and is widely used their diets.

Mannan Oligosaccharide

Products described as mannan oligosaccharides contain mannose, but are more complex than suggested by the term, being preparations of the outer layer of the cell wall of yeast. The mannose is key to one perceived mechanism of action of this product.

Most enteric pathogens must attach to the intestinal wall in order to proliferate and cause disease; more specifically they attach to carbohydrates as the binding sites. Several pathogens, including some *E. coli*, attach to mannose units on the mucosal surface. It is perceived that the yeast cell wall fragment containing a mannose unit in the lumen of the intestine may bind to the pathogens, preventing the pathogens from binding to the intestinal wall. The product must survive the digestive processes and reach the lower intestine in order to function in this manner. Recent results from our laboratory (Miguel et al., 2006) confirm that a mannan oligosaccharide product changes the microbial populations in the digestive tract of young pigs.

There is also growing evidence that mannan oligosaccharide modulates the immune system (Kim et al., 2000; Shashidhara and Devegowda, 2003).

It is clear from a meta-analysis that a mannan oligosaccharide increases growth rate of young weaned pigs by about 4% ($P<0.01$), with the response being larger where pigs grow more slowly (Miguel et al., 2004). This meta-analysis did not rely completely on published data, so the potential bias is reduced.

Fibre

Dietary fibre consists mainly of non-starch polysaccharides, carbohydrates that are not digested by the enzymes produced by animals. Because they escape digestion in the upper digestive tract, they are available for fermentation by the microbes that inhabit the lower gut to support their proliferation. The question of whether dietary fibre is either beneficial (Wenk, 2001) or detrimental (Hopwood et al., 2004) in disease resistance is controversial.

Practical Experiences with Antimicrobial-free (ABF) Pig Production

History and Challenges

Antibiotic-free (ABF) pork, the most extreme form of reduced antibiotic use, found its place in the US grocer's meat case during the early years of the 21 century[1]). Its development was driven largely by perceived marketing opportunities created by consumer interest in ABF pork which, in turn, was promoted by consumer advocacy groups.

Many consumers continue to believe in the health benefits of consuming meat products from animals raised free from antimicrobial (antibiotic) exposure, in spite of a lack of clarity in the scientific literature about the amount of antibiotic resistance in human medicine caused by use of antibiotics in animal production (Phillips et al., 2004; Lusk et al., 2006; Sørum et al., 2006; Zhang et al., 2006; Duriez and Topp, 2007; Gilchrist et al., 2007; Macovei and Zurek, 2007; McMahon et al., 2007).

It was soon discovered that production of ABF pork is more expensive than production of conventional pork[2]). The extra cost of ABF pigs occurs not only at the production level but also within the manufacturing, marketing and advertising efforts of those companies offering ABF products to the consumer. These added costs must be justified by higher value of the 'branded' ABF pork, but to date only a few cuts of pork meat (loins, chops, etc.) have captured the anticipated added value. The burden of the

1) ABF: Following accreditation the production takes place at premium standard farms assuring that marketed animals have never received antibiotics through feed, water, or via injection from birth to slaughter. The animals for these products are segregated during the production, transportation, slaughter and fabrication processes to ensure product integrity. http://www.psfarms.com/process_verified_pork.html. Editorial comment: In contrast to the ABF production in USA, antimicrobials are continuously used for therapy in the EU, regardless of the ban on the use of antimicrobials for growth promotion.

[2]) Comparison made with conventional production in USA where antimicrobials are used for therapy, prophylaxis and growth promotion.

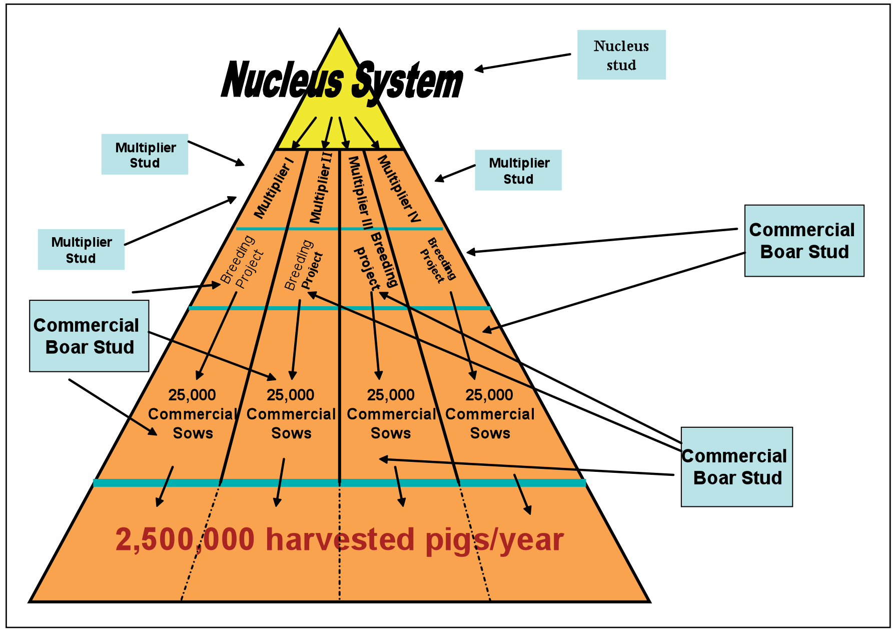

Figure 27.1. An example of a modern 'Production Pyramid' illustrating pig flow and semen inputs. Movement is always down the pyramid and never in an upwards vertical or horizontal direction. Source: Rodney B. Baker.

added cost must be borne by the increased value of only a small proportion of the final product. Most of the pig is never recognised as a branded, value-added product.

Pig well-being/welfare issues are likewise significant because ABF pigs typically suffer from endemic diseases at a greater intensity than conventionally reared pigs in the same production enterprise. Companies producing ABF pigs run the risk of being singled out over welfare issues by watchdog groups.

Thus far, large integrated systems have produced the vast majority of the ABF pigs produced in the United States, contrary to the expectation that ABF production would favour small producers. There are many reasons for this unintended consequence; primarily the inconsistency of supply and quality coming from the small pig farms has not met the needs of the modern food service industry.

Role of Breeding Herd Health

Producing healthy pigs is essential for competitiveness and begins with the health status of the breeding herd. Large operations typically move pigs down a production pyramid (Figure 27.1) beginning with the genetic nucleus where genetic progress is made. Pigs of this generation move to the multiplication level of production, which performs the duty of producing the replacement breeding animals (parent stock) whose offspring become the vast majority of pigs harvested for pork meat. The pyramid is similar for ABF and conventionally reared pigs. The health status of the market pigs is a result of the original health status of the nucleus level stock, as well as the deterioration of health status that occurs as pigs move down the pyramid to the different levels of production (Baker, 2006), ending at the harvest segment. This deterioration is due to lapses in knowledge or implementation of biosecurity at each step of the process, reflecting the presence of multiple weaknesses of exclusion, management and containment. All pig production systems suffer from this deterioration of health and struggle with the development and implementation of functional strategies which exclude or control pathogens. However, these strategies are especially critical when producing ABF pigs.

Production systems of all sizes struggle to maintain animal health. Small farms may claim innate biosecurity advantages but typically lack the veterinary sophistication, expertise, production infrastructure and motivation required to capture this opportunity. Small producers have also failed to meet the supply, uniformity and meat quality needs of the food service industry, a failure that to date has limited their sustainability in the market place.

Role of Growing Pig Health

Most of the significant added costs of ABF production arise from 1) interventions to maintain pig health, and 2) the poor health resulting from the inadequacy of those interventions. Although there are many useful tools for maintaining health, usage of additional immune stimulation products in the growing pig is common. Most efforts are directed at controlling the effects of population endemic bacterial and viral agents. Pathogen exclusion efforts are also costly and are addressed below in the biosecurity section. Numerous vaccinations not routinely used in conventional production are frequently utilised in ABF production with varying results. *Streptococcus suis (S.suis), Haemophilus parasuis (Hps), Actinobacillus suis (A.suis), Actinobacillus pleuropneumoniae (App), Mycoplasma hyopneumoniae (M.hyo), Lawsonia intracellularis (ileitis), Erysipelas, Salmonella, Escherichia coli (E.coli)*, Porcine Reproductive and Respiratory Syndrome virus (PRRS), multivalent swine influenza (SIV), Porcine Circovirus Type 2 (PCV2) and other commercial or autogenous vaccines are frequently administered to a needle-wary pig. Unfortunately, most of these vaccines have variable efficacy and often are of marginal or reduced value in ABF pigs. Many of the vaccines available for use in the US produce little value in high-challenge conditions.

A 'closed herd' where replacement stocks are produced within the farm may have a stabilising health effect for many of the agents plaguing ABF production, but this strategy has not been an effective deterrent against PRRS, the most costly swine disease of US pigs (Neumann et al., 2005) and the most difficult to control. Other somewhat more effective methods of ABF health control include all-in, all-out pig flow by age group (rooms, houses, & sites), small population size, detergent wash and disinfection between groups, continuous sanitation, single sourced pigs, geographical isolation, delayed weaning age, frequent hand washing, and other sound husbandry practices. Overall, ABF production is much like flying an airplane; inherently safe but disastrous results may occur from even minor human errors or natural events.

Freedom from Specific Disease Agents

When pigs are free from *Mycoplasma hyopneumoniae* and PRRS virus, ABF production can compete on a cost of production basis with conventionally raised pigs. Feed conversion may be slightly depressed but growth rates, mortality and morbidity may be comparable to conventional production counterparts that are challenged with these agents. Thus, disease freedom is a logical opportunity to improve the lot of the ABF pig, avoiding excessive losses. Several methods of elimination of PRRS virus and *M. hyo* have evolved since the mid 1990s. Eradication methods are simple and generally cost-effective providing these agents can be excluded for periods longer than 12 months. Successful long-term ABF production requires freedom from PRRS virus, *Mycoplasma hyopneumoniae, Actinobacillus pleuropneumoniae* and as many of the eradicable agents as possible. Unfortunately, exclusionary biosecurity strategies have largely been impotent in geographical areas of the US with heavy pig density.

Biosecurity

Functional biosecurity is especially important in ABF systems. It may be divided into three significant areas of concern in all pig production systems: external biosecurity (bio-exclusion), internal biosecurity (bio-management) and agent containment (bio-containment). All are especially valuable in ABF production due to the lack of antibiotic interventions.

Bio-containment applies when a new agent enters a system or industry. Often a new agent enters a segment of production (i.e. boar stud) only to be transmitted to other sites because functional (effective) biosecurity methods are lacking. As discussed, internal biosecurity measures are those methods that reduce the impact of disease agents already present in the operation. This is accomplished by reducing the dose of the agent, increasing herd immunity to the agent, controlling the timing of infection and reducing environmental and biological stress. External biosecurity is designed to prevent a the introduction of a new agent, and individual agents often require specific interventions. Before designing exclusionary biosecurity strategies, the 'ecology' (complete life cycle) of a specific agent must be reasonably understood. Unfortunately, we often do not have full knowledge of these life cycles as is the case with PRRS virus. One last generality worth considering is that if a farm does not have security against sabotage or other crimes, then functional biosecurity is lacking. Uncontrolled entry of people is often responsible for introduction of new disease agents. ABF pigs are most susceptible to new agents, thus allowing entry of unwanted intruders leaves the population open to new disease introduction.

ABF biosecurity strategies should be developed utilising a Hazard Analysis Critical Control Point (HACCP) approach. Knowledge from scientifically applied field trial methodology, peer-reviewed publications and significant field experience should be heavily relied upon when establishing the critical control points. Extensive interviews and inputs from all farm staff should be included in the early stages of the hazard analysis assessments. Without participation of the farm employees, many critical control points (CCP) will be overlooked. Once the CCPs are identified, then and only then can intervention strategies be developed. Only those evidence-based intervention strategies that have demonstrable usefulness in the field are applicable. A hierarchy of interventions based on relative risk assessment can then be developed, in the end focusing on those factors which have the greatest impact and opportunity for success. A formula which is helpful in ranking appropriate biosecurity implementation decisions is as follows:

$$\text{Biosecurity Intervention Value (BIV)} = \frac{\text{DEV} * \text{RR} - \text{IC}}{\text{DD}}$$

where:
DEV = Disease Exclusion Value per pig per year (typically greater in ABF pigs)
RR = percentage Risk Reduction per year for each intervention
DD = Degree of Difficulty (Ranking 1-10, with 10 = very difficult to implement and maintain)
IC = Intervention Cost per pig per year

This formula facilitates analysis of each agent and each intervention strategy. These computations can then be used for choosing those strategies that have a final BIV greater than zero. Although arbitrary, the DD allows us to consider a customised score for the complexity of an intervention and the ability of the farm staff to adopt, implement and sustain an intervention procedure or process. It becomes farm- or system-specific, which is ideal in practice. If the risk of several diseases can be reduced by the same intervention, then the DEVs can be added together and the sum entered into the equation. As multiple agents are considered, the strength of the intervention strategy becomes apparent. Of course we do not know all the risk factors, values of disease exclusion or the percentage RR, but from the risk assessment tool, published information and biosecurity experts, one can arrive at reasonable approximations. Developing a value equation for each disease is often a matter of benchmarking diseased pigs with those that are disease-free in the same system. Some average disease cost numbers have been published and also useful benchmarks. The amount of RR for each biosecurity intervention and the perceived value for exclusion helps us arrive at a logical expectation for those interventions. With this approach only those interventions that have a value greater than zero are applied.

Calculating IC can be difficult and often relies on farm or industry experience. For example, the cost of building a shower facility is relatively straightforward but the variable costs associated with implementing showers for all who enter the farm are highly variable. Clothing costs, frequency of replacement, increased water use, shampoo, soap, washing machine, clothes dryer, added electrical usage, employee work time, morale, employee retention and many other details should be objectively calculated. It is useful to compare showering to exchange of boots and outerwear, but when working with ABF pigs best practices are needed. Downtime rules often create significant costs but have very limited exclusion value. Determining IC for downtime rules is difficult but no more difficult than the calculation of its DEV. Establishing universal DEV and IC for each economically important disease agent is worthy of considerable research dollars. Many of these issues must be solved before ABF production can compete in the US and global marketplace.

Conclusions

The livestock industry has a key responsibility to contribute meaningfully to producing enough food for the world's people while limiting resource use. The animals in production systems must be healthy in order to meaningfully meet that goal, and also to provide adequate well-being of the animals. Reduction or elimination of antibiotic use increases markedly the challenge of keeping animals healthy, and it demands attention to the entire herd health management programme.

The industry now has available a rich supply of feed ingredients that may improve animal health, and a few of them are reviewed here. Spray-dried plasma provides dramatic benefits, and immune egg products, acids, lactose, mannan oligosaccharide and others appear useful. The impact of dietary fibre is unclear.

Producing pigs without antibiotics clearly increases costs; US consumers have been willing to pay that additional cost for only certain cuts. Antibiotic-free (ABF) pigs are generally less healthy than conventionally produced pigs. The production of ABF pigs in the US has been dominated by large integrated production systems because they can provide the consistency of volume and quality needed. Producing pigs without antibiotics intensifies the need to manage pig health throughout a production pyramid. Vaccines and management strategies are useful but often inadequate. Elimination of PRRS and *Mycoplasma hyopneumoniae* is very useful when producing ABF pigs. Functional biosecurity is critical; a method for evaluating specific biosecurity interventions is described.

Overall, ABF production is much like flying an airplane; inherently safe but disastrous results may occur from even minor human errors or natural events.

Antimicrobial Resistance in Scandinavia after Termination of Antimicrobials for Growth Promotion

28

Martin Wierup
Swedish University of Agricultural Sciences, Uppsala, Sweden
Björn Bengtsson
National Veterinary Institute, Uppsala, Sweden

Introduction

In the Scandinavian countries (Denmark, Finland, Norway and Sweden), antimicrobial growth promoters (AGP) were gradually introduced during the 1960s for use in chicken, beef and pig production. In the early 1970s there were doubts in Sweden about the growth-promoting effect in calves (Johnson and Jacobsson, 1973). The risk of antibiotic resistance was also discovered (Wierup et al., 1975). As a result, the use of AGP in calf and beef production had more or less come to an end before 1986, when all use of AGP was banned. All antimicrobials were then re-classified as veterinary medicines in Sweden and were available only by veterinary prescription.

About ten years later, when the risks of antibiotic resistance were increasingly apparent in other countries and in the EU, AGP were gradually removed in the other Scandinavian countries too (Grave et al., 2006).

These precautionary actions taken by the Scandinavian countries were a major incentive in the EU and internationally for a focus on the use of antibiotics and in particular AGP. A WHO risk assessment on the use of antimicrobials in food animals identified areas of concern in 1997 (WHO, 1997). Later, the WHO published global principles and strategies for the containment of antimicrobial resistance (WHO, 2000; WHO 2001a), as well as on monitoring the use of antimicrobials in food animals for the protection of human health (WHO, 2001b). In the EU, the Scientific Steering Committee of the European Commission (1999) adopted an opinion and recommendations on antimicrobial resistance, with special focus on the usage of antimicrobials for growth promotion, which resulted in the EU ban from July 1 1999 on the use of bacitracin, avoparcin, spiramycin, tylosin and virginiamycin as AGP. A total EU ban on the use of all antibiotics for growth promotion was introduced from 2006. The OIE and individual countries and organisations also focused on the subject.

This chapter describes the usage of antimicrobials and the resistance pattern following the ban on AGP in the Scandinavian countries.

Data Sources

In Sweden, the usage of antimicrobials in animals has been monitored since 1980 and antibiotic resistance has been monitored on a regular basis since 2000 (SVARM, 2007) Corresponding monitoring programmes on the use of antimicrobial agents and antimicrobial resistance are also in place in the other Scandinavian countries and the results are published annually for Denmark (DANMAP, 2007), Norway (NORM NORM-VET, 2007) and Finland (FINRES-Vet, 2007).

Consumption of Antimicrobials

Antibiotic Resistance

Sweden

On a national basis, the current annual usage (2008) in Sweden is approximately 65% lower than before the AGP ban (Figure 28.1). Directly following the ban on AGP, the total use of antibiotics increased but during the period 1988 to 1994 it remained stable at approximately 30 tonnes of active ingredient per year, a level approximately 35% below that before the new legislation was introduced. The consumption has since further decreased and in 2008 was about 16.4 tonnes, the bulk (47%) being penicillin intended for treatment of individual animals. The decreased use of antimicrobials is also evident when calculations are based on dose units instead of weight of active substance and related to changes in number of animals (Greko, 1998). Of special interest in the context of development of antibiotic resistance is the use of antimicrobials for group or flock medication. In 1984 about 65% of the total amount used was formulations intended for such use, whereas in 2008 this figure was about 16%, while 60% was used for injections and 24% for oral medication of individual animals.

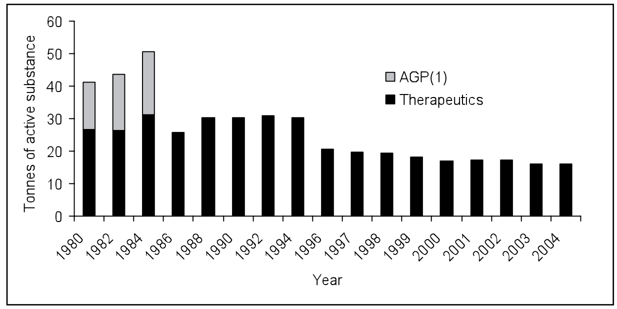

Figure 28.1. Total sales of antimicrobials for animal use in Sweden. Source: SVARM, 2008.
[1)] AGP =Antimicrobial Growth Promoter.

Denmark

Overall, the total consumption of antibiotics in food animal production in Denmark decreased by 47% from 1994 to 2004 (Figure 28.2). About 80% of the total consumption is used in pigs and during the period studied pig production steadily increased from 20.7 million head in 1994 to 25.1 million head in 2004.

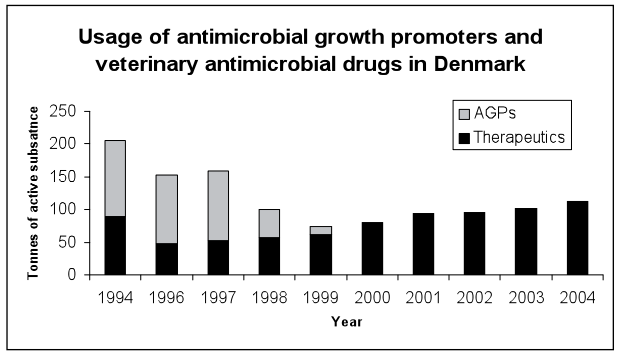

Figure 28.2. Usage of antimicrobial growth promoters (AGP) and veterinary antimicrobial drugs in Denmark in the period 1994-2004 (farmed fish not included). Source: DANMAP, 2004.

Norway

The annual usage of veterinary antimicrobial drugs in Norway decreased gradually by 40% from 1995 to 2001, and has thereafter remained stable (Figure 28.3). The patterns of use have gradually become more favourable as the proportion of penicillin use has increased.

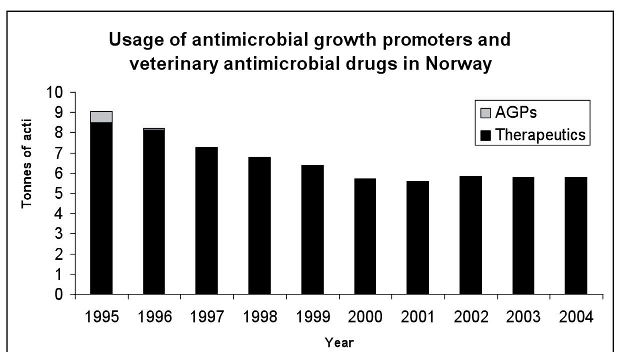

Figure 28.3. Usage of antimicrobial growth promoters (AGP) and veterinary antimicrobial drugs in Norway in the period 1995-2004 (farmed fish not included). Source: NORM NORM-VET.2004.

Finland

The total volume of antimicrobial products used in veterinary medicine in Finland has declined consistently over recent years: by 27% from 1995 to 2002 (FINRES-Vet, 2007). Beta-lactams accounted for 60%, trimethoprim-

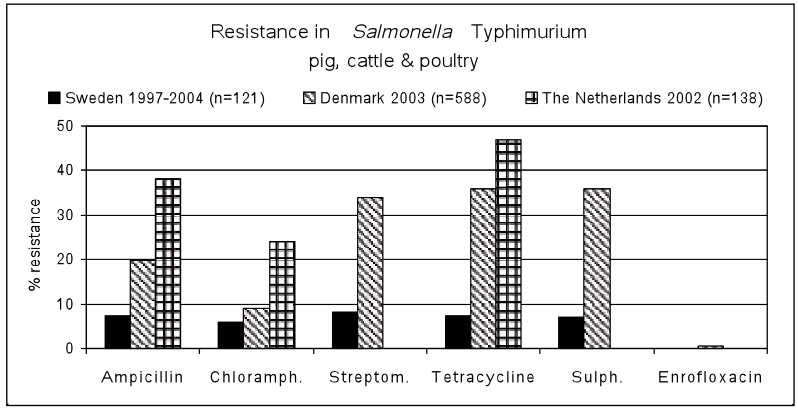

Figure 28.4. Antibiotic resistance in *Salmonella* Typhimurium isolated from pig, cattle and poultry. Sources: SVARM, 2004; DANMAP, 2002; MARAN, 2002; Bengtsson and Wierup, 2006.

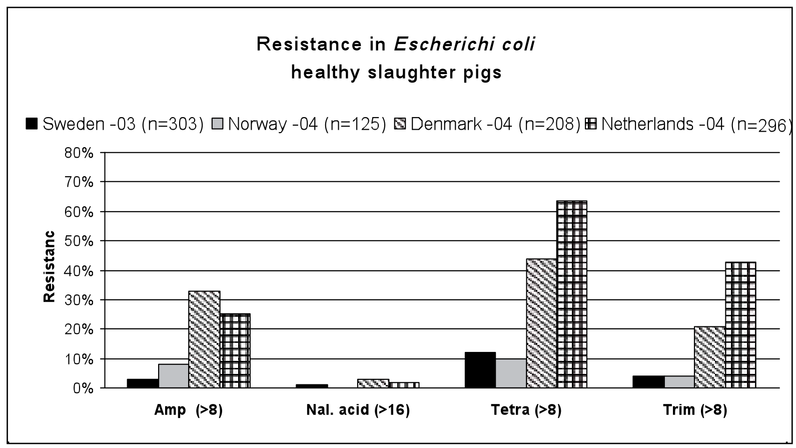

Figure 28.5. Antibiotic resistance in Escherichia coli isolated from healthy slaughter pigs. Breakpoints for resistance (mg/L) are indicated. Sources: SVARM, 2003; NORM-VET, 2004; DANMAP, 2004; MARAN, 2004; Bengtsson and Wierup, 2006.

sulphonamide for 18% and tetracyclines for 15% of the total veterinary antimicrobial sales in 2002. Penicillins sensitive to beta-lactamase accounted for 85% of the veterinary penicillin preparations sold. The proportions of antimicrobials used in medicated feed in 2002 accounted for only 4%, of which about two-fifths each were used for fur animals and fish, and one-fifth for pigs.

The monitoring in the Nordic programmes is principally directed towards zoonotic bacteria, indicator bacteria and to some extent pathogenic bacteria. In the results presented (Figures 28.4 to 28.9), data from some other EU countries are presented for comparison.

As the ionophores used as coccidiostats (not covered by the AGP ban) also have a preventive effect on outbreaks of necrotic enteritis, a study was performed on the resistance to those drugs in *Clostridium perfringens* in Sweden, Norway and Denmark (Johansson et al., 2004). It was found that all isolates tested (102 from 1996-2001) were susceptible to the ionophor narasin and in a similar study in Belgium all 47 isolates tested were found to be susceptible to monesin, lasalocid, salinomycin, maduramycin and narasin (Martel et al., 2004).

Antibiotic resistance in zoonotic bacteria is exemplified here by *Salmonella* Typhimurium isolated from pigs, cattle and poultry in Sweden, Denmark and the Netherlands. A clear difference can be seen in the prevalence of resistance to all the antimicrobials tested between the countries, Sweden having the most favourable situation (Figure 28.4).

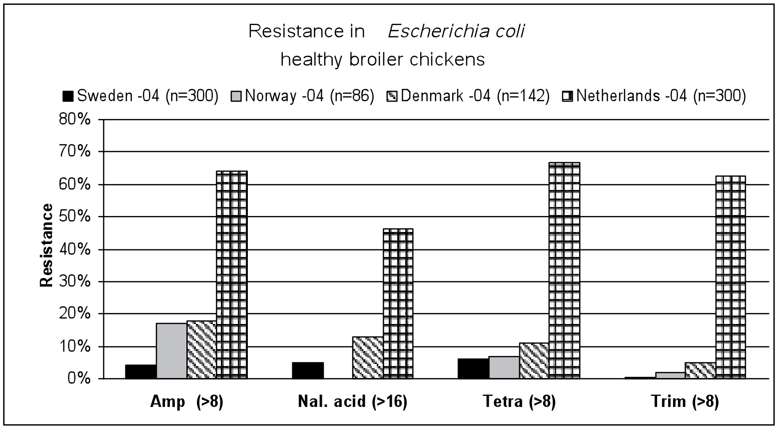

Figure 28.6. Antibiotic resistance in Escherichia coli isolated from healthy broiler chickens. Breakpoints for resistance (mg/L) are indicated. Sources: SVARM, 2004; NORM-VET, 2004; DANMAP, 2004; MARAN, 2004; Bengtsson and Wierup, 2006.

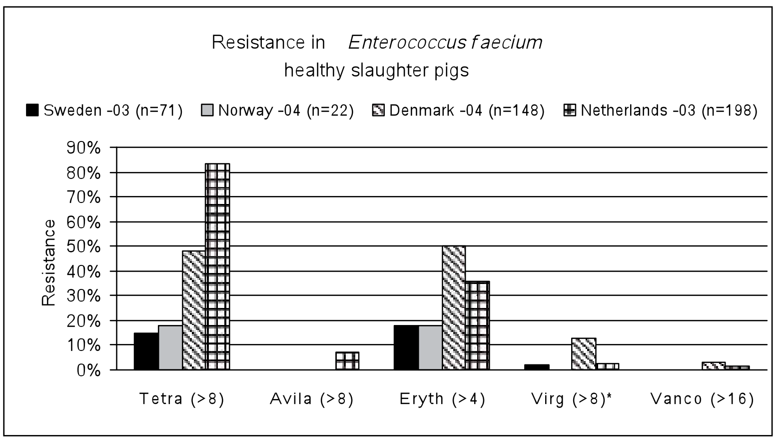

Figure 28.7. Antibiotic resistance in Enterococcus faecium isolated from healthy slaughter pigs. Breakpoints for resistance (mg/L) are indicated. * quinupristin/dalfopristin tested in Denmark. Sources: SVARM, 2003; NORM-VET 2004; DANMAP, 2004; MARAN, 2004; Bengtsson and Wierup, 2006.

Antibiotic resistance in indicator bacteria is exemplified here by *Escherichia coli* and *Enterococcus faecium* isolated from healthy slaughter pigs and broiler chickens (Figure 28.5 to Figure 28.8). Again, a similar picture as above can be seen.

The levels of resistance in the above three groups of bacteria (animal pathogens, zoonotic agents and indicator bacteria) are generally similar in Sweden, Norway and Finland and higher in Denmark. However, in all the Scandinavian countries the level is lower than in some other EU countries previously reported to have a higher use of antimicrobials (EMEA, 1999).

Figure 28.9 shows the resistance to tetracycline in *E. coli* from healthy pigs and the sale of tetracycline in four countries. As can be seen, the prevalence of resistance is rather proportional to the amounts sold.

No monitoring of indicator bacteria occurred in Sweden prior to the 1986 ban on AGP and such data are only available from Denmark. In that country, after the ban on AGP a dramatic reduction occurred in the animal reservoir of enterococci resistant to avoparcin, avilamycin and virginiamycin previously used as growth promoters, as shown in Figure 28.10.

Conclusions

The prevalence of antibiotic resistance in indicator bacteria, animal bacterial pathogens or zoonoses is considerably lower in the Scandinavian countries than in some other countries in the EU reported to have a higher use

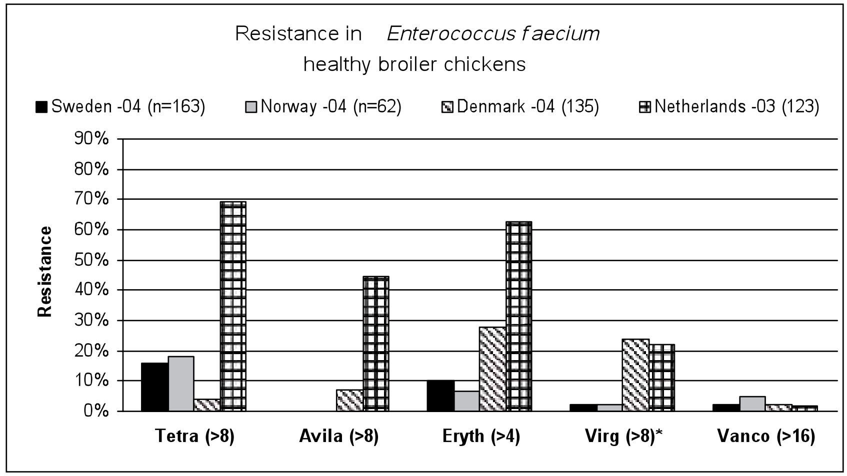

Figure 28.8. Antibiotic resistance in Enterococcus faecium isolated from healthy broiler chickens. Breakpoints for resistance (mg/L) are indicated. *quinupristin/dalfopristin tested in Denmark. Sources: SVARM, 2004; NORM-VET, 2004; DANMAP, 2004; MARAN, 2004; Bengtsson and Wierup, 2006.

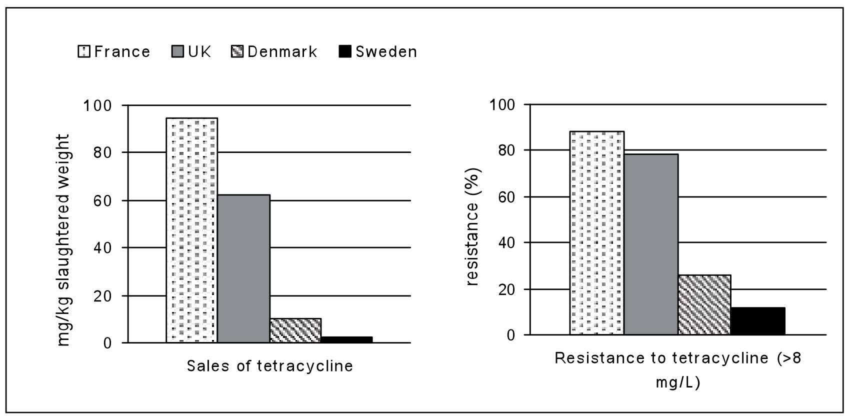

Figure 28.9. Use of tetracycline in animals and resistance in E. coli from healthy pigs in France, UK, Denmark and Sweden. Sources: DEFRA, 2004; AFSSA, 2003; DANMAP, 2002; SVARM, 2003; Bengtsson and Wierup, 2006.

of antimicrobials. Within the Scandinavian countries this prevalence is generally also highest in Denmark. These results most likely reflect the magnitude of exposure, as it is generally recognised that the risk of bacterial strains acquiring resistance to antibiotics increases with their exposure to such substances. As an example, Sweden, with a relatively very low prevalence of antibiotic resistance, banned the use of AGP 20 years ago and controlled use of antimicrobials was started even earlier, as well as organised actions for the prevention of infectious diseases in livestock production.

The termination of the use of AGP has not significantly influenced the resistance pattern in Norway, Finland and Sweden and the prevalence is maintained at a relatively low level. However, it is interesting that following the withdrawal of AGP in Denmark, a dramatic reduction occurred in the animal reservoir of enterococci resistant to avoparcin, avilamycin and virginiamycin, antimicrobials previously used as growth promoters. This effect is likely to be the result of the withdrawal of the selection pressure for strains resistant to AGP. The same event might also have occurred in the other Scandinavian countries but was not recorded due to lack of data before the ban.

The low prevalence of resistance in *Salmonella* in Sweden (Figure 28.4) reflects not only a limited use of antimicrobials, but also that Sweden has long applied a

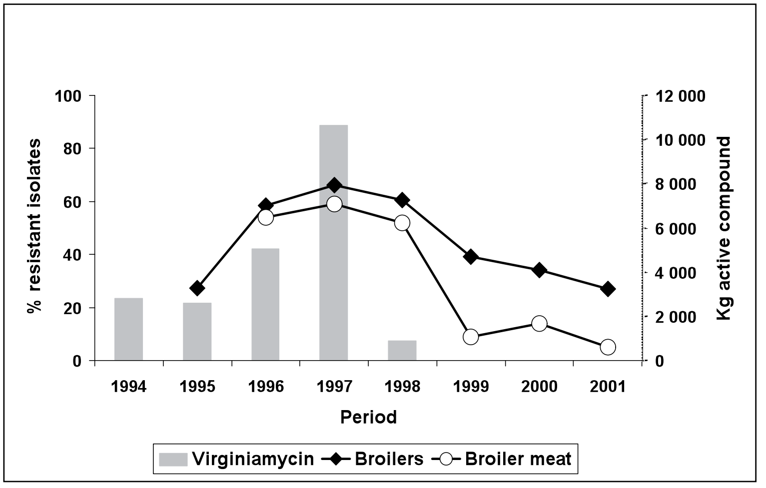

Figure 28.10. Trend in virginiamycin resistance among Enterococcus faecium from broilers and broiler meat and usage of the growth promoter virginiamycin in Denmark. Source: DANMAP, 2001.

zero tolerance control policy for *Salmonella* contamination (EFSA, 2006; EFSA, 2007). That policy limits the exposure of *Salmonella* to antimicrobials and reduces the spread of multiresistant strains, as salmonella infections in animals are not given antibiotic treatment.

It should be emphasised that the reduction in the use of antimicrobials and the relatively favourable situation for antibiotic resistance is not only a result of the withdrawal of AGP. As can be seen from the annual reports from all the Scandinavian countries, large efforts and industry-based campaigns have been devoted to implementation of optimal disease preventive management routines and proper use of antimicrobials, e.g. guidelines on antimicrobial drug therapy in food animals as described in other chapters of this book by Wierup (Chapter 25) and Pettigrew and Baker (Chapter 27). One of the main messages was critical selection of cases for antimicrobial therapy. In Denmark a decrease in the use of antimicrobials was seen when, as applied earlier in the other Scandinavian countries, a ban was introduced on economic incentives to veterinary surgeons for prescribing antibiotics to producers (Kjeldsen and Callesen, 2006).

The impact on animal health and production of AGP withdrawal is reported elsewhere (Wierup, 2001; WHO/CPE/ZFK/2003.1; Laine et al., 2004). However, it should be emphasised that weaner pig production requires special attention in order to prevent those clinical problems that to a varying extent occurred in Sweden (Wierup, 2001), Finland (Laine, et al., 2004) and Denmark (WHO/CPE/ZFK/2003; Kjeldsen and Callesen, 2006) following the withdrawal of AGP. The major growth promoting effect was thus found to be the control of enteric infections by the AGP. However, evaluations of controlled data generated in these countries demonstrated that in finisher swine production (> 25 to 30 kg) and in broiler production, no or limited negative effects were found (Wierup and Wegener, 2006).

In summary, the experiences from Denmark, Finland, Norway and Sweden show that termination of AGP has significantly decreased the overall usage of antimicrobials and the risk of future problems with antibiotic resistance. It has also considerably reduced the animal reservoir of enterococci resistant to antibiotics previously used for growth promotion and decreased the risk of human exposure via the food chain of genes coding for resistance to antimicrobials. Naturally, some antibiotic resistance in animal bacterial pathogens occurs in the Scandinavian countries, which is the reason for establishing a continuous monitoring programme of antimicrobial resistance, as well as a focus on prudent use of antimicrobials in food animal production.

Environmental Impacts of Antibiotic Use in the Animal Production Industry

29

Joanne C. Chee-Sanford
U.S. Department of Agriculture , Agricultural Research Service, Urbana, IL, USA
Ivan J. Krapac
Illinois State Geological Survey, Champaign, IL, USA
Anthony C. Yannarell and Roderick I. Mackie
University of Illinois at Urbana-Champaign, Urbana, IL, USA

Introduction

Antibiotics are routinely used in the livestock industry to treat and prevent disease. In addition, subtherapeutic concentrations of antimicrobials are commonly added to animal feed and/or drinking water sources as growth promoters, and have been a regular part of swine production since the early 1950s (Cromwell, 2001). When used in this manner, antibiotics can select for resistant bacteria in the gastrointestinal tract of production animals, providing a potential reservoir for dissemination of drug-resistant bacteria into other animals, humans and the environment (Andremont, 2003). Bacteria have been shown to readily exchange genetic information in nature, permitting the transfer of different resistance mechanisms already present in the environment from one bacterium to another (Stewart, 1989; Amábile-Cuevas and Chicurel, 1992; Salyers and Amábile-Cuevas, 1997). Transfer of resistance genes from faecal organisms to indigenous soil and water bacteria may occur (Lorenz and Wackernagel, 1994; Daane et al., 1996; DiGiovanni et al., 1996; Nielsen et al., 2000), and because native populations are generally better adapted for survival in aquatic or terrestrial ecosystems, persistence of resistance traits may be likely in natural environments once they are acquired. Antibiotic resistance has received considerable attention due to the problem of emergence and rapid expansion of antibiotic-resistant pathogenic bacteria.

The potential for long-term, cumulative inputs of antibiotics and correspondingly, their potential effects on acquisition and maintenance of antibiotic resistance mechanisms in bacteria, collectively suggest a degree of impact on the occurrence, persistence and mobility of resistance genes in natural environments. A number of reviews, reports and opinion papers have emerged to address the possible link between antibiotic use and the impact on antibiotic resistance development (e.g. Gustafson and Bowen, 1997; Khachatourians, 1998; USGAO, 1999; Isaacson and Torrence, 2002; Séveno et al., 2002; Kümmerer, 2004). These papers have highlighted various issues related to antibiotic use in agriculture, often focusing on the link to emerging antibiotic-resistant bacteria, gene transfer mechanisms and consequent risks to human and animal health.

Table 29.1. Survey of the most commonly used antibiotics in animal production in the United States. Source: AHI, 2001.

Antibiotic class	Amount (millions of kg)
Ionophores/arsenicals	3.53
Tetracyclines	3.25
Other antibiotics - includes macrolides, lincosamides, polypeptides, streptogramins, cephalosporins	1.94
Penicillins	0.823
Sulphonamides	0.269
Aminoglycosides	0.117
Fluoroquinolones	0.016

In the following review, we provide an overview of antibiotic use and animal waste management in the United States, the dissemination and fate of antibiotic residues, and the environmental persistence, mobility and transferability of antibiotic resistance determinants and their bacterial hosts in the context of environmental conditions encountered during the course of manure storage and in natural soil and water environments following the practice of land application of animal waste. This description provides a background to determining the true ecological impact of antibiotics and antibiotic resistance genes in natural environments.

Antibiotic Use in Animal Agriculture

In commercial livestock production, antibiotics are used: 1) therapeutically to treat existing disease conditions, 2) prophylactically at subtherapeutic doses to mitigate infection by bacterial pathogens of livestock animals undergoing high stress situations, and 3) subtherapeutically to enhance growth. A survey of members of the Animal Health Institute reported that overall, the ionophores/arsenicals and tetracycline classes of antibiotics were the most commonly used antimicrobials in animal production (AHI, 2001; Table 29.1). Among the antibiotics commonly used in swine, poultry and beef cattle, penicillins, macrolides, polypeptides, streptogramins and tetracyclines are used not only for purposes of disease treatment and disease prevention, but also for growth promotion (Table 29.2).

Table 29.2. Antibiotics commonly used in swine, poultry and beef cattle production industries in the United States. Source: USGAO, 1999.

Antibiotic class (examples)	Industry
Aminoglycosides (gentamicin, neomycin streptomycin)	Swine, Poultry, Beef Cattle
ß-Lactams (penicillins, ceftiofur)	Swine, Poultry, Beef Cattle
Chloramphenicol (florfenicol)	Beef Cattle
Ionophores (monensin, salinomycin, semduramicin, lasalocid)	Poultry, Beef Cattle
Lincosamides (lincomycin)	Swine, Poultry
Macrolides (erythromycin, tilmicosin, tylosin)	Swine, Poultry, Beef Cattle
Polypeptides (bacitracin)	Swine, Poultry
Quinolones (Fluoroquiniolones) (sarafloxacin, enrofloxacin)	Poultry, Beef Cattle
Streptogramins (virginiamycin)	Swine, Poultry, Beef Cattle
Sulphonamides (sulphadimethoxine, sulphamethazine, sulphisoxazole)	Swine, Poultry, Beef Cattle
Tetracyclines (chlortetracycline, oxytetracycline, tetracycline)	Swine, Poultry, Beef Cattle
Others	
Bambermycin	Swine, Poultry, Beef Cattle
Carbadox	Swine
Novobiocin	Poultry
Spectinomycin	Swine, Poultry

Other classes, such as quinolones, lincosamides and aminoglycosides, are primarily used only in disease treatment or prevention. The Animal Health Institute (AHI, 2001) and Union of Concerned Scientists (UCS, 2001) recently reported two different estimates of antibiotic usage in agriculture.

The AHI reported a total of 9.2 million kg of antibiotics sold for all animal use in 1999. Of these 9.2 million kg, 8 million kg were used for treatment and prevention of disease and only 1.3 million kg for improving feed efficiency and enhancing growth. In contrast, the UCS reported that 11.2 million kg of antibiotics were used for non-therapeutic purposes alone in the swine, poultry and cattle industries. According to the UCS report, livestock use accounts for the major share of total antimicrobials used in the United States, estimated at 22.7 million kg annually, based on extrapolation from a 1989 Institute of Medicine report (IOM, 1989).

Management of Animal Waste from Production Agriculture

Historically, until the mid- to late 1970s, livestock operations were usually part of larger integrated farming operations that produced crops. Swine production, in particular, has seen a trend towards specialised large production facilities. Over the last 25 years, swine production has largely shifted from such integrated farming systems to concentrated animal feeding operations (CAFOs) that may house thousands of animals. In 1984, there were approximately 690,000 US producers producing 9.1 billion kg of pork. By 2000, about 95,000 producers were producing 11.8 billion kg of pork (USDA NASS, 2002). Due to geographical patterns of feed grain production and other market forces, CAFOs have become concentrated in certain geographical regions in the US, primarily North Carolina and the Midwest. US Department of Agriculture surveys performed in 2000 found that 28.3% of swine facilities were located within half a mile of another swine production site and 53.9% were within one mile of another site (USDA, 2001a, 2001b).

Under the earlier integrated system of production, producers typically owned large tracts of land necessary for agronomic activity. Waste and effluent from a modest number of animals was applied rotationally over different fields, effectively diluting nutrients and recycling waste for fertiliser use. Swine each typically produce approx. 1.5 tonnes of fresh manure in the 5-6 months it takes to grow them to a market weight of 114 kg (Richert et al., 1995). The National Agricultural Statistics Service (NASS) estimated that in 2002, 185 million head of swine were sold in the US, generating approx. 2.8×10^8 tonnes of fresh manure annually. With the advent of CAFOs, large quantities of waste are concentrated in a single location and/or region, and producers may only own sufficient land to site their facilities.

The most common method to dispose of swine effluent in the United States is through land application, where application of liquid manure at agronomic rates can produce crop yields that equal those obtained with chemical fertilisers (Schmitt et al., 1995). In order to utilise and dispose of the manure effluent, CAFO operators often contract with neighbouring growers to apply effluent to their land or apply it to land surrounding the facility. Because it is costly to transport liquid effluent any great distance, there is an incentive to apply effluent as close to the source as possible. In the US the crop cycles coincide with seasonal cycles, with the application of manure occurring between crop cycles. For many locations, manure is stored for six months to one year before being applied to crop fields as fertiliser. Effluent differs from fresh manure in that it has a much greater water volume. Fresh swine waste contains approximately 10% solids, while deep pit effluents are 4-8% solids and lagoon effluents less than 0.5-1% (Fulhage and Post, 2005). O'Dell et al. (1995) found that the solids content ranged from 4-10 g/L in 18 separate tank loads of swine effluent that had been agitated for 24 h prior to application, suggesting effluent application rates can be highly variable.

Confinement livestock production, especially large animal facilities, is increasingly a source of surface water and groundwater contamination. The widespread practice of land application prompted the Environmental Protection Agency (EPA) in the 1990s to require nutrient management plans for CAFOs. Initially, nutrient management plans were nitrogen-based, requiring manure to be applied at a rate that would not exceed crop nitrogen requirements. Swine manure, however, has a high phosphorus content relative to nitrogen content; as excreted, swine manure contains a P_2O_5:N ratio of approximately 0.86:1 (LPES, 2005). Applying effluent to meet the nitrogen requirements of a crop often leads to a build-up of phosphorus in the soil, in some instances to values in excess of 2,000 mg/kg of total soil phosphorus (Lehmann et al., 2005).

The three primary methods used to apply effluent are: 1) surface application, 2) surface application followed by incorporation, and 3) direct soil injection. One primary reason to incorporate surface-applied effluent is to limit the loss of nitrogen by at least 50% compared with surface application alone (Rotz, 2004). Other reasons include odour reduction and minimisation of surface runoff. The preferred method of application from a nutrient management standpoint is deep injection into the soil, which eliminates the nitrogen losses associated with other methods, reduces odour and virtually eliminates the possibility of surface runoff.

Because surface application has been associated with nitrogen losses, it is often considered 'environmentally unfriendly', yet it has merits as a method of managing pathogen loads. Hutchison et al. (2004) reported that the mean D-value, or time needed to reduce the variable being measured by one order of magnitude, for four zoonotic pathogens, *Salmonella* sp., *E. coli* 0157, *Listeria* sp. and *Campylobacter* sp., was 1.42 days for unincorporated pig slurry and 2.48 days for slurry incorporated immediately after application. These pathogens also declined at similar rates regardless of season (summer versus winter). Desiccation may be an important factor in population decline because more intense UV radiation in the summer would be expected to accelerate cell mortality (Booth et al., 2001; Hoerter et al., 2005). A significant rainfall event immediately following surface application of effluent would likely result in vertical movement of bacteria and mobile compounds into the soil profile as well as off-site movement due to surface runoff (Saini et al., 2003). Surface applications to frozen soil are usually avoided because of the likelihood of significant runoff.

Entry of Antibiotics into the Environment

Antibiotics used in animal agriculture can enter the environment via a number of routes, including the drug manufacturing process, disposal of unused drugs and containers, and through the use and application of waste material containing the drugs. The excretion of waste products by grazing animals, atmospheric dispersal of feed and manure dust containing antibiotics, and the incidental release of products from spills or discharges are also potential pathways of antibiotic residue entry into the environment.

Many antibiotics are not completely absorbed in the gut, resulting in the excretion of the parent compound and its breakdown metabolites (Feinman and Matheson, 1978; Halling-Sørensen et al., 1998; Boxall et al., 2004). Elmund et al. (1971) estimated that as much as 75% of the antibiotics administered to feedlot animals could be excreted into the environment. Feinman and Matheson (1978) suggested that about 25% of the oral dose of tetracycline is excreted in faeces and another 50-60% is excreted unchanged or as an active metabolite in urine. Oral administration of the macrolide tylosin resulted in a maximum of 67% of the antibiotic excreted, mainly in the faeces.

The practice of land application of livestock manure provides large-scale areas for introduction of antibiotics into the environment. Once released into the environment, antibiotics can be transported either in a dissolved phase or (ad)sorbed to colloids or soil particles into surface water and groundwater (Campagnolo et al., 2002; Kolpin et al., 2002; Yang and Carlson, 2003; Krapac et al., 2004). Manure and waste slurries potentially contain significant amounts of antibiotics and their presence can persist in soil after land application (Donohoe, 1984; Gavalchin and Katz, 1994).

Chemical Characteristics of Antibiotics and Behaviour in Soil and Water

Veterinary antibiotics comprise a group of organic compounds that have a wide variety of functional groups that affect their chemical properties. The octanol-water partition coefficient (K_{ow}) is used as a general measure of hydrophobicity, and most antibiotics have log K_{ow} values less than 5, indicating that they are relatively non-hydrophobic (Tolls, 2001). In addition, the water solubility for many antibiotics exceeds 1 g/L, suggesting that they are relatively hydrophilic. Tolls (2001) and Boxall et al. (2004) compiled sorption coefficients (K_d) for a variety of antibiotics, soils and soil components measured over the course of many studies. Based on K_d values, antibiotics exhibit a range of affinities for the solid phase (K_d 0.2-6,000 L/kg), with consequent effects on their mobility in the environment. Estimations of antibiotic organic carbon-normalised sorption coefficients (K_{oc}) made by using a compound's octanol-water partition coefficient (K_{ow}) generally results in underestimates of the K_{oc} value, suggesting that mechanisms other than hydrophobic partitioning occur. Cation exchange, surface complexation and hydrogen bonding are included as likely mechanisms for antibiotic sorption to soils. Many of the acid dissociation constants (pK_a) for antibiotics are in the range of soil pH values, such that the protonation state of these compounds depends on the pH of the soil solution (Tolls, 2001).

Studies have shown that under a broad range of environmental conditions, tetracyclines (tetracycline, chlortetracycline and oxytetracycline) can adsorb strongly to clays (Pinck et al., 1961a, 1961b; Sithole and Guy, 1987a, 1987b; Allaire et al., 2006), soil (Krapac et al., 2004) and sediments (Rabolle and Spliid, 2000). Sorption of chlortetracycline also occurs rapidly in sandy loam soil (Allaire et al., 2006). Macrolides such as tylosin have a weaker tendency to sorb to soil materials (Rabolle and Spliid, 2000), although a sorption kinetic study showed that 95% of tylosin was sorbed within 3 h in both sandy loam and clay soils (Allaire et al., 2006). Sulphonamides exhibit weak sorption to soil, and are probably the most mobile of the antibiotics (Tolls, 2001). Pinck et al. (1962) determined that two macrolide antibiotics (carbomycin and erythromycin) sorbed significantly (231-263 mg/g) to montmorillonite and to a much lesser extent (0-39 mg/g) to vermiculite, illite and kaolinite. In a review on the fate of antibiotics in the environment Huang et al. (2001) concluded that there was little information on the sorption of aminoglycoside and beta-lactam antibiotics. Because aminoglycosides can be protonated under acidic conditions, they could be sorbed to clay minerals under certain conditions, while β-lactams are highly polar compounds and would not be expected to sorb readily to soil components. Because of the strong sorption of the tetracycline and macrolide antibiotics, their mobility in the environment may be facilitated by transport with manure and soil colloidal material (Kolz et al., 2005a). Interestingly, although most antibiotics do not require metal ion coordination to exert biological action, other compounds such as bacitracin, streptonigrin, bleomycin and tetracycline have prerequisites for binding of metals ions to function properly (Ming, 2003). Sorption of these drug compounds in clays, where intercalation of metal complexes occur, may provide suitable conditions for the drug to exert a biological effect.

Mechanisms of Antibiotic Degradation

Because antibiotics are generally introduced from livestock operations via water (effluent) into the environment, hydrolysis can be an important degradation pathway. β-Lactams, macrolides, and sulphonamides appear to be the most susceptible classes of antibiotics to hydrolysis (Huang et al., 2001). At near neutral pH, tylosin A was found to have a hydrolysis half-life of 300 to 500 hours at 60°C (Paesen, 1995). At more environmentally relevant temperatures, these half-lives are expected to be longer. Doi and Stoskopf (2000) determined that under relatively high temperatures (43°C) the half-life of oxytetracycline in deionised water was 0.26 days, but was relatively stable at 4°C. β-Lactams are rapidly hydrolysed under mild acidic and basic conditions (Hou and Poole, 1969; Huang et al., 2001). Photolysis can be another abiotic transformation process affecting antibiotics introduced into the environment. The photodegradation of antibiotics in soil can occur at the soil-atmosphere interface and at the surface of liquid manure. Soils can provide a much different photodegradation environment than aqueous solutions and transformation rates can vary significantly in soils compared with those in water (Balmer et al., 2000). Quinolones and tetracylines are susceptible to photodegradation (Huang et al., 2001), and photodegradation of oxytetracycline is three times more rapid under light than dark conditions (Doi and Stoskopf, 2000). Halling-Sørenson (2000) suggested that tylosin might be resistant to photolysis because it has only limited light absorbance in the visible spectrum, and Boxall et al. (2004) determined that sulphonamides would not be readily photodegraded. Beausse (2004) concluded that photodecomposition of antibiotics under field conditions was negligible when compared with other abiotic processes.

Limited numbers of studies have been conducted to assess the biodegradation of antibiotics. Biodegradation of organic compounds by microorganisms in soil is dependent in part on factors such as temperature, concentration, bioavailability, time of exposure, availability of other nutrients and the enzymatic capabilities of the extant microbial population. Aerobic processes have been the primary focus of such studies, and little attention has been devoted to anaerobic processes, the latter being of significance in the soil subsurface and microzones.

Depending on test conditions, biodegradation half-lives of organic compounds can widely vary. Studies using standard laboratory test assays have demonstrated limited or no degradation of antibiotics such as metro-

Table 29.3. Persistence of antibiotics in manure (modified from Boxall et al., 2004).

Antibiotic class	Half-life (d)
Aminoglycosides	30
β-Lactams	5
Macrolides	<2-21
Quinolones	100
Sulphonamides	<8-30
Tetracyclines	100

nidazole and oxytetracycline (Jacobsen and Berglind, 1988; Samuelsen et al., 1994; Kümmerer et al., 2000). In another study of 18 antibiotics tested, none were found to be readily biodegraded, although some activity occurred when additional nutrient supplement was made (Alexy et al., 2004). Penicillin G was found to be readily biodegradable along with some biodegradation of amoxicillin, imipenem and nystatin (Gartiser et al., 2007a; 2007b). A study of aquaculture sediments showed bacterial mineralisation of erythromycin A (Kim et al., 2004). Inherent to the process of biodegradation, the toxic effects of antibiotics on the resident bacteria have also been demonstrated. A range of antibiotic concentrations were found to inhibit activated sludge in wastewater treatment (Alexy et al., 2004; Gartiser et al., 2007a), but the exact effects of antibiotic entry in natural environments on microbial populations resident to these systems are not yet known.

Another biological mechanism of antibiotic fate, plant uptake and bioaccumulation of antibiotics, has received considerable interest due to issues of food safety and human health. A number of studies have shown this mechanism to occur with a variety of plant species (e.g. ; Kumar et al., 2005; Boxall et al., 2006; Dolliver et al., 2007), and while significant, discussion of these processes is outside of the scope of this review.

Persistence of Antibiotics in Manure

Antibiotics excreted from animals are often concentrated in the solid phase because of sorption dynamics (Tolls, 2001; Loke et al., 2002; Kolz et al., 2005a, 2005b). Half-lives that have been reported for a variety of antibiotic classes in manure (Boxall et al., 2004) (Table 29.3) were less than the anticipated storage period of manure, suggesting the possibility that significant degradation of the parent compounds might occur prior to land application.

Table 29.4. Antibiotic concentrations detected in manure from swine and poultry lagoons.

Antibiotic	Concentration	Reference
Lincomycin	2.5-240 (μg/L)	Campagnolo et al., 2002
Chlortetracycline	68-1000 (μg/L) 0.1 (mg/kg) <0.5-1.0 (mg/kg)	Campagnolo et al., 2002 Hamscher et al., 2002 Hamscher et al., 2005
Tetracycline/ Oxytetracycline	25-410 (μg/L) 4.0 (mg/kg) 14.1-41.2 (mg/kg)	Campagnolo et al., 2002 Hamscher et al., 2002 Hamscher et al., 2005
Sulphamethazine	2.5-380 (μg/L) 0.13-8.7 (mg/kg) 0.2-7.2 (mg/kg)	Campagnolo et al., 2002 Haller et al., 2002 Hamscher et al., 2005
Sulphadimethoxine	2.5 (μg/L)	Campagnolo et al., 2002
Erythromycin	2.5 (μg/L)	Campagnolo et al., 2002
Penicillin G	2.1-3.5 (μg/L)	Campagnolo et al., 2002

Quinolones and tetracyclines were the most persistent, with half-lives approaching 100 d. Kolz et al. (2005b) determined that 90% of tylosin, tylosin B and tylosin D was lost within 30 to 130 h in anaerobic manure slurries at 22°C. In aerobic manure slurries, 90% of tylosin was lost in 12 to 26 h. Although biodegradation and abiotic degradation occurred, the primary mechanism for tylosin loss was sorption to manure solids (Kolz et al., 2005a, 2005b). Residual tylosin and its breakdown product, dihydrodesmycosin, were also detected in the slurries after eight months. In several studies, tetracycline concentrations were found to be generally higher than those of macrolides, β-lactams and sulphonamides (Table 29.4). Tetracycline concentrations in some swine lagoons were as great as 1 mg/L (Campagnolo et al., 2002). Gavalchin and Katz (1994) determined the persistence of seven antibiotics in a soil-faeces matrix under laboratory conditions and found that the order of persistence was chlortetracycline > bacitracin > erythromycin > streptomycin > bambermycin > tylosin > penicillin with regard to their detection in the soil. The application of manure to agricultural fields also likely introduces breakdown products into the environment along with the parent compound, but persistence data for degradation products were not found in the literature reviewed.

Persistence of Antibiotics in Soil and Water

Until recently, information regarding the occurrence, fate and transport of antibiotics under field conditions has been limited. Spatiotemporal ‘hotspots’ of bacteria and antibiotic residues are likely to occur in effluent-applied soil due to the variable solids content of the waste effluent, soil characteristics and the frequency and timing of application. In a sandy soil that had repeated manure applications, tetracycline and chlortetracycline were detected down to a depth of 30 cm (Hamscher et al., 2002, 2005). The highest tetracycline and chlortetracyline concentrations, 198 and 7.3 μg/kg, respectively, were detected at soil depths of 10-20 cm and 20-30 cm, respectively. Sulphamethazine was generally not detected in soil samples, but was detected in groundwater collected at a depth of 1.4 m. Oxytetracycline, sulphadiazine, sulphathiazole, sulphamerazine, sulphamethoxypyridazine, sulphamethoxazole, sulphadimethoxine and tylosin were not detected in any soil or groundwater samples. While it appeared some of the tetracyclines could accumulate in soil, none of the antibiotics from the study were detected at soil depths greater than 30 cm and only sulphamethazine was detected in groundwater, suggesting limited transport even in highly porous sandy soils.

In a field study with clay loam soil that received swine manure spiked with the sulphonamide sulphachlorpyridazine (SCP), the antibiotic was found to be mobile and readily entered the field drain, with a maximum concentration of 590 μg/L detected seven days after manure application (Boxall et al., 2002). In the same study conducted with sandy loam field soil, SCP concentrations in soil pore water were significantly lower (max. concentration 0.78 μg/L) than for the field with clay loam, and contrasted with laboratory sorption studies that predicted larger soil water concentrations. The lower concentrations detected in the field samples were hypothesised to be the result of SCP degradation. In another soil transport study, SCP and oxytetracycline (OTC) were detected in soil at concentrations up to 365 and 1691 μg/kg, respectively (Kay et al., 2004). Similar to other investigations, these compounds were not detected below a depth of about 37 cm. SCP and OTC were detected in tile drainage at peak concentrations of 613 and 36 μg/L, respectively. Only 0.004% of the OTC that was applied was in the particulate phase, and 23% of OTC moved to tile drainage. The investigators concluded that the antibiotics behaved similarly to pesticides under field conditions, and that tile drainage may be a significant route for these compounds to migrate to surface waters. The manure in this study was surface-applied without incorporation into the soil and the authors suggested that tillage prior to or during manure application might limit transport of antibiotics. In a later study by the same authors, swine manure spiked with SCP, OTC and tylosin was surface-applied to wheat stubble in a clay loam soil and mass recovery of SCP and OTC lost in surface runoff was 0.42 and 0.07%, respectively (Kay et al., 2005). While surface runoff did not appear to be a significant transport loss, the authors suggested that incorporation of manure into the soil would further limit loss from the soil. Tylosin was not detected in any samples, suggesting its rapid degradation in the manure and supporting previous evidence that macrolides may more readily undergo microbial degradation processes. In a study where swine manure was spiked with sulphadiazine and sulphathiazone and irrigated on to grassland, less than 5% of sulphonamide applied was lost to runoff (Burkhardt et al., 2005). The sulphonamide losses were 10 to 40 times greater on the manured plots when compared with control plots, the latter receiving only aqueous solutions of the compounds. The authors concluded that the manure formed a seal at the soil surface, creating conditions for more runoff. In addition, the high manure pH may have caused deprotonation of the sulphonamides, resulting in decreased sorption to the soil. These results suggest that repeated surface application of manure may yield a higher likelihood of runoff situations.

While detection of antibiotic residues poses a challenge in any environmental matrix, detection of low levels of compounds, particularly in natural waterways, is highly challenging. The US Geological Survey (USGS) has a comprehensive stream-monitoring network throughout the US and has improved detection of compounds by developing state-of-the-art analytical techniques such as LC-MS-MS. A recent study by the USGS (Kolpin et al., 2002) conducted a reconnaissance of the occurrence of pharmaceuticals, hormones and other organic wastewater contaminants in water resources. In 139 streams sampled across 30 states during 1999 and 2000, a number of antibiotics occurred in appreciable amounts (Table 29.5).

Carbodox, doxycycline, enrofloxacin, sarafloxacin, sulphachlorpyridazine, sulphamerazine, sulphathiazole and virginiamycin were not detected in any samples. Many of the compounds that were not detected are commonly used in livestock operations, suggesting limited transport of these compounds to surface waters in the aqueous phase. As analytical technologies improve, detection of compounds can provide a more accurate characterisation of the quantities and occurrence of antibiotics in natural soil and water systems.

In a study to investigate the occurrence of five tetracyclines and six sulphonamides in water collected along the Cache la Poudre River, Colorado, no antibiotics were detected in a pristine mountain stretch of the river (Yang and Carlson, 2003). Few sulphonamides were detected along the entire river, but the frequency of detection and concentration of tetracyclines increased as the river water quality became affected by urban and agricultural sources. Tetracycline concentrations in filtered samples ranged from 0.08 to 0.30 µg/L. Photolysis, biodegradation and sorption of the tetracyclines could have occurred in various reaches of the stream but the authors concluded that proximate agricultural activity influenced tetracycline occurrence in the river. In a study to detect antibiotics in surface waters and groundwater, Campagnolo et al. (2002) found 31% and 67% of the samples collected near swine and poultry confinement facilities, respectively, had detectable quantities, albeit low, with less than 10 µg/L.

Few studies have determined the occurrence of veterinary antibiotics in groundwater. Krapac et al. (2004) collected shallow (<8 m) groundwater samples near two swine confinement facilities. Fewer than 5% of the samples contained any of the tetracyclines at either of the facilities. Parent tetracycline compounds were detected in a small number of groundwater samples collected from wells that had also been significantly impacted by manure seepage as evident by elevated chloride, ammonium and potassium concentrations. Tetracycline breakdown products were detected in some groundwater samples even when the parent compound was not detected. When detected, antibiotic concentrations were less than 0.5 µg/L. Hirsch et al. (1999) collected more than 30 groundwater samples from agricultural areas in Germany containing large numbers of animal confinement facilities. Of the 18 antibiotics representing macrolides, sulphonamides, penicillins and tetracyclines, only sulphonamide residues were detected in four samples, and none of the other antibiotics were detected in the groundwater samples. The authors concluded that sulphonamides in two of the samples were the result of sewage irrigation and sulphamethazine detected in the other samples was likely from veterinary use.

Table 29.5. Detection frequency and maximum concentrations of selected antibiotics in 139 filtered stream samples from 30 U.S. states (modified from Kolpin et al., 2002).

Antibiotic	Frequency of Detection (%)	Maximum Concentration (µg/L)
Trimethoprim	27.4	0.30
Erythromycin-H	20.2	1.5-1.7
Lincomycin	21.5	1.7
Sulphamethoxazole	19.0	0.52
Tylosin	13.5	0.28
Roxithromycin	4.8	0.18
Ciprofloxacin	2.6	0.03
Chlortetracycline	2.4	0.69
Oxytetracycline	1.2	0.34

Occurrence of Bacteria and Development of Antibiotic Resistance in Animal Guts

Antibiotic resistance among commensal bacteria represents a major avenue for the development of resistance in bacterial pathogens, since resistances increase first in commensals and are then transferred to pathogens later. First, commensal gut bacteria are likely to be highly efficient contributors to antibiotic resistance because the numbers of commensal bacteria in the intestinal ecosystem are large, often more than 10^{14} bacteria comprising several hundred species (Andremont, 2003). Anaerobic bacteria dominate this ecosystem and number 10^{11}-10^{12} cells/g of intestinal content, whereas enterobacteria and enterococci are relatively minor players ranging from 10^{6} to 10^{8} cells/g of intestinal content. Second, the commensal genetic pool is large and encompasses the potential for many different mechanisms conferring antibiotic resistance. Third, antibiotic-resistant commensal bacteria may be selected each time an antibiotic is administered regardless of the health status of the animal. This microbial population is excreted in faeces and stored as manure where it undergoes changes in the numbers and proportions of the dominant bacterial species. An analysis of stored swine manure indicated that the predominant culturable microorganisms from these environments were obligately anaerobic, low mol% G + C Gram positive bacteria (Firmicutes) comprising members of the Clostridial, Eubacterial and Lactobacillus/Streptococcus phylogenetic groups (Cotta et al., 2003).

Although reports of the percentage of viable, culturable antibiotic-resistant bacteria in swine effluent vary, it is clear that antibiotic resistance is a common phenomenon. A study conducted in the 1980s of coliforms in swine waste found that 97% of *E. coli* were resistant to at least one of the following antibiotics: ampicillin, furatrizine, chloramphenicol, kanamycin, streptomycin, sulphonamides or tetracycline (Hanzawa et al., 1984). Haack and Andrews (2000) found that 71% of *Enterococcus faecalis* isolates from swine farrowing house effluent were resistant to tetracycline. Cotta et al. (2003) found that between 4 and 32% of the bacteria in swine manure were resistant to tylosin, depending on the depth from which the sample was collected in the manure holding pits.

Persistence of Bacteria Introduced to Soil

Land application of animal manure, with its high concentration of microbial biomass, is a significant route for the introduction of new bacteria into the terrestrial environment, including potential pathogens (e.g. *E. coli* O157:H7) and some human enteric viruses (e.g. rotavirus). The persistence and transport of these organisms in the environment continues to be a concern for environmental quality and food safety, as well as human and animal health. Gavalchin and Katz (1994) concluded that the longer an antibiotic persists in the soil in an active form, the greater the potential for native soil bacterial populations to be affected. Nutrient amendment via the application of animal waste to soil has been hypothesised to promote faster adaptation of the soil microbial community to antibiotic effects (Schmitt et al. 2005). In addition, biologically active antibiotics (or antibiotic breakdown products) introduced to the soil may confer a selective advantage for soil commensal bacteria carrying resistance genes, or exert selective pressure for acquisition of resistance genes in soil commensal populations.

It has been well documented over the years that many microorganisms survive the transition from effluent pit or lagoon into soil (Kibbey et al., 1978; Chandler et al., 1981; Stoddard et al., 1998; Bolton et al., 1999; Lee and Stotzky, 1999; Jiang et al., 2002; Guan and Holley, 2003; Boes et al., 2005). However, most investigations have focused on pathogens of clinical interest. The length of time that introduced organisms can persist in the soil varies with temperature, moisture, pH and the indigenous community present. The wide range of persistence times of four well-studied pathogens in different environments and at different temperatures has been reported (Table 29.6).

However, a recent study examining the survival of *E. coli* and *Salmonella typhimurium* applied to a clay soil with swine effluent found considerably shorter persistence times (21 days for *E. coli* and 7 days for *Salmonella typhimurium*) (Boes et al., 2005), highlighting the variation in survival times under different environmental conditions. Sengelov et al. (2003) studied the persistence of culturable aerobic, heterotrophic, tetracycline-resistant bacteria in four Danish farm soils following variable rates of pig slurry application. An increase in numbers of resistant bacteria was seen following application, with greater increases occurring in the more heavily manured soils. Five months following application, the proportion of tetracycline-resistant bacteria in all of the treated soils had returned to levels within the range of the non-manured control samples. Andrews et al. (2004) found enterococci declined from 4.8 X 10^5 colony forming units (CFU)/g soil to < 10 CFU/g in soil microcosms over a five-week period. These studies suggest that although a decline in numbers occurs with time, there may be sufficient time and opportunity for mechanisms of resistance selection and gene transfer to occur.

Table 29.6. Persistence times of pathogenic bacteria in different environments.

Environment	Temp (°C)	Estimation of survival time			
		[1]Salmonella	[2]Campylobacter	[3]Yersinia enterocolitica	[4]E. coli 0157:H7
Water	<0 ~5 ~30	~6 mo ~6 mo ~6 mo	<8 wks 1 wk - 4 mo ~4 d	>1 yr >1 yr m ~10 d	>300 d >300 d ~84 d
Soil	<0 ~5 ~30	>6 mo <28 wks ~4 wks	<28 wks ~2 wks ~1 wk	>1 yr >1 yr ~10 d	>300 d ~100 d ~2 d
Slurry		<75 d	<112 d	>28 d	100 d
Dry surfaces		<7 d	~1 d	~1 d	~1 d

References cited:
[1]Guo et al., 2002; Santo Domingo et al., 2000; Mitscherlich and Marth, 1984; Zibilske and Weaver, 1978.
[2]Buswell et al., 1998; Rollins and Colwell, 1986; Mitscherlich and Marth, 1984; Blaser et al. 1980.
[3]Karapinar and Gonul, 1991; Chao et al., 1988.
[4]Wang and Doyle, 1998; Tauxe 1997; Zhao et al. 1995; Cieslak et al., 1993.

Detection of Antibiotic Resistance Genes in the Environment

Accurate and meaningful information on the persistence and dissemination of antibiotic resistance genes in bacteria is of fundamental importance in assessing potential health risks and environmental quality. The detection of specific genes and their bacterial hosts are important components, and recently developed techniques have been applied for detection of specific resistance genes and bacteria in natural environments. In particular, the use of molecular techniques provides rapid, sensitive and specific detection without the requirement for bacterial growth and isolation, which often poses a major challenge given the vast unknown of environmental microbial species. Commonly used molecular microbial techniques are based on unique sequence features of genes to allow detection and identification of microorganisms. Gene probes and the use of polymerase chain reaction (PCR) amplification of nucleic acids is now widely used to enable detection and quantification of low levels of target sequences, and has become a key procedure in the detection and identification of bacteria and genes from a variety of environments including soil, water and faecal material (Josephson et al., 1993; Karch et al., 1995; Wang et al., 1996; Malik et al. 2008). New approaches such as microarray technology have already being developed specifically to detect and identify antimicrobial resistances in clinical and environmental bacteria (Call et al., 2003; Volokhov et al., 2003). A recent study using a gene array approach simultaneously screened for the presence of 23 tetracycline resistance genes and 10 erythromycin resistance genes in soil and faecal samples from swine to find the most prevalent genes (Patterson et al., 2007). Molecular fingerprinting tools and robotic technology have facilitated more accurate and sensitive microbial characterisation of complex environmental samples and have proven to be essential in providing more informative data in environmental monitoring studies. The recent development of a number of probes that target specific antibiotic resistance genes has increased the number of studies investigating the occurrence of these genes in natural environments. Such studies include detection of genes from antibiotic-producing bacteria, as well as genes resident in the background of natural populations. The following section highlights the application of molecular-based methods for detection and quantification of antibiotic resistance genes in bacteria and environmental samples.

Specific classes of antibiotics can be characteristic of the industry in which they are used, and multiple antibiotic resistance phenotype profiles of bacteria have been used to identify sources of faecal pollution (e.g. human, poultry, cattle, swine) in environmental samples (Kaspar et al., 1990; Pillai et al., 1997; Wiggins et al., 1999). Many of these studies focus on bacterial strains of clinical importance and do not fully address the characterisation of populations that have acquired resistance genes in natural environments. To circumvent issues related to cultivation of bacteria, analysis of antibiotic resistance genes can be used to characterise the genetic pool from an environment, with the possibility of tracking the source of faecal contamination in surface waters and groundwater. Similar to the strategy used in microbial diversity studies, the starting point in the design of probes and primers for detection of antibiotic resistance genes is a robust phylogenetic analysis. Specific gene sequences can be targeted for detection, and such an approach has been used to demonstrate the diversity of antibiotic-resistant genes present in swine lagoon and pit effluent. For example, Aminov et al. (2001, 2002) and Chee-Sanford et al. (2001) found that the tetracycline resistance efflux genes (tet B, C, E, H, Y, Z) and the ribosomal protection protein (RPP) genes (tet W, O, Q, M, S, T, B(P), and otr A) were all present in a single swine waste lagoon. Koike et al. (2007) detected tet (M), (O), (Q), (W), (C), (H) and (Z) continually over a three-year period in groundwater underlying two swine farms. Furthermore, tet (W) sequences detected in the groundwater were nearly identical (99.8%) to those found in the corresponding lagoon. In the same study, the application of the same PCR primers further allowed the detection of unique and novel tetracycline resistance gene sequences. Using molecular-based detection, agricultural soils were found to be a rich reservoir of genes closely related to the glycopeptide resistance gene vanA in enterococci (Guardabassi and Agersø, 2006).

Tetracycline resistance genes have been found in large numbers in lagoon effluent. In a study of a cattle feedlot lagoon, a real time PCR method was used to detect and quantify tet (O), (W), and (Q) genes, and correlated gene copy numbers to tetracycline levels (Smith et al., 2004). As noted earlier, effluent loading can have a significant effect on bacterial levels and the upshot of resistance genes. Fields receiving multiple, high-volume (190,000-280,000 liters/hectare) applications of swine effluent each year showed consistently greater diversity and occurrence of tetracycline resistance genes than fields that received moderate volume (90,000-140,000 liters/hectare) applications of effluent on a two- or three-year rotation (S. Maxwell, unpublished). Within field variations also occurred following effluent application, as demonstrated by uneven distribution of tetracycline resistance genes. A recent study showed the persistent effects of manure and the presence of sulphadiazine on soil bacterial communities, where the numbers of culturable resistant bacteria and sulphonamide resistance genes increased (Heuer and Smalla, 2007).

Recent studies have reported isolation of a wide range of antibiotic-resistant bacteria recovered from soil and water environments (Chee-Sanford et al., 2001; Ash et al., 2002; Esiobu et al., 2002; Onan and LaPara, 2003; Dang et al., 2008). A number of soil samples used in these studies were directly exposed to animal waste. Furthermore, sequences of resistance genes detected in bacterial isolates were found to be identical to sequences found in lagoon or animal waste. Nikolakopoulou et al. (2005) screened tetracycline-resistant streptomycete isolates from a range of environmental samples for oxytetracycline resistance genes and found resistance genes in non-tetracycline producing isolates. Identical isolates from the same samples have also been found to host different homologues of tetracycline resistance genes, as well as findings of individual isolates harbouring multiple determinants conferring the same type of resistance mechanism (S. Maxwell, unpublished). It is also noteworthy that cultivation strategies, particularly for populations from environmental samples, have thus far only provided an underestimate of bacteria, suggesting the possibility that a much higher diversity of antibiotic resistant bacteria may exist but has not yet been accounted for. Furthermore, archaea are now thought to be ubiquitous in many soil environments, including agricultural soils (Gattinger et al., 2006; Leininger et al., 2006). Far less is known about soil archaea and the extent of their resistance mechanisms or their contributions to genetic exchange within the soil metagenome.

Conclusions

The impacts resulting from agricultural use of antibiotics and the practice of land application of animal waste on environmental quality and health risk potential is not completely clear, albeit there are demonstrated links to increased and accelerated incidences of antibiotic resistant bacteria. However, the phenomenon is not a simple relationship of cause and effect. What is evident is the myriad complexity of antibiotic and biological mechanisms, and the ecological interactions that can occur at numerous points along the course of antibiotic use and disposal of livestock waste in soil environments, beginning with entry of antibiotics into animal gut systems. Regulatory aspects related to continued use of land application for waste management in animal production have real current concerns for nutrient (nitrogen, phosphorus) loads in soil; the practical impact of loading antibiotic residues and resistance genes is not yet known. The collective examination of specific mechanisms that affect the fate of compounds, microorganisms and the genetic pool will provide a better understanding of the true impacts of land application of effluent, as well as the general nature of the microbial and molecular ecology of antibiotic resistance.

Field information on the fate and transport of antibiotics is still limited, but in general, low amounts have been detected in soil and water environments, including the presence of breakdown metabolites. Predictive measures for solute and bacteria transport in soil and water have relied on existing models, which do not adequately predict contamination, and indicate a clear need for a larger database to develop and better inform models. The physicochemical characteristics of the soil environment are likely to influence compound persistence, bacterial survival and genetic mechanisms at work. Trace amounts of antibiotics or other compounds (e.g. heavy metals) could act as selection pressures for maintenance and (co-) transfer of antibiotic resistance genes.

While the half-lives of antibiotics in manure are relatively short, it remains possible that drug residues may exert effects on biological functions within bacterial populations present in soils. Studies have shown application of animal manure to soil can readily lead to groundwater contamination with faecal bacteria. The acquisition of antibiotic resistances, however, appears to span a diverse phylogenetic range of bacteria, including those native to soil and water environments. Phylogenetic analyses of genes involved in tetracycline and erythromycin resistance demonstrate the evolution of these genes over time, and suggest that obtaining resistance genes from antibiotic-producing bacteria is not a major mechanism of resistance acquisition evident in a broad range of bacteria. Resistance genes have been maintained in bacteria prior to the modern antibiotic era, even though the origin and purpose of these genes is not yet clear. The exact mechanisms contributing to antibiotic resistance gene acquisition and maintenance in natural environments are not yet well established, although increasing numbers of studies support lateral gene transfer events. Acquisition of antibiotic resistance through mechanisms of selective mutations and lateral gene transfer may be acting in concert with other natural mechanisms of genetic adaptation among a diverse range of bacteria in natural soil and water environments.

Part G

Anthropogenic Contamination of Ecosystems

Consequences and Actions

Authors: Anne Fairbrother, Stephen Garbaciak, Leif Norrgren
and Kishore Rajagopalan

Coordinating Author: Anne Fairbrother

Prevention and Reduction of Chemical Contamination on Ecosystems

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Anne Fairbrother
Exponent, Seattle, WA, USA

Introduction

In the United States, the Resource Conservation and Recovery Act (RCRA; PL# 94-580), enacted in 1976, requires that all solid waste (i.e. normal garbage) and hazardous waste be disposed of in licensed landfills. Similarly, the European Landfill Directive (Council Directive 99/31/EC) which came into force on 16.07.1999 (http://ec.europa.eu/environment/waste/landfill_index.htm) governs how and where waste disposal may occur. These laws and their associated regulations require that landfills have impervious liners, mechanisms for collecting leachate and gases, and other similar controls to reduce off-site contamination. However, old landfills, previously known as dumpsites, continue to pollute the surrounding environment, generally through the groundwater although in some instances contaminated dust can be a major source of pollution. It has been shown that once a landfill is saturated, annual precipitation of approximately 90 cm per year can percolate 3.7 million litres of contaminated water per acre (Salvato et al., 1971). Contaminants in the leachate include NAPLs (non-aqueous phase liquids), VOCs (volatile organic compounds), s-VOCs (semi-volatile organic compounds), OCs (organochlorines), other pesticides, and metals/metalloids. Leachates are complex mixtures of compounds from all these classes, each of which has compound-specific soil migration rates, complexation properties, biodegradation potential and toxicity. Thus, determination of the comparative risks of leachate components reaching surface waters and/or drinking water wells becomes a very complex task, and includes site monitoring, transport modelling and comparative toxicology. Specific points to consider when assessing the risk of a landfill to humans or the aquatic environment include (Kurian et al., 2005):

- mass rate of release of waterborne and airborne pollutants
- aerial extent and concentration of contamination through groundwater plumes or via air transport and deposition
- total time over which pollutant release will occur (or has occurred)
- characteristics of the site such as the size and depth of solid waste and degree of compaction
- classes of pollutants accepted by the landfill (if known)
- characteristics of the soils and groundwater underneath and adjacent to the dumpsite
- persistence and transformation of the pollutants and their transformation products
- biomagnification potential of the pollutants
- relative toxicity of the landfill constituents
- synergistic and antagonistic impacts of other pollutant releases or adverse health conditions that might cause an exposed population to be more (or less) susceptible to pollutants derived from the site.

Site Monitoring

Monitoring begins with a site inspection and analysis of available historical information. Information about the types of materials that had been put into the landfill is particularly important, although frequently unavailable for old landfills (Kjeldsen, 1993). All information about pollutants should be considered only a partial list until verified either by direct sampling or definitive records. The size of the site and its location in relation to other areas of human use should be determined and mapped. Features such as nearby water bodies (lakes, streams, rivers) and wetlands should be mapped as well and a general survey of surface vegetation should be completed. Groundwater maps and other similar data about direction and flow of groundwater should be gathered. This may include analysis of logs of public or private wells in the area (including location, depth to water, and flow rates). In addition, an understanding of the types of soils under and around the dumpsite will aid in determining the potential direction of groundwater flow, as well as in possible surface permeability and vegetative cover. Chemicals will move through various soils at different speeds, depending upon their class (volatile organics/organochlorines/metals, etc.) and their inherent biodegradation capacity.

Old dumpsites generally were not lined, so leachate will move off the site through a wide underground plume. Newer dumpsites and landfills are lined, so the plume will likely be narrower (a ‘finger’ plume) as it escapes through holes in the liner (Lee and Jones-Lee, 1994). It is important to distinguish between these two types of landfill construction, as the potential width of the plume will affect the sampling design used to collect and monitor groundwater for contaminants. This can be done through drilling monitoring wells and/or placement of piezometers. Hydraulic pressure and direction of flow will enable modelling of the size of the plume, the direction in which it is moving, and the rate of movement. This, in turn, will allow predictions of when the contamination is likely to encounter surface water. At this point, the hydraulic pressure of the surface water body must be included in the calculations (e.g. whether it is a losing or gaining reach of a stream relative to the pressure of the plume will determine the penetration rate of the groundwater into the stream).

Concentration of the chemicals of potential concern (COPCs) in the groundwater can be measured on samples taken from the monitoring wells. Various *in situ* collection devices are available, including (but not limited to) positively and negatively charged ion resins that will bind to most organic substances and metals. These can be left in place for varying lengths of time and then removed to determine concentrations of the COPCs in the groundwater. Combining concentration with flow rate will allow calculation of the total load of chemicals and how much will enter a surface water body or groundwater aquifer per unit time. If the flow rate of the surface water body is known, then concentrations of the COPCs in the surface water can be predicted (in the case where the plume has not yet reached the water body).

Because chemical analyses are expensive, it is desirable to reduce the list of initial COPCs as soon as possible. This is done initially through reviewing old records, if available, of what was put into the landfill. If no information is available, a small number of appropriately chosen random samples can be analysed for a full suite of contaminants, and the remainder of the samples analysed only for those with concentrations above levels of concern (e.g. above drinking water standards). Any COPCs that are not detected in any of the samples can be eliminated from further consideration (assuming sample detection limits are at or below benchmark toxicity values). The samples should be collected from as close to the landfill as possible to ensure that the highest concentrations will be present. The number of samples will depend upon the size of the landfill and suspected plume of contaminated groundwater, but should number at least five samples. All the general classes of compounds should be part of this initial test, including: organochlorines (pesticides plus industrial chemicals such as PCBs), VOCs, sVOCs, PAHs, and metals. PCBs can be analysed as ‘total PCBs’ or mixtures (e.g. Arochlors), rather than specific congeners. Dioxins and furans are very expensive analyses and can be deferred until after the organochlorine screen is returned. If no organochlorines are detected, then the dioxin/furans do not need to be assessed.

Once the list of chemicals has been established, groundwater monitoring can be conducted. First, an estimation of the size of the plume is accomplished by drilling sampling wells at what is thought to be the periphery of

European Regulation System for Chemicals

In 2003, a framework was established for a new European regulatory system for chemicals called REACH (Registration, Evaluation, Authorisation and restrictions of CHemicals), and the revised version is referred to as the REACH or the REACH system.
One of the major purposes of REACH is to certify a high level of protection of human health and the environment. With respect to risk assessment, the objectives of REACH can be reviewed as two overall goals:

1. Increasing the knowledge about the properties and uses of separate chemical substances.
2. Increasing the speed and efficiency of the risk assessment process, and making importers and producers of chemicals responsible for this procedure.

All general industrial chemicals are regulated in a single system in REACH, preventing the previous large difference in test requirements between 'new' and 'existing' substances. Each chemical's production volume is the general criterion for priority setting, the higher the production volume, the more extensive test batteries are applicable and the assumption is also that the potential of exposure and therefore the risk of adverse effects is dependent on the production volume.
All chemicals produced in amounts of 1 tonne or more per year must be registered in a central database, and this implies that substances that are not registered are not allowed to be manufactured or imported into the EU. The authorities evaluate the registration, the registration dossier is checked for completeness and the quality of the industry's testing proposals is examined.
The industry is required to make a preliminary risk assessment, a chemical safety assessment, as a way to improve the efficiency of the risk assessment, one of the major aims of REACH.
The chemical safety assessment should contain the following parts, based on the information contained in the technical dossier:

1. Human health hazard assessment
2. Human health hazard assessment of physicochemical properties
3. Environmental hazard assessment
4. PBT (Persistent, Bioaccumulating and Toxic) and vPvB (very Persistent and very Bioaccumulating) assessment

If the manufacturer or importer concludes that the substance or preparation meets the criteria for classification as dangerous as a result of steps 1 to 4, the following steps should also be considered:

5. Exposure assessment
6. Risk characterisation

The authorisation procedure is applicable to substances that are:

1. Classified as carcinogenic, mutagenic or toxic to reproduction (CMR)
2. Persistent, bioaccumulating and toxic (PBT)
3. Very persistent and very bioaccumulating (vPvB)
4. Endocrine disrupting chemicals (ED)
5. Causing other serious and irreversible effects to humans or the environment identified on a case-by-case basis.

The authorisation requirements to meet the REACH criteria refer to an estimate of the chemical's half-life, the bioconcentration factor for estimating bioaccumulation and long-term aquatic toxicity, or CMR data, or evidence of chronic toxicity.
If the risks to human health are adequately controlled or the socio-economic benefits of using the substance outweigh the risks to human health and/or the environment with no suitable alternative substances, authorisation is granted.
Restriction procedures can be instigated for any substance if its use poses an 'unacceptable risk to human health or the environment', although no criteria for what is considered 'unacceptable risk' are given in the regulations.

Leif Norrgren

Table 30.1. Required data and tests.

Production volume (per year)			
1-10 tonnes	10-100 tonnes	100-1,000 tonnes	Over 1,000 tonnes
• Physico-chemical properties • In vivo skin sensitisation • In vitro test for gene mutations in bacteria • Acute toxicity to Daphnia	Additional tests (apart from them required for 1-10 tonnes) required are: • In vivo skin and eye irritation • Two in vitro cytogenicity/ mutagenicity tests using mammalian cells • Acute mammalian toxicity study • a 28-day mammalian toxicity study • Screening for reproductive toxicity • Acute toxicity to algae, fish and microorganisms • Data on biotic degradation and hydrolysis, and an adsorption/desorption screening study	Additional tests (apart from them required for 1-100 tonnes) required are: • Data on fate and behaviour • Long-term toxicity to fish and Daphnia • Fish reproduction • Sub-chronic toxicity to mammals • Developmental toxicity • Two-generation reproductive toxicity study.	Additional tests (apart from them required for 1-1,000 tonnes) required are: • Long-term effect data on sediment living organisms, earthworms, soil-invertebrates and higher plants • Additional data on fish reproduction

the plume. After characterising the size and shape of the plume, a sampling grid is established to delineate gradients and monitor the rate of movement of the chemical(s) in the plume. The number of wells and grid density is determined by available budget, size of the plume and need for detailed information (e.g. proximity to human habitation or drinking water wells). This information is then used to support transport models that predict the fate (e.g. degradation) and movement of the various chemical classes.

Transport Models

The evaluation of pollutant transport requires a determination of the distribution of the chemicals among water, particulate or vapour phases within each environmental compartment (air, water and soil), as well as the movement (i.e. the transport) of each of these phases within and among the various compartments (Mackay and Mackay, 2007). Including all compartments in every model is not necessary, and most landfill studies will focus primarily on the soil/groundwater compartment.

There are many models available for estimating pollutant transport from a point source such as an abandoned landfill. The USEPA 3MRA model is one of the most complex and consists of a series of transport models representing all media compartments within a system framework (US EPA, 2003). The UK Environment Agency's LandSim (http://www.landsim.co.uk) is an interactive programme that tracks leachate production, chemistry, migration and leakage through engineered and non-engineered structures, followed by leachate migration through the unsaturated zone to assess the ultimate impact on the groundwater aquifer. More simple models for estimating only groundwater dispersion are available (US EPA. 2011a). Selection of the model to use will depend upon modeller's preference, types of contaminants of concern and desired precision. Model selection should be based on the primary pollutants of interest, the amount of time and money available, and the required precision in estimating when, where and at what concentration the pollutants will intercept drinking water aquifers or surface water bodies. Degradation of organic pollutants or ageing and speciation of metals should be considered in all cases, as this will significantly increase model accuracy.

Comparative Toxicology

Landfills with highly hazardous pollutants should take priority over those with less hazardous substances. Determination of relative hazards can be carried out in one of two ways: 1) bioassays of the pollutant mixture or 2) characterisation and hazard ranking of each of the constituents.

Bioassays measure some biological response to estimate the relative effect of a mixture. These may be followed by toxicity identification and evaluation (TIE) should there be a need to identify which of the constituents in the mixture are most responsible for the observed toxic response (US EPA, 2007). For landfill leachates, common bioassays include luminescent bacteria (*Vibrio fischeri*), algae (*Selenastrum capricornutum*), and a crustacean (*Daphnia magna*). Genetic toxicity tests using the umuC gene expression test from *Salmonella typhimurium* are used occasionally but generally not in a screening mode (Baun et al., 2000). The relative hazard of the leachate can be determined by the number of organisms responding and the strength of their response (e.g. the amount of dilution of the leachate that represents where responses are first measured).

The alternative approach is to list all the chemicals that are present in the leachate and rank order them by their aquatic toxicity or human health hazard benchmarks. Comparative rankings of the relative hazard among landfill leachates can be used to prioritise clean-up. The difficulty with this approach is that some chemicals in landfill leachates do not have toxicity benchmarks and interactive effects of the chemicals are not taken into account. Furthermore, it requires much more detailed and extensive analytical chemistry which is generally more expensive than conducting the above-mentioned bioassays. Therefore, it is recommended that the bioassay approach be given preference.

Exposure Assessment

Because risk is a function of both hazard and exposure, the final step in assessing potential risks of abandoned landfills is to determine the probable level of exposure to humans and the environment. This will depend upon 1) initial concentrations of materials at the landfill; 2) rate of movement into the groundwater; 3) distance from the source where people or the environment will be exposed; and 4) attenuation of the groundwater plume as it moves off-site. The assessment endpoint is the concentration of the pollutants in drinking water, surface water, or indoor air. Using the above-referenced models, the loading of the chemicals of concern into these media can be determined. The volume of water in the wells and/or surface water body will need to be known to calculate the final concentration of the pollutants (i.e. loading X volume = concentration). The concentration of the pollutants in the water or air (e.g. µg/L) is the amount of exposure to which a person or plant/animal will be exposed and is the same unit used for expressing hazard (e.g. acceptable drinking water concentrations).

Risk Characterisation

The end result of exposure modelling and estimation plus the bioassay information is an estimate of potential risk for people and the surrounding aquatic environments. The risk should be characterised in terms of magnitude and the probability of occurrence. This should include a description of the spatial extent of the groundwater plume and its probable path for continued migration outward. A time-frame for how long it will take for the leading edge of the plume to intercept drinking water wells or surface water bodies should be included in the risk description. Any uncertainties with these estimates should be stated (e.g. the interception is likely to occur in 10 years, plus or minus 2 years). In addition, it is helpful to describe the type of effects that might occur should the predicted exposure actually happen, such as a reproductive risk or a cancer risk. The possible risk level should be characterised as 'high,' 'medium,' or 'low' depending upon how long it will take for exposure to occur, the expected exposure concentrations, and the intrinsic hazards of the chemicals of concern.

Human Health Risks

Risks to people generally occur from drinking contaminated groundwater. The contamination plume from the landfill may intercept drinking water wells that are off-site. Wells can be tested to determine whether unsafe levels of chemicals are present (e.g. by comparing chemical concentrations with drinking water standards) (US EPA, 2011b and European Commission, 1998). Predictions of future risk should be made by modelling the plume dispersion and migration rates and predicting when (if ever) it will intercept drinking water wells; estimates of probable concentrations can also be made although there will be considerable uncertainty associated with them. Contaminants of particular concern are heavy metals (e.g. lead and mercury), and chlorinated organics (e.g. dioxins and PCBs).

Additional consideration should be given to the possibility that the contaminated groundwater plume will migrate under buildings. If this occurs, then there may be risks associated with intrusion of vapours from volatile and semi-volatile organic compounds into the buildings. The potential for such intrusion can be modelled using the Johnson and Ettinger (1991) model (See http://www.epa.gov/oswer/riskassessment/airmodel/johnson_ettinger.htm for recent modifications and detailed descriptions. See also http://www.epa.gov/Athens/learn2model/parttwo/onsite/JnE_lite.htm for available on-line tools to run the model). Air concentrations are then compared to inhalation benchmark values (e.g. the USEPA Integrated Risk Information System, IRIS) (US EPA, 2011c).

Environmental Risks

The most common environmental risk associated with abandoned landfills is contamination of surface water (lakes, streams, rivers). This occurs when the contaminated groundwater intercepts a surface water body. Estimates of the time to interception and the resulting concentration in the receiving water body can be modelled. The resulting concentrations are then compared with ambient water quality criteria for the protection of aquatic life (US EPA, 2010 and European Commission, 1998) to estimate risk. Metals (e.g. cadmium, copper and silver) and certain organic materials (e.g. PCBs and PAHs) are of particular concern to aquatic organisms.

Ranking of Risks at Landfills

Comparison and ranking of risks from various landfills can be accomplished through a simple comparison of the risk level and/or on the basis of time-to-effect or relative hazard of contaminants. A site where the groundwater plume already has intercepted drinking water wells or important water bodies should take priority. If there are several in this category, those with the most hazardous contaminants should be considered for immediate clean-up, particularly if there are volatile organic compounds resulting in vapour intrusion into buildings. Second tier consideration should be given to those sites where the groundwater plume is likely to intercept drinking water wells, dwellings, or important water bodies within 5-10 years. Prioritising landfills for clean-up should not be done strictly on the basis of which pollutants are present. Amount (e.g. concentrations in drinking water) and potential for exposure to humans or valued ecological resources (e.g. density of population in the area or proximity to a highly productive stream) are necessary parts of the risk equation and should figure prominently in the priority ranking.

Minimising Industrial Wastes from the Fabricated Metal Products Industries

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CASE STUDY USA

Kishore Rajagopalan
University of Illinois at Urbana Champaign, Champaign, IL, USA

Introduction

The fabricated metal products and machinery sector in Illinois employs over 80,000 people. The sector includes farm machinery, packaging machinery and construction machinery. Illinois also has many highly specialized niches within the fabricated metals sector in areas such as bolts, nuts, screw and rivet, spring, and crown and closure manufacturing. Additionally, there are a number of specialized small/machine fabrication shops that cater to automotive industry Tier I and Tier II suppliers. Collectively, these areas represent a dominant sector of the Illinois economy and form part of the Advanced Manufacturing Cluster, a strategic sector targeted for growth by the State of Illinois. Contrary to popular belief, these old economy industries are by no means low-tech industries. Instead, they have sought to foster and embrace innovations to maintain global competitiveness. Even more noteworthy is that these industries form the bedrock on which the new economy industries such as telecommunications and biotechnology are anchored. Examples of the latter are the critical role micro-machined components and dedicated machinery such as gene sequencers play in the "new economy" sectors.

The fabricated metal products industries sector engages in the forming/shaping of metals and their surface preparation. While a large variety of alloys are processed in these facilities, they can be broadly categorized as ferrous and non-ferrous. Surface preparation includes a myriad of operations that range from electroplating to powder coating. The primary markets served by this sector include automotive, electronics, aerospace and consumer durables among others (US EPA, 1995).

Metals usually are shipped in bulk as sheets, rods, wires and tubes. A smaller fraction is shipped as castings. Once received at the manufacturing facility, these materials are further shaped. Examples of shaping processes include shearing, punching, stamping, and coining among others. While the details of the processes vary (SME, 1984), they enable the transformation of the bulk metal to a predetermined shape. Castings differ as the metal is shaped to closely approximate the final form through molding of molten metal. In a number of plants, several such forming operations may be necessary to produce a final product. In addition to these operations, many processes categorized as machining are also frequently part of the manufacturing process. These include drilling, milling, turning and the like. These processes accomplish their objective through precise removal of metal and are termed metal removal processes.

In addition to basic forming/shaping processes, the surfaces of the metal parts are frequently manipulated in

Table 31.1. Process inputs and wastes. Source: US EPA, 1995.

Process	Material input	Air emission	Process wastewater	Solid waste
		Metal shaping		
Metal cutting/or forming	Cutting oils, degreasing and cleaning solvents, acids, alkali and heavy metals	Solvent wastes	Waste oils, spent acid, alkaline and solvent wastes	Metal chips, solvent still bottom wastes, metal bearing cutting fluid sludge
		Surface preparation		
Solvent degreasing, alkaline and acid cleaning	Solvents, alkali, acids	Solvents	Solvent, alkaline and acid waste	Ignitable wastes, solvent wastes, still bottoms
		Surface preparation		
Anodizing	Acids	Metal-ion bearing mists and acid mists	Acid wastes	Wastewater treatment sludge
Chemical conversion coating	Metals and acids	Metal-ion bearing mists and acid mists	Metal salts, acid and base wastes	Spent solutions, wastewater treatment sludge
Electroplating	Acid/alkaline solutions, heavy metal bearing solutions, cyanide bearing solutions	Metal-ion bearing mists and acid mists	Acid/alkaline cyanide and metal wastes	Metal and reactive wastes
Painting	Solvents and paints	Solvents	Solvent wastes	Still bottoms, sludge, paint solvents

various ways to enhance functional properties. One common operation is painting. Other examples include electroplating, electroless plating, anodizing and phosphating. These surface treatment operations frequently require the surfaces themselves to be clean prior to application- or adhesion can be impaired and functionality compromised. Solvents, alkali, and acids are used in immersion, spray, or pickling processes to achieve cleanliness.

The combination of operations enumerated above and the use of energy in the form of compressed air, electrical motors, hot air, hot water/steam and the like result in gaseous, liquid, and solid effluents. It is the object of this chapter to illustrate with a few examples some ways to minimize the generation of these wastes.

Common Industrial Wastes in the Fabricated Metal Product Industries

The most common wastes arising from the fabricated metal products sector are metal, paint, electroplating sludge, sludge from various processes, acids, alkali, used industrial fluids, volatile organic compounds, water, and combustion products such as CO_2. The common processes and the wastes associated with them are shown in Table 31.1 (US EPA, 1995).

Minimizing Waste

A useful tool for achieving waste minimization is to recognize that every situation can be reduced to four elements: people, process, management controls, and externalities (Figure 31.1).

This, of course, is a variation on the Materials, Machines, People, and Methods approach used in the fish bone cause and effect diagrams in problem solving. However, the terminology used in Figure 31.1 is more descriptive of the issues identified by our interactions with our clients over a decade and will be used instead.

The major categories contributing to waste as identified in Figure 31.1 must be disaggregated to provide a finer degree of detail. More importantly, the degree of detail should be no finer than required to promote efficient problem solving.

A brief definition of the terms used in Figure 1 is provided to familiarize the reader with the scope intended with their use.

- *Waste*: As used in this document and consistent with its focus, waste will primarily refer to materials (including water and energy) lost to the environment.

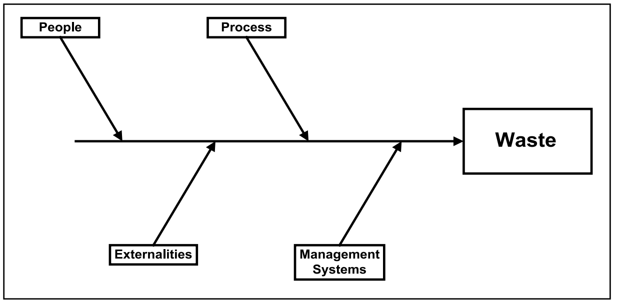

Figure 31.1. Waste minimization tool. Source: Author.

- *Process*: Pertains to the materials and mode of manufacture used to transform raw materials to final products capable of achieving predetermined functional and aesthetic attributes
- *Management System*: All aspects of the system that direct and/or support the transformation of raw materials to final products. A nonexclusive list is management organization and style, accounting systems, logistics, purchasing culture, and interface with the supplier base.
- *People*: Those directly involved in the operation of the machinery or process that transforms raw materials to products.
- *Externalities*: These refer to constraints imposed on the industry by external parties. Typically these are regulations that govern environmental discharges, health and safety conditions, customer-mandated manufacturing processes etc.

In the following section, we illustrate with hypothetical examples ways of leveraging the above to minimize waste.

Illustrative Examples of Waste Minimization

Leveraging Management Systems to Minimize Waste Metalworking Fluids

Metalworking fluids are used in most metal cutting operations to provide cooling, lubrication, corrosion protection and chip removal. The correct choice and application of metalworking fluids is critical for modern high-speed machining operations. In the past, most metalworking fluids were oil based. However, in the U.S.A most metalworking fluids are oil/water emulsions, a use motivated in part by safety but also due to the ability to combine both cooling and lubrication in a single fluid. These advantages are balanced by the increased need for close monitoring and chemical expertise for preventive maintenance. For example, the contamination of these fluids by bacteria can cause selective degradation of important functional components of the fluid formulation such as corrosion inhibitors. Other examples include the accumulation of ions such as calcium and magnesium leading to emulsion destabilization and loss of lubrication. The increased complexity in the management and maintenance of these water based metalworking fluids leads to both increased waste volume and increased manufacturing costs. It has been estimated that for every dollar spent on metalworking fluid purchases, $1.5-5.5 are spent on managing, maintaining, and disposal (Bierma and Waterstraat, 2004).

Problem Recognition and Root Cause Analysis

Given the above as background, let us turn our attention to a situation often encountered in a fabrication facility. In this hypothetical example, a plant has identified increased volume of metal working waste as an issue. Discussions with plant personnel reveal that fluid breakdown and oil contamination lead to frequent disposal. The problem of oil contamination is traced to leaking hydraulic oil valves and fluid breakdown caused by use of water of high hardness. Both are examples of process related issues. Fluid breakdown and oil contamination are also recognized as facilitating increased bacterial growth and associated problems such as dermatitis. Moreover, it is also discovered that a few central sumps supply many of the machining centers. A few of the machining centers are sources of regulated heavy metals. In an effort to keep these metals from exceeding thresholds triggering hazardous waste classification, and to combat worker complaints related to dermatitis, the plant had adopted a strategy of pumping the fluids out on schedule rather than performance. Both the dermatitis and the hazardous waste limits are examples of externalities encountered in manufacturing plants. Further discussions with operators reveal that fluids are mixed at improper concentrations, rarely monitored for concentration, and the central sumps

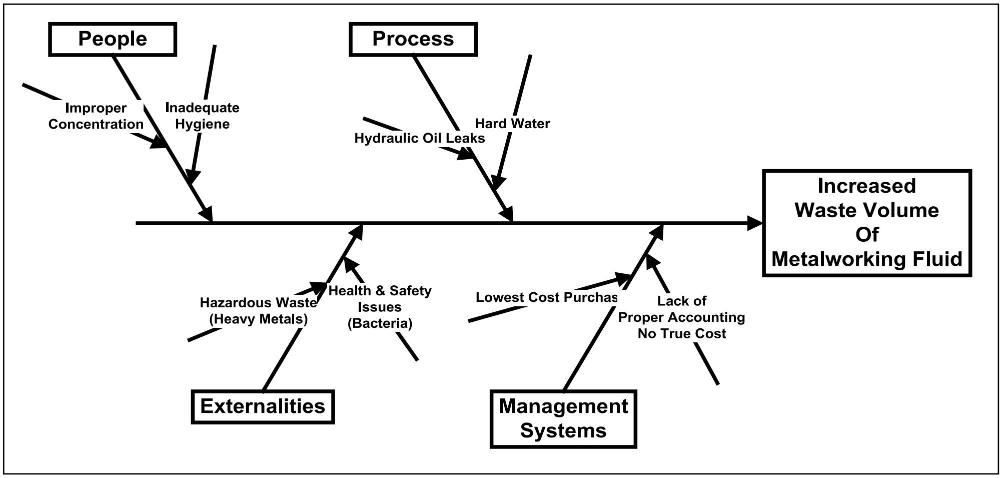

Figure 31.2. Example application of waste minimizing tool. Source: Author.

are viewed as convenient reservoirs for plant trash. All of these are examples of lack of awareness on the part of the operators about the interconnectedness between the metalworking fluid sumps and product quality. These are examples of people related issues. Finally, a review of the management systems reveals that purchases are made on a per gallon or pound basis, often on first price alone. No comprehensive cost accounting practices are in place. These are examples of management system problems.

Solution

Having identified many of the important causes (Figure 31.2) contributing to the waste issue, the plant realized the need for a comprehensive approach to improve the situation.

The plant lacked the knowledge and technical resources to comprehensively address these issues on its own, although it was recognized that the supplier of the chemicals had the resources and the knowledge to comprehensively address all of the identified issues (Figure 31.3). However, the plant personnel had doubts about turning over this portion of the operation to the chemical supplier. Opposition was particularly vocal from the purchasing department as they believed this would increase costs. To address this concern, the plant decided to restructure the relationship between the plant and the supplier from an adversarial one to one of shared partnership. This was accomplished by a contractual shift that

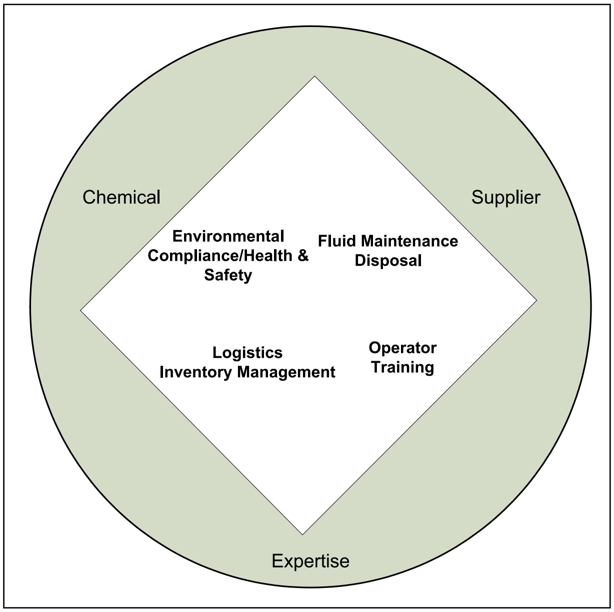

Figure 31.3. Chart showing areas of chemical supplier expertise. Source: Author.

eliminated the purchasing of fluid to one that purchased "fluid performance." Under the new contract, the supplier was paid a flat fee for supplying "metalworking fluid performance" based on historical costs. Furthermore, the plant also negotiated a targeted reduction in costs for subsequent years. Under the new contractual agreement, the chemical supplier "owned" the fluids and did not get paid on a per gallon basis. The supplier stood to make additional profits through reducing chemical use within the plant while still providing performance. This contractual agreement aligned the interests of both the plant and the supplier and allowed the plant to leverage the expertise of the supplier.

The scenario sketched out above is not a flight-of-fancy. It has been tried successfully at several large corporations and is called Chemical Management Systems (Table 31.2). Some examples of successful implementation in Illinois are provided in Bierma and Waterstraat (1997).

The above example of aligning supplier interests with facility interests is one of many options available to leverage management systems to minimize waste. Others include adopting frameworks for holistic planning and decision-making, instituting transparent and well thought-out Total Management Accounting systems that pinpoint process inefficiencies, and encouraging a culture of constant improvement and personnel development.

Leveraging Process Improvements – An Electroplating Example

Electroplating, the process of depositing metal over a substrate through the use of electricity, is a commonly used technique to obtain both functional and decorative finishes. An example of a functional finish is chromium deposition for corrosion resistance. Achieving an antique brass finish on a steel part is an example of a decorative finish.

Quality coatings can only be achieved if the parts are cleaned thoroughly prior to being plated. Cleaning is accomplished using alkaline solutions formulated with surfactants to achieve removal of tenacious oil films and other types of soils. Following this, the surfaces are rinsed and then processed through acid solutions to remove oxide scales. Residual acid is removed by rinsing. Finally, the parts are plated, rinsed and subjected to further processing as desired. Electroplating can be done from both alkaline and acidic solutions.

The movement of parts through the sequence of steps results in carry-over of solutions from the chemical tanks to the rinse tanks. The carry-over causes contamination of the rinse tanks with acids and alkali. Rinsing ability is compromised as contaminants accumulate in rinse tanks. To counter the build-up of contaminants, the rinse water is bled either periodically or continuously and replaced with fresh water. As the rinse tanks contain hazardous metals and metal complexing agents such as cyanide, they require treatment prior to discharge. These treatments frequently are pH adjustment followed by heavy metal precipitation. Treatment for hexavalent chromium tends to be more complicated. In all these cases, the sludge formed is deemed hazardous and disposal is expensive.

Problem Identification and Root Cause Analysis

In the example, we illustrate the utility of leveraging process improvements in an electroplating shop (Figure 31.4).

An electroplating job-shop in Chicago, IL was experiencing difficulties with higher cost of electroplating

Table 31.2. Examples of Chemical Management Systems (Bierma and Waterstaat, 1997).

Plant	What is managed?	Contract	Resultant benefits
Navistar International, Melrose Park, Illinois	Metalworking fluids, cleaners	Fixed fee staffing level/staffing fees	Metalworking fluid usage reduced by 50%; Waste haulage reduced by 90% Reduced production downtime; Improved product quality reduced engine block and head rework 93%
Ford Assembly Plant Chicago, IL	All fluids except paints, sealers, adhesives All solvents	Fee/vehicle for solvent; Fee/vehicle for other chemicals; Fixed annual fee for chemicals unrelated to production	VOC (volatile organic emission) reduced by 57% Wastewater sludge reduced by 27%
Chrysler, Assembly Plant, Belvidere, IL	All chemicals for cleaning, treating, and coating autobody	Fixed fee/vehicle	Over $1 million in first year; dramatic reductions in VOC, improved product quality, improved health and safety

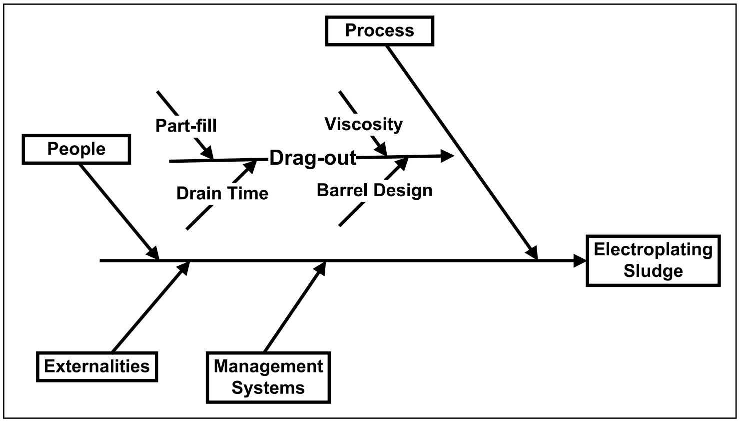

Figure 31.4. Electroplating shop example of the waste management tool. Source: Author.

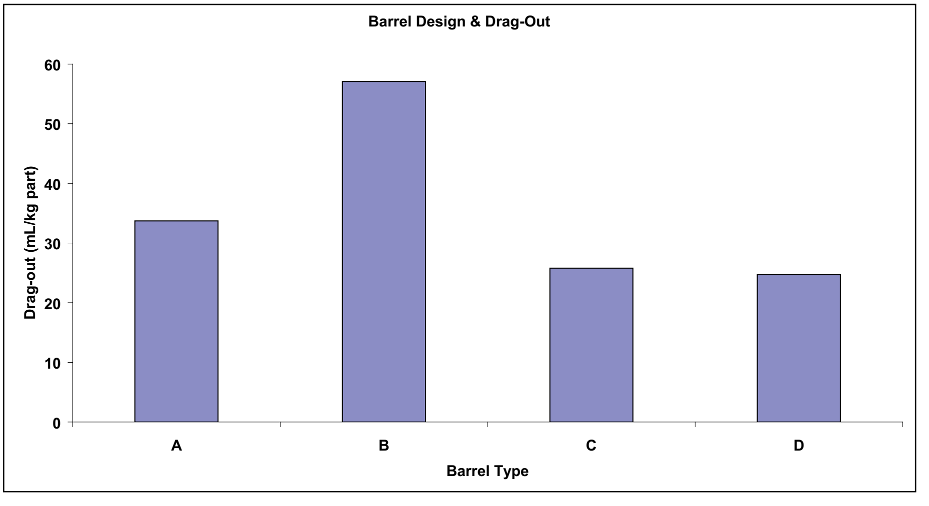

Figure 31.5. Results of barrel design test (CMFI, 2002).

sludge disposal. The shop specialized in plating fasteners and other small parts. These are typically barrel-plated. A root-cause analysis of the situation identified "drag-out" as a leading contributor to the stated problem. Additional queries led to identifying various other factors as secondary contributors: barrel design, viscosity of solution, part-fill quantity, drain-time among others.

Solution

Suggestions to improve the situation included: reducing plating bath solution concentration and increasing temperature to reduce viscosity, increasing drain-time to allow adequate drainage, and reducing unnecessary hold-up through careful part-filling. These were implemented and found to be successful in improving the situation. The remaining question was whether the barrel design could be improved to reduce drag-out even further. To test this, four barrel designs were evaluated with the results shown in Figure 31.5.

Compared to the barrel that was being used (Type A), barrels C and D resulted in 25% reduction in carry-over. The estimated reduction for that facility was 11,000 Liters of various solutions used in the process.

Leveraging process systems requires fine-grained knowledge about the underlying physics of the process. Such knowledge can be scarce in many instances. Even when it exists it is scattered across several disciplines in the literature. Falling within the realm of applied research, it is also found more in specialized trade literature than in the more easily accessible peer reviewed scientific literature. Again, suppliers can be surprisingly knowledgeable about some of these systems and should be consulted when solving such problems.

Leveraging Externalities

In this example, we illustrate how externalities such as environmental regulations can influence process waste.

A company manufactures several parts, including an automotive component. The relationship with the auto manufacturer is considered important for strategic reasons and cultivated zealously. Initially 100% of the parts processed are ferrous and typically go through the same processing steps including phosphating. No chromium or cyanides are used in the process and the process typically is well controlled. The wastewater from the facility is treated in a central facility and the metal precipitated. The sludge is dewatered and shipped to another facility where it is converted to phosphate glass.

The automotive company requests that the manufacturer process aluminum parts in an effort to reduce the weight of the final vehicle. However, this caused a regulatory dilemma to the manufacturer. Under USEPA regulations, the sludge produced from the phosphating of aluminum is designated a hazardous waste. Ironically, USEPA's original intent was to regulate outdated processing practices that utilized chromium and cyanide solutions; a practice not in use at the manufacturer. Furthermore, contamination of a large quantity of non-hazardous waste sludge with a small amount of a classified hazardous waste would require the entire waste sludge material to be classified as hazardous and disposed of accordingly. This of course would increase disposal costs, compliance activity, and create potential legal liability.

Solution

Under these circumstances, the manufacturer had two options. One was to segregate the aluminum phosphating line from the rest and treat it separately and deal with the resulting sludge as a hazardous waste. The second was to characterize the aluminum sludge for hazardous characteristics, prove its nonhazardous nature, and petition the government for delisting or relief from the law. The first option creates an extra cost. The second option involves both cost and time. In this example, a combination of the first and second option was the preferable approach to achieve waste minimization. Combining the two sludges would have prevented the beneficial reuse of the ferrous sludge material.

Examples such as this that highlight the unintended consequences of well-intentioned laws are not uncommon but do pose a problem for manufacturers and in some cases can increase waste generation.

Conclusions

The Illinois Sustainable Technology Center (ISTC) has over the past twenty years worked with numerous metal fabricators and machinery manufacturers within Illinois assisting them with reducing manufacturing costs through greater process efficiency. These efforts have helped reduce use of fresh water, manufacturing energy intensity, volatile organic emissions in the Greater Chicago Metropolitan area, heavy metals, and nutrients release to Illinois rivers and Lake Michigan, hazardous wastes and sludges to Illinois landfills, and increased worker health and safety through encouraging adoption of green processes in manufacturing. Select examples of success can be found in the various annual reports of the ISTC (http://www.istc.illinois.edu/) and the annual Illinois Governors Pollution Prevention Award lists.

Strategies adopted to achieve this level of success include a sector approach to pollution prevention, partnerships with the IEPA (Illinois Environmental Protection Agency) and local wastewater treatment agencies, and trade associations. ISTC relies on the problem solving approach outlined here as the underpinning of its broader waste minimization strategy. This approach is crucial to achieving plant level success. Achieving sustained improvements in manufacturing efficiency and waste minimization requires constant technological innovation and an ability to leverage advances from unrelated sectors. Organizations such as the ISTC fulfill this critical role of "cross-pollination."

In conclusion, the fabricated metal products industry is diverse, incorporating many different processes and materials. Many of these operations have the potential to generate waste if incorrectly operated, poorly managed or over regulated. It would be unwieldy to provide suggestions and solutions for waste minimization under every possible scenario. Hence, our focus in this document was on the problem solving approach to achieving waste minimization. That being said, there are many excellent sources of information on specific ways to reducing waste within this industry, a few of which are listed as starting points for further reading.

Sediment Management

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Stephen Garbaciak Jr.
ARCADIS US, Chicago, IL, USA

Introduction

For a very long time, sediments only came to attention when their physical presence interfered with the human use of a water body, primarily when a river mouth or harbour became silted in and too shallow for ships to navigate. In those situations, the mud from the bottom of the river was removed, or dredged, and cast to the riverbank to allow for ship traffic to resume. As the Great Lakes basin became industrialised in the 19th century, water bodies became blocked with sediment materials that included not only the soil particles washing downstream from the surrounding watershed, but also by the physical by-products of tanning, meat packing, steel manufacturing, and other industrial operations that introduced their own unique waste materials.

These solid materials were not thought of as pollution, or contaminants, in the same way that we think today of constituents such as PCBs, dioxin, or lead. Instead, the focus of those trying to manage the waterways impacted by these materials was to simply remove the materials and find some other place to dispose of them. Dredging for the maintenance of navigation channels has been practised for hundreds of years, again with the emphasis until recently placed solely on removing solids from the navigable portion of the waterway.

With regard to the maintenance of navigable waterways, a variety of concerns about the environmental effects of dredging and management of contaminated sediments has been expressed by resource agencies and the public since the 1960s (Miller, 2003). In response to these concerns, the US Congress passed the Rivers and Harbors Act of 1970, which authorised the construction of confined disposal facilities, or CDFs, specialised structures to confine the sediments dredged from navigational channels. These CDFs were the first systematic approach to the management of contaminated sediments in North America, but the sediments themselves were not being managed solely for the purpose of removing them and their associated contaminants from the aquatic ecosystem.

Sediment contamination, and its impact on aquatic ecosystem, came into greater focus with the identification of several large Superfund sites in the early 1980s, among them the Hudson River, Kalamazoo River, Palos Verde Shelf, Commencement Bay Tidelands, and the Passaic River, all of which were centred on contamination issues in sediments. Each of the sites has been subject to intensive study for periods of decades, and remediation of them is estimated to cost billions of dollars.

The US EPA has developed comprehensive guidance for their project managers who are faced with addressing contaminated sediment sites (US EPA, 2005). This section can only provide a brief glimpse into the very basics of the complexity of managing contaminated sediment sites, which is covered in detail in the US EPA document. US EPA (2005) contains a very useful discussion of the definition of contaminated sediments and their sources:

> *'Contaminated sediment is soil, sand, organic matter, or other minerals that accumulate on the bottom of a water body and contain toxic or hazardous materials at levels that may adversely affect human health or the environment. Contaminants adsorbed to soil or in other forms may wash from land, be deposited from air, erode from aquatic banks or beds, or form from the underwater breakdown or buildup of minerals. Contaminated sediment may be present in wetlands, streams, rivers, lakes, reservoirs, harbors, along ocean margins, or in other water bodies. Some contaminants have both anthropogenic sources and natural sources.'*

The US EPA definition of contaminated sediments is functional, but does not completely illustrate why management of them as an environmental matrix is important. Fortunately, the guidance goes on to say:

> *'Many contaminants persist for years or decades because the contaminant does not degrade or degrades very slowly in the aquatic environment. Contaminants sorbed to sediment normally develop an equilibrium with the dissolved fraction in the pore water and in the overlying surface water to be taken up by fish and other aquatic organisms. Some bottom-dwelling organisms ingest contaminated sediment, and in shallow water environments, humans may also come into direct contact with contaminated sediment. Some contaminants, such as most metals, are hazardous primarily because of direct toxicity. Although some metals do accumulate in biota (i.e., bioaccumulate), generally they do not significantly increase in concentration as they are passed up the food chain (i.e., biomagnify). Others, called persistent bioaccumulative toxics (PBTs) [e.g., PCBs, pesticides, and methyl mercury] are of concern primarily because they may both bioaccumulate and biomagnify. Concentrations of PBTs in fish may endanger humans and wildlife that eat fish. Women of childbearing age, young children, people who derive much of their diet from fish and shellfish, and people with impaired immune systems may be especially at risk.'*

Let us look at how we can determine exactly what kind of problem a contaminated sediment site may present and what we can do to address the problem.

Site Assessment

As we observed with the history of sediment management, a problem with contaminated sediments can arise in two main ways – either the sediments need to be removed from a water body to provide navigable water depth, and the contaminants they contain make their handling and disposal complex (they cannot be simply cast to the side of the river), or the contaminants present in the sediments themselves present an unacceptable risk to human health or the environment, requiring the risk be managed by removing or otherwise managing the sediments.

Before identification of the best management strategy for addressing the contaminated sediment is possible, a thorough understanding of the physical, chemical and geotechnical characteristics of the site must be performed, and a comprehensive assessment of the risks posed by the sediment contaminants is critical. The mere presence of a constituent in a sediment matrix does not predetermine a need to manage the site – it is the risk that must be addressed.

The fundamental purpose of the site assessment is to develop a conceptual site model (CSM). The CSM is a representation of an environmental system that is a tool for identifying releases, pathways of migration, potential receptors and the risk posed by the contaminated sediments. Developing a CSM requires the collection of data on contaminant concentrations in sediments, water, soil, and biota. The types of data to be collected from sediment sites include (from US EPA, 2005):

Physical Data

- Sediment particle size distribution
- In situ bulk density
- Specific gravity

- Bathymetry
- Resuspension rates
- Flood frequencies and velocity distributions

Chemical Data

- Contaminant concentrations in surface and deep sediment layers
- Contaminant concentrations in biota tissue, groundwater, surface water and pore water
- Conventional pollutant concentrations, including TOC in sediments and pH in water
- Simultaneously extracted metals and acid volatile sulphides (SEM/AVS) in sediments
- Radiometric dating of sediment core profiles for deposition rate estimates

Biological Data

- Sediment toxicity – acute and chronic
- Human consumption rates of fish and shellfish in the project area
- Abundance/diversity of bottom-dwelling species, fishes, vegetation
- Tumour and abnormality surveys
- Habitat stressors

The data collected are initially used for developing the CSM, establishing the nature and extent of the contaminants of concern (CoCs), determining whether any sources of CoCs are not yet controlled and could pose a threat to recontaminate the site after remediation, identifying risk pathways through which the CoCs can adversely affect human or ecosystem health, and finally for evaluating the fate and transport mechanisms that drive these processes.

Ultimately, the CSM must be used to drive an assessment of risks posed by the sediment CoCs. Screening and baseline risk assessments are designed to evaluate the potential threat to human health and the environment in the absence of any remedial action (US EPA, 2005). The risk assessment process is key to the overall ability to determine whether a sediment site is posing unacceptable risks, and how those risks can be feasibly managed. The risks of implementing possible remedial options should be considered during the risk assessment, to provide a basis for comparing alternatives.

When risks have been adequately characterised at the site, and a determination has been made by the appropriate group of regulators and stakeholders that unacceptable risks are present and they must be managed, the next step is to develop clearly defined remedial action objectives (RAOs). RAOs are typically general in nature, and are used to develop and compare alternatives. They may also lead to the development of contaminant-specific remediation goals (RGs). RGs are generally expressed as numerical values for CoCs in sediments at the site. Examples of RAOs include (from US EPA, 2005):

- Reduce to acceptable levels the risks to adults and children from ingestion of contaminated fish and shellfish taken from the site
- Reduce to acceptable levels the toxicity to benthic aquatic organisms at the site

Remediation Options and Implementation Considerations

The menu of remedial options available to address a contaminated sediment problem has evolved from the basic choice of how to remove the target sediments from the waterway and where to dispose of the dredged material to a variety of management options that take into consideration the natural processes that may work to assist or deter from the achievement of project success.

There are three basic options available for remediating sediments:

Removal

Historically the preferred remedial alternative was to excavate the solids, water and associated contaminants from the bottom of the water body and dispose of them in a seemingly more secure and more readily monitored location. Sediments can be removed through mechanical means, where a cable-operated clamshell bucket or hydraulic excavator is used to physically dislodge the sediment from the water body and bring it to the surface. When the crane or excavator is mounted on a floating barge, the process is referred to as dredging; when the equipment is shore-based the term excavation is more typically used. Sediments can also be dredged using hydraulic methods, where a centrifugal pump is used to suction the sediment solids and water from the water body bottom, sometimes with the additional use of a rotating cutter head at the suction inlet to dislodge the sediment and help introduce the slurry into the pipeline.

In Situ Capping
In many situations the ability to remove the target sediments is difficult, the available options for disposal of the sediments are limited, or the impacts to the surrounding ecosystem from removal are too damaging, leaving it more effective to isolate the contaminated sediments in place by capping them with a layer or layers of clean material. Caps can be as simple as the placement of a thin lift of silty sand that covers and dilutes the surface concentrations of the contaminants of concern or as complex as multiple layers of geotextile, sand, silts and large armour stone to resist erosive forces and ensure the cap and the underlying target sediments remain in place.

Monitored Natural Recovery
Although it is the most recently-accepted major category of contaminated sediment management techniques, all projects should consider the natural forces acting on a contaminated sediment site to determine whether the remedial goals set for the site will be achieved with little or no human intervention. Monitored natural recovery (MNR) utilises the processes of deposition, resuspension, transport, attenuation, biodegradation and volatilisation to reduce the concentrations of constituents of concern in the surface sediments to acceptable levels in a reasonable time frame.

The identification of an appropriate remedial alternative, or combination of alternatives, must consider all components of the process. Sediment management begins with the consideration of the main technological approach – determining the ramifications of that decision involves the consideration of dredged material transport, final disposal locations, the need for the placement of residual capping materials, etc. Let us consider the main purposes of the three major remediation approaches, the site conditions that are most conducive to their implementation, and their advantages.

Monitored Natural Recovery

Monitored natural recovery (MNR) relies on multiple naturally-occurring processes to reduce the risk posed by contaminated sediments to acceptable levels with little or no human intervention. According to US EPA (2005), the following are the different processes listed in order from most to least preferable:

- The contaminant is converted to a less toxic form through transformation processes, such as biodegradation or abiotic transformations
- Contaminant mobility and bioavailability are reduced through sorption or other processes binding contaminants to the sediment matrix
- Exposure levels are reduced by a decrease in contaminant concentration levels in the near-surface sediment zone through burial or mixing-in-place with cleaner sediment
- Exposure levels are reduced by a decrease in contaminant concentration levels in the near-surface sediment zone through dispersion of particle-bound contaminants or diffusive or advective transport of contaminants to the water column

Site conditions that are particularly suitable for consideration of MNR as a remedial alternative include (from US EPA, 2005):

- Natural recovery processes have a reasonable degree of certainty to continue at rates that will contain, destroy, or reduce the bioavailability or toxicity of contaminants within an acceptable time frame
- Expected human exposure is low or can be reasonably controlled by institutional controls
- Contaminant concentrations in biota and in the biologically active zone of sediment are moving towards risk-based goals on their own
- Contaminants are readily biodegradable or transform to lower toxicity forms
- Contaminant concentrations are low and cover large areas

The primary advantage of MNR is:

- Little or no need for direct human intervention and disturbance

The disadvantages of MNR include:

- Contaminants are left in place at unacceptable levels for some period of time
- Burial-based risk reduction processes may be upset by unexpected strong forces
- Higher level of uncertainty in the prediction of remedial success

Capping

Caps are intended to reduce risk through the following primary functions [xxi]:

- Physical isolation of the contaminated sediment sufficient to reduce exposure due to direct contact and to reduce the ability of burrowing organisms to move contaminants to the surface
- Stabilisation of contaminated sediment and erosion protection of sediment and cap, sufficient to reduce resuspension and transport to other sites
- Chemical isolation of contaminated sediment sufficient to reduce exposure from dissolved and colloidally bound contaminants transported into the water column

There are certain site conditions that are particularly conducive to in situ capping and, if present, capping should receive detailed consideration as a remedial option for the site (from US EPA, 2005):

- Suitable types and quantities of cap material are readily available
- Water depth is adequate to accommodate the cap with anticipated uses (e.g. navigation)
- Incidence of cap-disturbing human behaviour, such as boat anchoring, is low or controllable
- Habitat improvements are provided by the cap
- Rates of groundwater flow in the cap area are low or not likely to create unacceptable contaminant releases through the cap
- Contamination covers contiguous areas

The advantages of capping include:

- Rapid reduction of risk posed by concentrations of CoCs in surface sediments
- Minimal or beneficial changes in site bathymetry
- No need to transport and dispose of dredged material
- Lower potential for release and transport of CoCs during remedial activities

The disadvantages of capping include:

- Leaves CoC-containing sediments in the water body
- Prevents future dredging for navigation or other purposes
- Inevitable contaminant breakthrough
- Reduction in hydraulic carrying capacity possibly leading to changes in flood stage

Dredging

Palermo et al. (2008) define environmental dredging as ‘the removal of contaminated sediments from a water body for purpose of sediment remediation’ and go on to define the objectives of an environmental dredging operation to normally include:

- Dredge with sufficient accuracy such that contaminated sediment is removed and sediment clean-up levels are met without excessive removal of clean sediment
- Dredge the sediment in a reasonable period of time and in a condition compatible with subsequent transport for treatment or disposal
- Reduce and/or control resuspension of contaminated sediments, downstream transport of re-suspended sediments, and releases of CoCs to water and air
- Dredge the sediments such that generation of residuals is reduced and/or controlled

Site conditions conducive to dredging include from US EPA, 2005):

- Suitable disposal site is available and easily reached by preferred transportation method
- Suitable area is available for staging and handling of dredged material
- Navigation dredging in the project area is scheduled or planned
- Adequate water depth to accommodate dredge
- Water diversion to facilitate excavation in dry conditions is practical
- Contaminated sediments to be removed are underlain by clean sediments, to allow for over-dredging
- High contaminant concentrations are located in relatively small areas and volumes

The advantages of dredging include:

- Permanent removal of CoCs from the water body
- Increased flexibility of future uses of the water body
- Rapid achievement of RAOs when residual concentrations are low

Disadvantages of dredging include:

- Need for transportation, dewatering, re-handling and disposal of dredged material
- Inability of dredging techniques to remove all sediments and associated CoCs
- Loss of CoCs during remediation resulting in downstream contamination

The information presented above is but the briefest summary of the three fundamental remedial options for contaminated sediment. Each of them is a subject worthy of exploration and detailed discussion in their own chapter, or complete textbook. Beyond these three fundamental options, consideration must be given to several additional components of the sediment remediation process to ensure that the most efficient, effective and least-cost option is selected for the site being managed. These additional components include:

- **Dredged material transportation method** – the methods available for transporting dredged material from the water body to the processing or disposal site depend on the dredging method. Mechanical dredges typically utilise barges or scows to move the material from the dredge to a shore-based offloading location where they are unloaded and the materials are further processed to remove excess water, or discharged directly to the disposal facility. Hydraulic dredges typically pump the dredged material in a slurry form through a pipeline directly to the offloading or disposal location. The distance to the disposal facility, and the ability of that facility to accept dredged material without additional processing, are key considerations in the selection of the dredging and dredged material transport method.
- **Dredged material processing requirements** – the facility where the dredged material will be disposed of will dictate the physical and chemical requirements for the material it can receive. If the disposal facility has been specifically built to accept dredged material, very little processing if any may be required, and direct disposal of the mechanically or hydraulically dredged material is possible. For contaminated sediment remediation projects it is more common that the dredged material will be disposed of in an existing commercial landfill, and those landfills typically set requirements that include the elimination of all free water, plus requirements on the bearing strength of what they consider to be the waste material. Since dredged material is inherently a very wet soil matrix, these requirements by the disposal facility necessitate the removal of water, and often increase in strength, of the dredged material. Dewatering methods can range from the passive use of gravity drainage through the active use of mechanical dewatering techniques such as plate-and-frame filter presses. Solidification agents such as Portland cement may be added to further reduce moisture contents and increase bearing capacity. Each of these processing steps adds time and cost to the overall project.
- Water treatment requirements – if the dredged material requires dewatering as described above, the separated water will very often require treating to remove any of the CoCs in the water. With mechanically

dredged material being dewatered through gravity drainage, this can be a relatively simple process with low treatment flow rates. However, careful consideration must be given to the water treatment needs of a dredged material slurry generated from hydraulically dredged material, where the total volume of water requiring treatment can be 20 times the volume of in situ sediments being removed from the waterway.
- **Sediment treatment requirements** – in addition to the removal and subsequent treatment of water from the dredged material, certain CoCs may need to be removed, or their concentrations reduced in the dredged material matrix before they can be disposed at a landfill. There are many chemical, physical, biological and thermal technologies available to destroy, extract or reduce the concentration of these target CoCs. These technologies may require further processing of the dredged material in order to render it suitable for treatment. These technologies all add significant additional time and cost to the overall project, and those factors must be weighed against the benefit of the reduction or elimination of the target CoC from the waste material.

Measuring Success

The final step in the sediment management process is to develop and implement a monitoring programme that will not only measure the short-term impacts and benefits of the management approach, but will provide long-term data to determine whether the project has achieved the remedial goals set for it, i.e. has reduced the risk posed by the contaminated sediments.

US EPA (2005) identifies the key points that must be considered when monitoring the success of a sediment remediation project:

- Presentation of a monitoring plan is important for all types of sediment remedies, both during and following any physical construction, to ensure that exposure pathways and risks have been adequately managed
- Development of monitoring plans should follow a systematic planning process that identifies monitoring objectives, decision criteria, endpoints and data collection, and data interpretation methods
- Before implementing a remedial action, project managers should determine if adequate baseline data exists for comparison to future monitoring data and, if not, collect additional data
- Where background conditions may be changing or where uncertainty exists concerning continuing off-site contaminant contributions to a site, it may be necessary to continue collecting data from upstream or other reference areas for comparison to site monitoring data
- Monitoring needs include both monitoring of construction and operation and monitoring intended to measure whether clean-up levels in sediment and remedial action objectives for biota or other media have been met
- Monitoring plans should be designed to evaluate whether performance standards of the remedial action are being met and should be flexible enough to allow revision if operating procedures are revised
- Field measurement methods and quick turnaround analysis methods with real-time feedback are especially useful during capping and dredging operations to identify potential problems which may be corrected as the work progresses
- After completion of remedial action, long-term monitoring should be used to identify recontamination, to assess continued containment of buried or capped contaminants, and to monitor dredging residuals and on-site disposal facilities

Conclusions

Sediment management is a continually evolving and advancing combination of science, engineering and art. It is a very young practice as the very oldest sites where contaminated sediments were actively remediated are not even 40 years old, and few examples of in situ capping are over 20 years old. As time passes, we are expanding our understanding of which techniques are most effective, which do not appear to work well, and what improvements can be made to reduce the impacts and costs of implementation while maximising the net benefits to the environment.

The reader is strongly encouraged to consult the references for further details on the specific aspects of their unique sediment management project.

Remediation of Lake Järnsjön, Sweden

33

CASE STUDY
Sweden

Leif Norrgren
Swedish University of Agricultural Sciences

Introduction

The largest watercourse in the south-east of Sweden, Emån, is 220 km in length and discharges its waters into the Baltic Sea. Its drainage basin covers an area of 4,500 km^2 and the upper part of the catchment area has many lakes, while the lower part has none. This results in large variations in water flow, causing problems such as flooding during spring and lack of water during the summer.

The river Emån and its several tributaries are of national interest and the major variety of habitats is a good source for great biodiversity in terrestrial and aquatic environments, for example a high diversity of benthic organisms.

Due to deposition of nitrogen from agriculture, cities and the atmosphere, the nitrogen concentration is high to moderate in the main watercourse and in most of the tributaries, and has increased over the past three decades. On the other hand, the concentration of phosphorus has decreased over the same period and is low to moderate today. The river Emån and some of its tributaries are polluted by metals such as cadmium, nickel and lead as a result of discharges from a battery factory and other factories that are now shut down. PCB and mercury from paper mills have also affected the watercourse.

High levels of PCB were measured in foam of Emån and in fish, and large quantities accumulated in Järnsjön, in sediments discharging PCB to the river, see Figure 33.1.

Because of a gradual reduction in the amount of PCBs, the discharge rate would probably decrease and the problem would therefore remain for a longer period of time. Remediation of Järnsjön was therefore required, using a geotextile screen (Figure 33.2).

Lake Järnsjön has been found to contribute about 3.4 kg of persistent pollutants per year to the Baltic Sea, which is a substantial amount in comparison with larger rivers discharging waters into the same sea. The Baltic Sea is very vulnerable to pollution due to its low temperature and low salinity of the water, its location and the long residence-time of the water. This characteristic leads to a high degree of accumulation of the persistent pollutants.

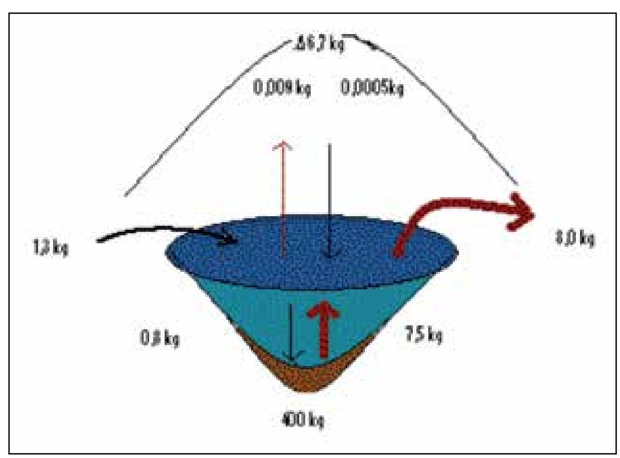

Figure 33.1. PCB budget for Lake Järnsjön before remediation (kg PCB/ year). Source: Ambio, 1998,

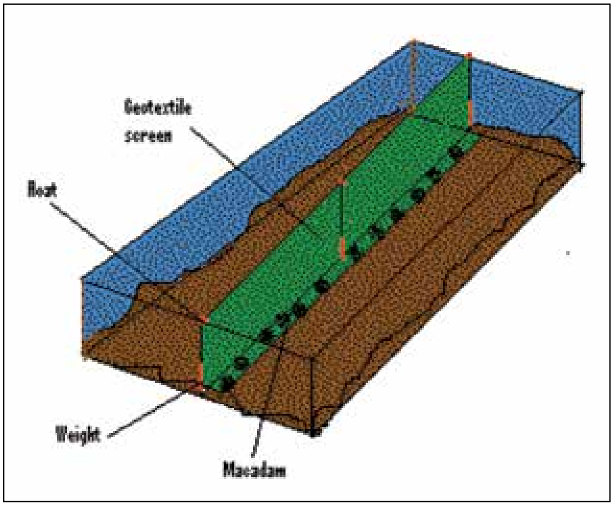

Figure 33.2. Geotextile screen. Dredging was carried out with a suction dredger in the protection of geotextile screens, which were used to separate the eastern part of the lake (highly contaminated area) from the main river flow. The sediments from Järnsjön were disposed of in landfill, covered with over a metre of sand and gravel. Source: Ambio, 1998,

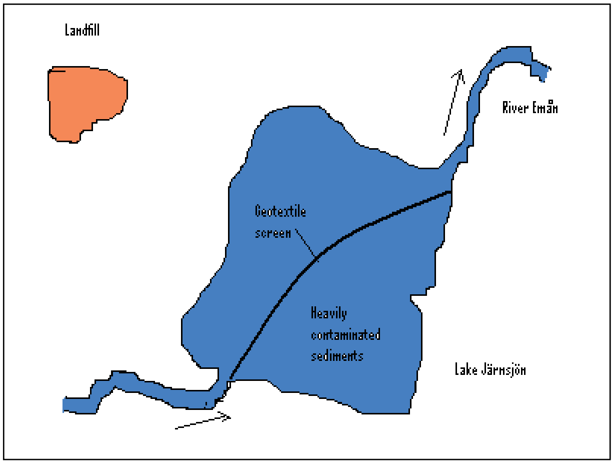

Figure 33.3. The principal geological conditions in the area of Lake Järnsjön. Source: Ambio, 1998,

Remediation of Lake Järnsjön

Methods

For remedial actions, a primary alternative was selected, including hydraulic dredging, mechanical dewatering and direct disposal of the sediments in a landfill. No in situ treatment had proved successful for PCB-contaminated sediments, and therefore removal of sediments seemed to be the best choice.

A hydraulic dredger, specially designed for contaminated material with minimal spillage, was chosen to minimise resuspension of contaminated sediments. The auger is divided into two separate parts which rotate in opposite directions, swinging from right to left, transporting the sediment to the centre of the head of the auger. Since the dredger that was used was designed for soft sediments, a bucket dredger had to be used in the southern parts of Järnsjön, where the sediments contained denser materials such as dense sand and gravel. Dredging of the eastern part of the lake was carried out during May to November 1993, and to reduce the risk to the aquatic life, dredging was stopped during December-April. The western part was dredged in the summer of 1994. After dredging, 97% of the total amount of PCB was removed, about 80% of this originated from the eastern part of the lake, the enclosed area.

An extensive programme for environmental monitoring was run during the dredging operation, measuring the turbidity and concentrations of suspended solids and PCB upstream, within the enclosed area and downstream. A higher concentration was measured occasionally downstream compared with upstream during dredging. Generally, the downstream values were lower than the upstream values, revealing that the lake acts as a sedimentation basin under normal conditions. Less than 0.5% spillage was estimated from dredging.

A geotextile screen was used as protection against spreading of contaminants to the other part of the lake, during dredging of the eastern part, see Figure 33.4. When it came to disposal of the dredged sediments, landfilling was chosen because of the relatively low degree of contamination and slow release of PCB.

The amount of transported PCB in Emån downstream of the lake was suggested to be lower during dredging in 1993 than during dredging in 1994, despite the larger amounts of PCB that were removed in the first year, when the geotextile screen was used. As a consequence, the concentrations of PCB recorded in the river were no higher during dredging than levels recorded under normal conditions, before remediation.

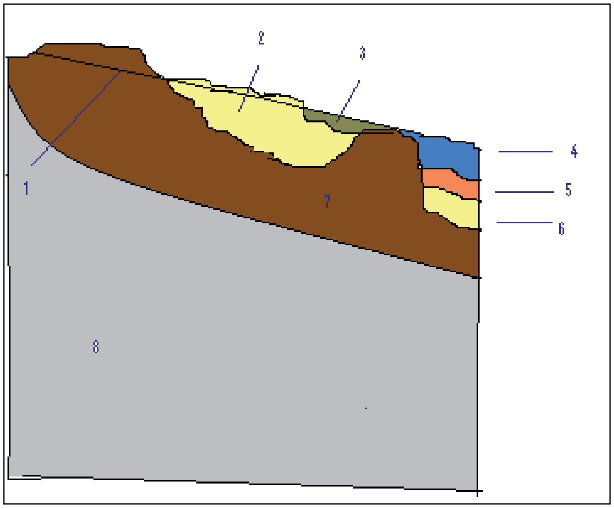

Figure 33.4. Map of Lake Järnsjön showing the position of the geotextile screen used during dredging in 1993, and the location of the landfill.
1 Groundwater level
2 Silty and sandy sediments
3 Peat
4 Lake Järnsjön
5 Contaminated sediments
6 Silty and sandy sediments
7 Glaciofluvial sediments
8 Moraine

Four possible sites were considered for the landfill, the final choice was chosen because of the closeness to the lake (250 m) and the direction of the groundwater flow.

Environmental Monitoring

Stations were set up at Lake Järnsjön and the Järnforsen waterfall to monitor the emissions of PCB from the lake sediments and their effects and to maintain guidance and limit values. Extended monitoring was carried out to follow up on environmental effects, involving:

- Studies of PCB in water at five different stations.
- Studies of PCB in fish at four different stations.
- Chemical and biological characterisation of suspended solids in Emån with analysis of PCB, PAH and PCDD/PCDF.
- Fish physiology studies were performed that focused on the detoxification system of the liver, enzyme activity, liver and gonad size and histopathological studies of gills and liver.

Moreover, PCB content in air during remediation was measured. Remediation resulted in lowered PCB concentrations in sediments, fish and to a certain degree in water. During landfilling, PCB concentrations in air were elevated but after closing of the landfill the levels went back to background levels.

Dioxin-like effects were tested using EROD-induction in cultured chicken embryo livers, and the results show that the remediation had been successful, with only 1% of the activity of the pre-dredged sediment. EROD activity of the liver, skin lesions and fin erosion was the result of caging juvenile rainbow trout before remediation. The fish showed no signs of histopathological changes two years after remediation but the EROD activity was still slightly induced. Before remediation, the health of perch was examined with physiological and morphological methods, which showed gill and liver damage with small biochemical/physiological effects. Two years after removal of PCB, the perch seemed healthy.

Part H

Food Safety and Public Health

Authors: John A. Herrmann, Marja-Liisa Hänninen, Ari Hörman and Mati Roasto

Coordinating Author: Mati Roasto

Food-borne Pathogens and Public Health

34

Mati Roasto
Estonian University of Life Sciences, Tartu, Estonia
Ari Hörman
Finnish Defence Forces, Helsinki, Finland
Marja-Liisa Hänninen
University of Helsinki, Helsinki, Finland

Introduction

Balanced nutrition, housing, clothing, health, a safe environment and personal integrity are some of fundamental needs of people. Public health is a concept of preventive measures provided by society to ensure and support these needs. Each society, following its international commitments, is obliged to focus on the relevant public health problems for its own population. Veterinary medicine is a discipline responsible for the health and welfare of domesticated animals, especially food production animals, in order to provide healthy and wholesome food for consumers. Veterinary surgeons are also important in the control and prevention of zoonotic diseases because their expertise covers the whole food animal production chain starting from primary production, continuing through slaughter and meat processing to retail, food catering and the consumer's plate. The management of food safety requires a multidisciplinary approach that employs the expertise of researchers at universities and trained experts in the field monitoring food safety. The 'farm to fork' approach includes all stages in food production and includes monitoring, surveillance and all aspects of risk identification, analysis, management and communication at each stage of the production chain. A simplified diagram of the food production chain is shown in Figure 34.1.

For many years, the community of food safety professionals has been trying to draw the attention of consumers and society to the importance of food safety for human health and the economics of food production. The importance of public health and maintaining high standards are fundamental objectives of European Union (EU) food laws as laid down in European Commission (EC) regulation No 178/2002. Throughout the EU, the aware-

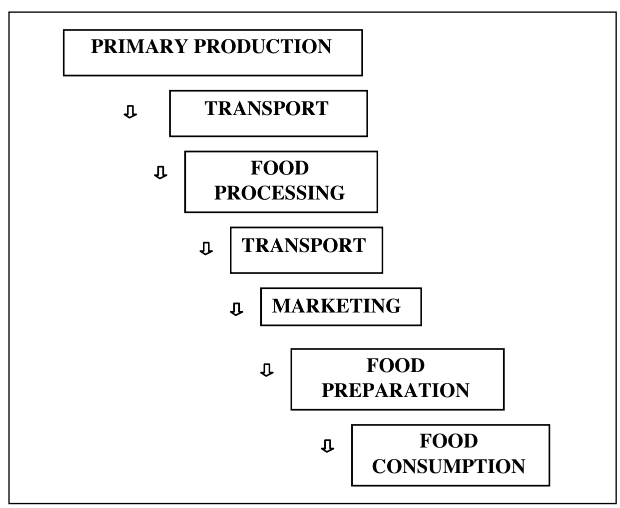

Figure 34.1. Schematic diagram of the human food chain.

ness of consumers has been increasing and consumers require the food industry to provide them with safe, nutritious and healthy food which also has high-grade sensory quality and prolonged shelf-life. To meet the demand for healthier food with high sensory quality, the use of food additives and preservatives has been reduced or eliminated and minimal processing techniques have been introduced. To increase food safety and quality, considerable amounts of time, effort and money have been spent on food safety control and management systems (ISO 22000:2005), including better packaging methods and improved and novel methods for the detection of pathogenic microbes. Nevertheless, there are still few signs in the official statistics of a significant reduction in the incidence of food-borne illnesses in the EU countries.

Todd (1997) reported that in the beginning of 1990s, 73-100% of all European outbreaks with known aetiology were caused by bacteria. Particular concerns were species such as *Listeria monocytogenes, Campylobacter jejuni, Salmonella* and *Escherichia coli* O157:H7. Biofilm formation and other problems in the production environment have been in focus lately. Wirtanen et al. (2003) reported that pathogens such as *L. monocytogenes*, *Salmonella* Typhimurium and *Yersinia enterocolitica* can readily produce biofilms, causing severe disinfection and cleaning problems on surfaces in the food industry. There is also a growing problem of human infections that are difficult to treat because of antimicrobial resistance, which is emerging partly because of the increased use of antimicrobials in human medicine and in animal production. Diseases caused by resistant strains can be significantly more severe than diseases caused by susceptible strains (INFOSAN, 2005). Problems related to the occurrence of simultaneous resistance to several antimicrobials within one strain (multiresistance) seem to be increasing and appear to be linked to the use of these antimicrobials in animal production (WHO, 2004).

The objective of this subchapter is to provide a summary of the most important bacterial food-borne pathogens, their contamination sources and routes in the food production. In addition, emerging antimicrobial resistance and current food legislation matters are discussed.

Emerging Zoonotic Food-borne Pathogens

Food-borne disease is any illness that results from ingestion of food. Food can contain microbiological pathogens that cause infections or intoxications, or chemical agents that cause acute or chronic intoxications. Biological and chemical contamination of the environment carries an important risk of food-borne disease, especially where it interfaces with food producing, processing or preparation and consumption. Drinking water supplies and waste disposal systems may be inadequate, particularly in developing countries, thus markedly increasing the risk of spreading of food-borne pathogens. Among the emerging and re-emerging pathogens, food-borne pathogens are prominently represented. They include *Campylobacter jejuni* and *C. coli*., enterohaemorrhagic *Escherichia coli*, *Listeria monocytogenes*, *Yersinia enterocolitica*, *Cryptosporidium*, multidrug resistant *Salmonella* Typhimurium DT 104, etc. As the etiological agent remains unknown in a considerable proportion of cases of food-borne diseases, many more pathogens need still to be recognised (Käferstein, 2004). Obtaining information on the epidemiology, costs and risks of food-borne pathogens is the key to controlling food-borne diseases (Snowdon et al., 2002). Examples of food-borne disease outbreaks, associated foods and etiological agents are shown in Table 34.1.

Campylobacter spp.

The genus *Campylobacter* spp. consists of 17 species and 6 subspecies (Euzeby, 2006). These bacteria are microaerophilic (85% N_2, 10% CO_2 and 5% O_2), but some can also grow aerobically or anaerobically. The most important species of *Campylobacter* are the thermophilic species: *C. jejuni* subsp. *jejuni*, *C. coli* and *C. lari*. Other species which are known gastrointestinal pathogens include *C. sputorum*, C. *upsaliensis*, *C. hyointestinalis*, *C. mucosalis*, *C. fetus* ssp. *fetus* and *C. curvus* (EFSA, 2004; Abbott et al., 2005; Euzeby, 2006). *Campylobacter jejuni* and *C. coli* are Gram-negative, spirally shaped microaerophilic bacteria. *Campylobacter* cells are mostly slender, spirally curved rods, 0.2-0.5 µm wide and 0.5-5 µm long. The rods may have one or more helical turns and can be as long as 8 µm. They also appear S-shaped and gull-wing-

Table 34.1. Food-borne disease outbreaks and associated foods.

Country of origin	Food item	Pathogen	No. of illnesses (deaths)	Reference
Canada	Chocolate	S. enterica serotype Eastbourne	119	CDC, 2004
Italy	Salami	S. enterica serotype Typhimurium	23	Connor et al., 2001
Brazil	Mango	S. enterica serotype Newport	78 (2)	Dubos, 1998
Israel	Drinking water	E. coli (ETEC)	229	Fratamico, 2002
France	Brie cheese	E. coli O27:H20 (ETEC)	45	Buzby et al., 1997
USA	Minced beef	E. coli O157:H7	3	De Jong et al., 2006
USA	Turkey meat	Listeria monocytogenes	29 (7)	Harris, 2002
USA	Frankfurters	Listeria monocytogenes	101 (21)	Harris, 2002
Sweden	Cold-smoked and gravad rainbow trout	Listeria monocytogenes	9 (2)	Colomba et al., 2006
Hungary	Raw milk	Campylobacter spp.	52	Erkmen, 1996
USA	Undercooked, barbecued chicken meat	Campylobacter jejuni	11	Eggertson, 2005
Germany	Chocolate drink made from raw milk	Campylobacter jejuni	24	Farmer et al., 1980

shaped when two cells form short chains (Holt et al., 1994). Cells of some species are predominantly curved or straight rods. Cells in old cultures may form coccoid bodies which are considered degenerative forms rather than a dormant stage of the organism (Hazeleger et al., 1994). The cells of most species are motile, with a characteristic corkscrew-like motion by means of a single, unsheathed, polar flagellum at one or both ends of the cells. Cells of some species such as *C. hominis* and *C. gracilis* are non motile, while other species such as *C. showae* have multiple flagella. *Campylobacter* spp. are relatively inactive biochemically, obtaining their energy from amino acids or tricarboxylic acid cycle intermediates rather than carbohydrates. Carbohydrates are neither fermented nor oxidised. This makes them difficult to speciate by use of classical biochemical tests (On, 1996), so they are often identified to species level by use of DNA-based methods (Bolton et al., 2002; On and Jordan 2003).

Some species are pathogenic for humans and animals. *Campylobacter jejuni* and *C. coli* are the major pathogenic species of interest. They are found in the intestinal tract and oral cavity of man and animals. The predominant species in human infection can be readily grown under a microaerobic atmosphere on selective media without the necessity to use hydrogen. Campylobacters grow optimally in an atmosphere containing 5% oxygen and have an optimum growth temperature between approximately 30 and 45ºC. They survive during storage at refrigerated temperatures better than at room temperature. The cells are sensitive to freezing, drying and salt concentrations above 1% sodium chloride. Campylobacters are also sensitive to standard concentrations of common disinfectants (Anonymous, 1993). *Campylobacter* spp. are relatively sensitive to heat and irradiation, and they can readily be inactivated during cooking (ICMSF, 1996). *C. jejuni* and *C. coli* are the main cause of *Campylobacter* enteritis in humans (Skirrow and Blaser, 2000; Hänninen et al., 2003). *C. jejuni* is responsible for 80-90% of campylobacteriosis. It causes more common enteric diseases than *Shigella* spp. and *Salmonella* spp. combined. In the industrialised countries, including Western Europe, USA, Canada, Australia and New Zealand, the rate of human *Campylobacter* infections has been increasing steadily since the mid-1990s. On 12 April 2000, the Scientific Committee on Veterinary Measures relating to Public Health (SCVMPH) issued an opinion on food-borne zoonoses (SCVMPH, 2000). In this opinion the Committee identified *Campylobacter* spp. as one of the public health priorities among the food-borne zoonotic pathogens. Campylobacteriosis represents an important public health problem with considerable socio-economic impact in the EU. In 2004, a total of 183,961 human cases of campylobacteriosis were reported from the 25 member states of the European Union. The EU incidence was 47.6 cases per 100,000 population. In 2005, 2631 confirmed campylobacteriosis cases were reported in Norway, 4002 cases in Finland, 3677 cases in Denmark, 5969 cases in Sweden and 124 cases in Estonia. The overall incidence of campylobacteriosis in the EU in that year was 51.6 per 100,000 population, ranging from <0.1 to 302.7 cases per 100,000 population (EFSA, 2006). In 2007, a total of 200,507 human cases of campylobacteriosis were reported in EU member states, an increase of almost 25,000 cases compared with 2006, and the EU incidence was 45.2 cases per 100,000 population (EFSA, 2009a). In 2000, 78 campylobacteriosis cases were recorded in Denmark but the estimated incidence of *Campylobacter* infections may have been 600-8300 cases per 100,000 population (Rosenquist et al., 2003). An estimated 2.5 million cases of *Campylobacter* infection occur each year in the United States, and 80% of these cases have been found to be the consequence of a food-borne transmission (Bhaduri and Cottrell, 2004). The high *Campylobacter* contamination of raw poultry products observed by Praakle-Amin et al. (2007) in retail outlets in Estonia may indicate that the prevalence of human campylobacteriosis in Estonia is greater than the 154 cases (11.5 cases per 100,000 inhabitants) reported by the Estonian Health Protection Inspectorate in 2009. Other Baltic countries report low numbers as well. In Lithuania, a total of 564 cases (notification rate 16.4 per 100,000 inhabitants) of human campylobacteriosis were reported in 2007, while no cases were reported in Latvia (EFSA, 2009a).

Associated Foods and Environment

Campylobacter spp. are widespread in nature, not only in wildlife but also among food animals, such as cattle, sheep, swine and avian species, as commensal organisms (Friedman et al., 2000). The avian species are the most common hosts for *Campylobacter*, probably because of their higher body temperature (Skirrow, 1977), as well their breeding in large dense animal populations which

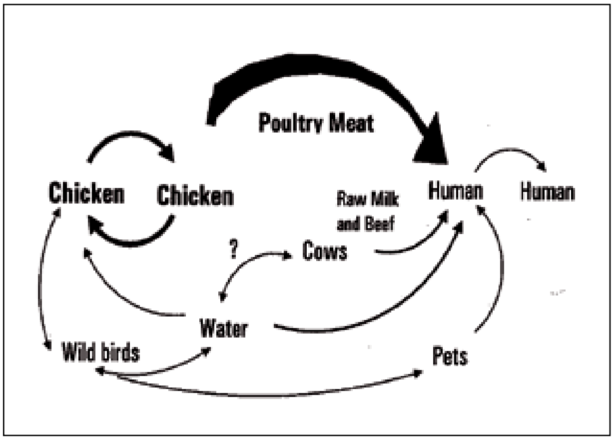

Figure 34.2. Campylobacter jejuni transmission. Source: Hänninen, 1999.

promote transmission of campylobacteria within a flock. Figure 34.2 illustrates possible routes of *Campylobacter jejuni* transmission. Monitoring studies indicate that chicken flocks are commonly colonised with *C. jejuni*. There are large differences in the annual prevalence of positive flocks between different countries in the EU (EFSA 2009a). Studies in Europe indicate flock prevalence ranging from 18 to over 90%, with northern countries showing a lower proportion of positive flocks (Barrios et al., 2006). Intestinal colonisation usually leads to contamination of the final product, which cannot be prevented in the processing plant (Hänninen, 2009). Studies carried out in slaughterhouses have shown that the main source of the spread of *C. jejuni* on poultry carcasses is their intestinal contents (Stern et al., 2003). *Campylobacter* have also been isolated from food items such as raw milk, pork, beef, lamb and seafood (Duffy et al., 2001).

The presence of *Campylobacter* spp. in raw food materials and products of animal origin may represent a source of infection, but a real health hazard exists only when meat consumed is raw or undercooked (Domingues et al., 2002). The other major hazard may be a result of improper hygiene habits and disregard of Good Manufacturing Practice (GMP) principles. This is related to the transfer of bacteria from raw meat to other foodstuffs (cross-contamination). An effective quality control programme in a large-scale poultry processing plant in Estonia accounted for the lower contamination levels of fresh chicken meat compared with the contamination level for the same type of products in a small-scale plant (Roasto et al., 2005). Altogether, 279 samples of Estonian raw chicken meat (breasts, carcasses, legs, minced meat, thighs and wings) were analysed during 2000 and 2002 (Roasto et al., 2005). Of these, 90 were collected directly from the end of the slaughter line of a small-scale poultry meat plant and 189 from traditional market halls of Tartu town, Estonia. All chicken meat samples from market halls were sold fresh and unpacked. Of the raw chicken products of Estonian origin, 15.8% tested positive for *Campylobacter*. The prevalence of *Campylobacter* in the products (breasts, carcasses, thighs and wings) of the small-scale poultry meat plant (35.6%) was significantly higher than in those originating from the large-scale company (6.3%) ($P<0.001$). The occurrence of *Campylobacter* spp. in broiler chicken production in Estonia in the period 2002-2007 was analysed by Meremäe et al. (2010). *Campylobacter* spp. were isolated in 163 (12.3%) of 1320 chicken meat samples tested in 2002-2007 and in 115 (6.3%) of 1819 caecal samples tested in 2005-2007. All 1254 faecal samples collected in 2005 and 2006 from two farms with a total of 60 flocks, each containing 20,000 birds, were negative. *C. jejuni* was the most commonly isolated species in Estonian analyses (98.2%), followed by *C. coli* (1.4%) and *C. lari* (0.4%). The seasonal peak of *Campylobacter* contamination was from July to September.

More data about the prevalence of *Campylobacter* spp. on fresh poultry meat world-wide are presented in Table 34.2.

Listeria Monocytogenes

Listeria monocytogenes is transmitted via three main routes: direct contact with animals, cross-infection of new-born babies in hospital and food-borne infection. The latter two sources result in the majority of listeriosis cases in humans. Listeriosis is an uncommon but a serious food-borne disease that can be life-threatening to the elderly, people with a weakened immune system and pregnant women (Frye et al., 2002). In EU member states, 1558 human cases of listeriosis were reported in 2007 (EFSA, 2009b). *Listeria monocytogenes* accounts for about 2500 cases, 2289 hospitalisations and 449 deaths each year in the United States. The mortality rate of *L. monocytogenes* (~28%) remains the highest of all food-borne pathogens (Wesley, 2009). In humans, severe illness mainly occurs in unborn children, infants, the elderly and those with a compromised immune system. Symptoms vary, ranging from mild flu-like symptoms and diarrhoea to life-threatening infections characterised by septicaemia and meningoencephalitis. In pregnant women the infection can spread to the foetus, which may either be born severely ill or die in the uterus and result in abortion.

L. monocytogenes is a Gram-positive and motile bacterium that is commonly present in the environment and occurs in almost all raw food materials sporadically. According to current knowledge the genus *Listeria* contains six clearly distinguishable species. The most commonly occurring species in food are *L. monocytogenes* and *L. innocua,* although *L. monocytogenes* is the only

Table 34.2. Campylobacter spp. isolated from fresh poultry meat.

Product	Country of origin	No. of positive samples (%)	Reference
Chicken carcassa	Finland	28 (14)	Aho and Hirn, 1988
Goose carcass	Poland	76 (38)	Kwiatek et al., 1990
Chicken wings	Nothern Ireland	99 (65)	Flynn et al., 1994
Poultry meata	Chile	117 (93)	Fernandez and Pison, 1996
Retail poultry meat	The Netherlands	431 (37)	de Boer et al., 2000
Turkey meat	Denmark	78 (25)	Hald et al., 1998
Chicken carcass	Japan	13 (59)	Ono and Yamamoto, 1999
Retail chicken meat	Spain	98 (50)	Dominguez et al., 2002
Retail poultry meat	South Africa	1 (7)	van Nierop et al., 2005
Retail poultry meat	Estonia	163 (12)	Meremäe et al., 2010
Retail poultry meat	Latvia	125 (10)	EFSA, 2006

important human pathogen of the genus (Catteau, 1995). Some studies suggest that 1-10% of humans may be intestinal carriers of *L. monocytogenes*. It has been isolated from at least 37 mammalian species, both domestic and feral, as well as at least 17 species of birds and possibly some species of fish and shellfish. Healthy birds may asymptomatically shed *L. monocytogenes* in faecal material (Skovgaard and Morgen, 1988). Poultry meat is contaminated during slaughtering and processing (Rørvik et al., 2003). *Listeria* can be isolated from water, soil, silage and other environmental sources. *L. monocytogenes* is quite hardy and resists the detrimental effects of freezing, drying and heat remarkably well for a non spore-forming bacterium (Johansson, 1999).

Associated Foods

L. monocytogenes has been associated with food sources such as raw milk, improperly pasteurised milk, cheeses (particularly soft-ripened varieties), ice cream, raw vegetables, fermented raw-meat sausages, raw and cooked poultry, raw meats (all types) and raw and smoked fish (Farber and Peterkin, 1991). *Listeria* is able to grow at temperatures as low as 3°C and this permits its multiplication in refrigerated foods. It can survive or even grow at pH values as low as 4.4 and at salt concentrations of up to 14% (Berziņš et al., 2007). In a study by Praakle-Amin et al. (2006), a total of 240 raw broiler legs (120 of Estonian origin and 120 of foreign origin) from 12 retail stores in the two biggest cities (Tallinn and Tartu) in Estonia were investigated from January to December 2002. Of the raw broiler legs, 70% tested positive for *L. monocytogenes*. The prevalence of *L. monocytogenes* in broiler legs of Estonian origin (88%) was significantly higher than in broiler legs of foreign origin (53%) (P<0.001). Praakle-Amin et al. (2006) concluded that the high prevalence of *L. monocytogenes* showing various PFGE types in the broiler legs could be caused by cross-contamination at retail level. Ready-to-eat meat products with a long shelf-life are associated with a risk of transmission of *L. monocytogenes* (Farber and Peterkin, 1991).

The prevalence of *L. monocytogenes* in cold smoked, sliced, vacuum-packaged pork products during 15-month period from 2003 to 2004 was studied by Berziņš et al. (2007). Samples originated from 8 Latvian and 7 Lithuanian manufacturers. The prevalence of *L. monocytogenes* in cold-smoked pork varied from 0-67% in Latvian products and 10-73% in Lithuanian products. In order to identify the main risk factors associated with *L. monocytogenes* contamination, all production steps were studied separately in each meat processing plant. Berziņš et al. (2007) suggested that brining by injection was a significant (P<0.05) risk factor in contamination. Moreover, long cold-smoking times (12 h) had a significant (P<0.014) predictive value for a sample to be positive for *L. monocytogenes*. Cold-smoking temperatures between 24 and 30°C can have an inhibitory effect on the presence of *L. monocytogenes*. Low numbers of *L. monocytogenes* at the end of shelf-life (< 100cfu/g) can be explained by the use of starter cultures during processing, which have an antilisterial effect and inhibit the multiplication of *L. monocytogenes* in pork products. It is recognised that the presence of *L. monocytogenes* in raw foods cannot be completely eliminated, but through the application of effective hygiene measures, it is possible to reduce its occurrence and level in food products. In order to ensure the safety of food products, growing, harvesting, handling, storage, processing and food supply systems must be managed by food handlers in such a way that they are able to reliably control the growth of *L. monocytogenes* and to prevent its multiplication to the potentially harmful level of >100/g (Roasto, 2009).

Salmonella

Salmonella is a member of the family *Enterobacteriaceae*, which comprises a large and diverse group of Gram-negative rods. Members of the genus *Salmonella* are zoonotic and can be pathogenic in man and animals. Salmonellae are facultatively anaerobic, Gram-negative, straight, small rods, which are usually motile with peritrichous flagella. They are non lactose-fermenting and non spore-forming. There are currently well over 2400 serotypes. Epidemiological classification of *Salmonella* is based on host preference. One group includes serotypes that infect only humans, for example, *S. typhi* and *S. paratyphi*.

Salmonellosis is one of the most common and widely occurring food-borne diseases and constitutes a major public health burden and represents a massive cost to society in many countries. Millions of cases are reported world-wide every year, resulting in thousands of deaths. A *Salmonella* control programme in food animal production has been in place for several years in Denmark and the annual estimated cost of this control programme is 10.8 million euros. It is estimated that this programme saves 19.6 million euros for the Danish economy annually. Some countries have managed to limit and even reverse salmonellosis but the spread of two strains of *Salmonella,* namely *Salmonella* Enteritidis and *Salmonella* Typhimurium, is causing increased concern (European Commission, 2004). Multiresistant strains of *Salmonella* are now encountered frequently. The occurrence of multiresistance has increased considerably in recent years owing to the global spread of multiresistant *Salmonella* Typhimurium DT104. While the spread of DT104 may have been facilitated by the use of antimicrobials, international and national trade in infected animals is thought to play a major role in its dissemination (INFOSAN, 2005).

Associated Foods

A wide variety of foods have been identified in outbreaks caused by several serotypes of S*almonella*: raw meats, poultry, eggs, milk and dairy products, fish, shrimp, frog legs, yeast, coconut, sauces and salad dressing, cake mixes, cream-filled desserts and toppings, dried gelatine, peanut butter, cocoa, chocolate and even dried chilli (Figure 34.3). Various *Salmonella* serotypes have long been isolated from the outer surface of egg shells. The current status of *S. enteritidis* is complicated by the presence of the organism inside the egg, in the yolk. This and other information strongly suggest vertical transmission, i.e. deposition of the organism in the yolk by an infected layer hen prior to shell deposition. Foods other than eggs have also caused outbreaks of *S. enteritidis* disease. *Salmonella* is still the most frequently recorded pathogen in the production chain of food of animal origin. At present the predominant serotypes are *S. enteritidis* and *S.* Typhimurium, particularly in terms of the most important meats from pig and poultry. In areas such as Scandinavia, measures against this pathogen have traditionally been more thoroughly implemented, ultimately resulting in a lower prevalence of *Salmonella* in these countries compared with Central Europe (Roasto et al., 2006). Whatever the *Salmonella* serotype, effective controls for minimising/eliminating the hazard of *Salmonella* from foods involve control of the following steps: raw materials, personal and environmental hygiene, process conditions, post-process contamination, retail and catering practices and consumer handling (Roasto et al., 2006).

Figure 34.3. Possible route of contamination of chillies with Salmonella during drying. Photo: Gul Chotrani, Lombok, Indonesia, 2005.

Escherichia Coli

E. coli is a short, typically motile, facultatively anaerobic, non spore-forming, Gram-negative, rod-shaped bacterium (1.1-1.5 µm and 2.0-6.0 µm) within the family *Enterobacteriaceae.* The optimum growth temperature is 37ºC. The combination of O and H antigens defines the *E. coli* serotype, and serotyping of isolates is useful for characterisation of commensal and pathogenic serotypes and as a tool for epidemiological investigations. *E. coli* forms a part of the natural gastrointestinal microbiota of man and warm-blooded animals. Normally, *E. coli* serves a useful function in the body by suppressing the growth of harmful bacterial species and by synthesising appreciable amounts of vitamins. Although most *E. coli* strains are harmless commensal organisms, there are many pathogenic strains capable of causing a variety of illnesses in humans.

There are six recognised groups of pathogenic *E. coli* (EAEC, EPEC, ETEC, EIEC, EHEC, VTEC). Each group has different virulence features and mechanisms of pathogenity (Fratamico et al., 2002; Duffy, 2006). The enteroaggregative *E. coli* (EAEC) are associated with persistent diarrhoea in young children, especially in developing countries. These strains produce three toxins which stimulate intestinal secretion. The enteroaggregative *E. coli* (EPEC) cause severe diarrhoea in infants. Certain EPEC strains produce one or more cytotoxins. The enteroaggregative *E. coli* (ETEC) cause also diarrhoea in humans, both in infants and adults, and in the latter the world-wide illness known as traveller's diarrhoea. ETEC strains produce enterotoxins. The enteroaggregative *E. coli* (EIEC) produce a cytotoxin and often induce rather severe illness such as colitis and a form of dysentery, accompanied by fever and bloody stools. The enteroaggregative *E. coli* (EHEC) produce cytotoxins which give more severe symptoms. These toxins (verotoxin 1 and verotoxin 2) are closely related or identical to the toxin produced by *Shigella dysenteriae*. They have the same biological activity but can be distinguished immunologically. The toxins are lethal to Vero cells and hence are known as Verocytotoxin producing *Escherichia coli* or VTEC. The toxins destroy the intestinal cells of the human colon, causing haemorrhagic colitis (HC) which is characterised by severe abdominal pain and diarrhoea. About 15% of HC cases, notably in children, develop haemolytic uraemic syndrome (HUS). This is a form of renal failure and haemolytic anaemia and may result in permanent kidney damage. *E. coli* serotype O157:H7 is the most well-known EHEC strain. Due to current detection procedures *E. coli* O157:H7 is the only serotype routinely identified, but other verotoxigenic *E. coli* serotypes such as *E. coli* O26:H11 are known (Forsythe and Hayes, 1998; Fratamico et al., 2002).

Associated Foods

Strains of EAEC have been isolated from the contents of infant feeding bottles and outbreaks have been associated with food. Disease due to EPEC occurs predominantly in developing countries, with vehicles including different foods or infant formula, contaminated hands and contaminated utensils such as linen, scales, toys, etc. ETEC strains have been implicated in several large water-borne outbreaks, while a wide range of foods (various meats and poultry, mashed potatoes, milk products) have been implicated as sources. Many outbreaks have been attributed to EIEC strains in cheeses, milk, meats, potato salads and other foods. Outbreaks of the most well-known EHEC strain, *E. coli* O157:H7, have mostly been attributed to undercooked or raw minced beef meat. However, *E. coli* O157:H7 outbreaks have also resulted from contaminated alfalfa sprouts, unpasteurised fruit juices, drinking water, dry-cured salami, lettuce, game meat and cheese curds. Raw milk was the vehicle in a school outbreak in Canada (Forsythe and Hayes, 1998; Fratamico et al., 2002).

Antimicrobial Resistance

Many food-borne pathogens can have natural habitats in food animals. They can enter meat and milk production at slaughter and milking, or contaminate raw vegetables when the soil is fertilised with animal manure. The molecular analysis of antibiotic resistance genes, plasmids and transposons has demonstrated that identical elements are found in animals and humans. The use of antibiotics in veterinary medicine, either for preventive or prophylactic purposes or in therapy, selects resistant pathogenic, opportunistic and commensal bacteria. These bacteria are released into the environment. Specific food items, water and direct contact can spread these bacteria from animal microflora to human microflora (Teuber, 2004).

The development of antimicrobial resistance in pathogenic bacteria is a matter of increasing concern. There is growing scientific evidence that the use of antimicrobials in food animals leads to the development of resistant pathogenic bacteria that can reach humans through the food chain (Van Looveren et al., 2001). These human food safety concerns have been influential in prompting the European Union to ban the use of antimicrobials as growth promoters in food production and to increase their surveillance for antimicrobial resistance of food-borne pathogens and indicator organisms (Smith et al., 2007). In farm environments, commensal and environmental bacteria may be a reservoir for the transfer of antimicrobial resistance genes to pathogenic bacteria. Bacteria may acquire resistance genes through horizontal transfer. Conjugative genetic elements such as plasmids and transposons are common vectors for the dissemination of antimicrobial resistance genes to diverse microorganisms (Smith et al., 2007). Many scientists and public health specialists expect this resistance problem to worsen unless we act decisively.

Estonian antimicrobial susceptibility studies (Roasto et al., 2007) of *Campylobacter* strains revealed high resistance patterns for several antimicrobials (Table 34.3). A high proportion (27.5% of isolates) of multidrug resistance to three or more unrelated antimicrobials was found. All these isolates were resistant to enrofloxacin and all except one were resistant to nalidixic acid. Hakanen et al. (2003) noted that 20% of the Finnish human *Campylobacter* isolates associated with travel were resistant to three or more antimicrobials. Multiresistant isolates in the study by Roasto et al. (2007) consisted of a combination of all tested antimicrobials. The results showed that multiresistance was significantly associated with enrofloxacin and nalidixic acid resistance (correlation coefficient 0.372 and 0.310, $P<0.01$). These findings suggest that the use of fluoroquinolones may select multiresistant strains, since resistance to erythromycin, gentamicin or oxytetracycline was exceptional without simultaneous resistance to fluoroquinolones. A recent study on antimicrobial resistance of *Escherichia coli* at a farm where no antimicrobial treatment of the birds was performed during one year before the sampling showed that the resistance to tetracycline, gentamicin and streptomycin persisted but all isolates were susceptible to enrofloxacin (Smith et al., 2007). Thus multiresistant strains may reflect the past history of antimicrobial usage during a longer period. This phenomenon may also partly explain the rather high number of multiresistant strains in the Estonian study (Roasto et al., 2007). Antimicrobial susceptibility patterns of *C. jejuni* isolates from broiler chickens in Estonia in 2005-2006 are shown in Table 34.3

Table 34.3. Antimicrobial susceptibility of C. jejuni isolates (n = 131) from broiler chickens in Estonia, 2005-2006.

Antimicrobial agent[a]	Antimicrobial concentration range (µg/ml) VetMICTMCamp	Breakpoint (µg/ml)	No. of resistant strains (%)
Am	0.5-64	32	10 (7.6)
Ef	0.03-4	1	96 (73.3)
Em	0.12-16	16	26 (19.8)
Gm	0.25-8	8	25 (19.1)
Nal	1-128	32	99 (75.6)
Tc	0.25-32	4	42 (32.1)

[a]Antimicrobial agents: Am, Ampicillin; Ef, Enrofloxacin; Em, Erythromycin; Gm, Gentamicin; Nal, Nalidixic acid; Tc, Oxytetracycline.

The costs of treating antimicrobial-resistant human infections place a significant burden on society. For example, it has been estimated that the in-hospital cost of hospital-acquired infections caused by just six common kinds of resistant bacteria was at least 1.3 billion USD (1 billion euros) in 1992 (http://www.hhs.gov/news/press/2001pres/20010118b.html; accessed 24 May 2007). Multiresistance is a major public health problem because it limits chemotherapeutic options. The concept of Critically Important Antimicrobials for humans and animals should be used by EU member states for setting priorities for improved management of antimicrobials in animal production. Figure 34.4 and 36.5 illustrate the susceptible and resistant *C. jejuni* strains identified using minimal inhibitory concentration testing by E-test™.

Legislation

The aim of EU legislation is to be clear about the responsibilities of the various stakeholders in the food production chain. Legislation aims to clarify that responsibility for the safety of food on the market rests on the food operators; that the relevant authorities in individual member states are in charge of monitoring and enforcing this responsibility through national surveillance and control systems; and finally that the Commission must concentrate on evaluating the ability of the relevant authorities to carry out these tasks through audits and inspections. The Commission should continue to reinforce its farm to table policy, covering all sectors of the food chain, including feed production, production on the farm, food processing, storage, transport and retail sale (Daelman, 2002).

General Food Law – Principles

The food law aims at ensuring a high level of protection of human life and health, as well taking into account the

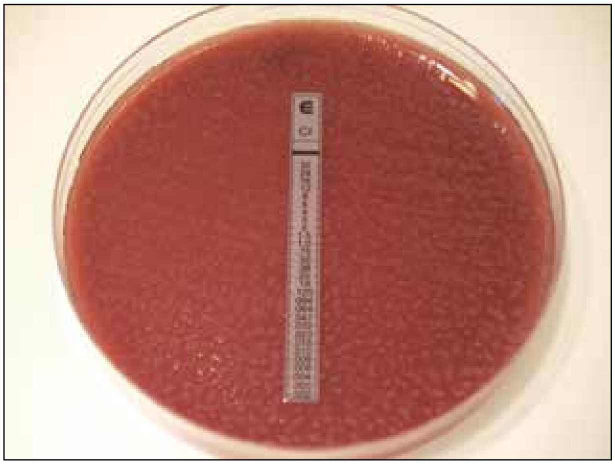

Figure 34.4. Campylobacter jejuni strain resistant to ciprofloxacin in E-test™. Photo: Mati Roasto, 2005.

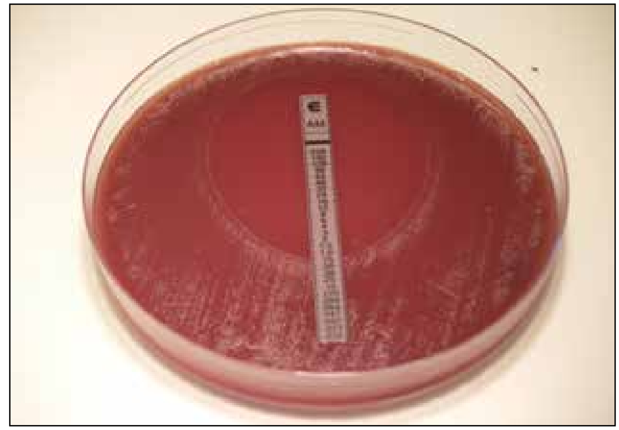

Figure 34.5. Campylobacter jejuni strain susceptible to ampicillin in E-test™. Photo: Mati Roasto, 2005.

protection of animal health and welfare, plant health and the environment. This integrated 'farm to fork' approach is now considered a general principle for EU food safety policy. Food law, both at national and EU level, establishes the rights of consumers to safe food and to accurate and honest information. The EU food law aims to harmonise existing national requirements in order to ensure the free movement of food and feed in the EU. The food law recognises the European Union commitment to its international obligations and will be developed and adapted taking international standards into consideration, except where this might undermine the high level of consumer protection pursued by the EU

Risk Analysis

EU regulation EC 178/2002 establishes the principles of risk analysis in relation to food and establishes the structures and mechanisms for the scientific and technical evaluations which are undertaken by the European Food Safety Authority (EFSA).

Depending on the nature of the measure, food law, and in particular measures relating to food safety, must be underpinned by strong science. The EU has been at the forefront of the development of the risk analysis principles and their subsequent international acceptance. Regulation EC 178/2002 establishes in EU law that the three interrelated components of risk analysis (risk assessment, risk management and risk communication) provide the basis for food law as appropriate to the measure under consideration. Clearly not all food law has a scientific basis, e.g. food law relating to consumer information or the prevention of misleading practices does not need a scientific foundation.

Scientific assessment of risk must be undertaken in an independent, objective and transparent manner based on the best available science. Risk management is the process of weighing policy alternatives in the light of results of a risk assessment and, if required, selecting the appropriate actions necessary to prevent, reduce or eliminate the risk to ensure the high level of health protection determined as appropriate in the EU.

In the risk management phase, the decision-makers need to consider a range of information in addition to the scientific risk assessment. These include, for example, the feasibility of controlling a risk, the most effective risk reduction actions depending on the part of the food supply chain where the problem occurs, the practical arrangements needed, the socio-economic effects and the environmental impact. Regulation EC/178/2002 establishes the principle that risk management actions are not just based on a scientific assessment of risk but also take into consideration a wide range of other factors legitimate to the matter under consideration.

Transparency

Food safety and the protection of consumer interests are of increasing concern to the general public, non-government organisations, professional associations, international trading partners and trade organisations. Therefore, the EU Regulation establishes a framework for the greater involvement of stakeholders at all stages in the development of food law and establishes the mechanisms necessary to increase consumer confidence in food law.

This consumer confidence is an essential outcome of a successful food policy and is therefore a primary goal of EU action related to food. Transparency of legislation and effective public consultation are essential elements of building this greater confidence. Better communication about food safety and the evaluation and explanation of potential risks, including full transparency of scientific opinions, are of key importance.

EU Commission Regulation (EC) No 2073/2005 of 15 November 2005 on microbiological criteria for foodstuffs constitutes that foodstuffs should not contain microorganisms or their toxins or metabolites in quantities that present an unacceptable risk for human health. EU Regulation (EC) No 178/2002 lays down general food safety requirements, according to which food must not be placed on the market if it is unsafe. The use of microbiological criteria should form an integral part of the implementation of HACCP-based procedures and other hygiene control measures. According to Article 4 of EU Regulation (EC) No 852/2004, food business operators are to comply with microbiological criteria. This should include testing against the values set for the criteria through the taking of samples, the conduct of analyses and the implementation of corrective actions, in accordance with food law and the instructions given by the competent authority. Article 5 of EU Regulation (EC) No 2073/2005 lays down specific rules for testing and sampling, according to which the ISO standard 18593 must be used as a reference method. Food business operators manufacturing ready-to-eat foods, which may pose a *L. monocytogenes* risk for public health, must sample the processing areas and equipment for *L. monocytogenes* as part of their sampling scheme. Food safety and process hygiene criteria are given in chapter 1 and 2 of EU Commission Regulation (EC) No 2073/2005 for microbiological criteria for foodstuff.

The basic principles of food hygiene introduce a certain level of flexibility that is believed essential in order to take account of particular situations. This is in particular the case with regard to the implementation of the HACCP system, traditional ways of preparing certain food, and for certain small enterprises. It is clearly stated that this must not compromise food safety.

Finally, all the legislation alone is not able to guarantee the quality and safety of the food. Food hygiene legislation and detailed microbiological standards are meaningless if the legislation is impossible to apply in practice. Most important, however, is the care taken in the whole chain of food handling operations. The quality of raw materials, the hygiene environment within the food processing enterprise, the processing standards applied and the attitude of food enterprise personnel are all of crucial importance.

Future Needs

There is likely to be an increased need for attention to the safety of the food supply in relation to population growth, and therefore the need to provide food and to dispose safely of faecal and chemical waste will increase. There is a need for research on more sensitive, reliable and cost-effective tools, particularly sampling methodologies, for analysing food and environmental samples (e.g. high priority commodities including eggs and seafood) for microbial pathogens. This is especially needed where the frequency and extent of contamination are expected to be low and for identification and evaluation of relevant characteristics of different forms of product packing and handling on the safety of a variety of foods. Other areas where research is needed include: development of modelling techniques to assess microbial behaviour in various foods, human exposure and dose-response relations to certain food-borne pathogens (e.g. enumerative detection methods for pathogens), the potential risk of those pathogens causing human illness and the setting of safety performance standards to regulate microbial content of food, and determining the population trends with respect to food safety knowledge, attitudes and practices, especially behaviours that may be a significant risk factors for food-borne illness (e.g. food consumption, in-home food preparation and handling).

The microbiological safety of food has been advanced substantially by the introduction and implementation of the hazard analysis of critical control point (HACCP) concept. HACCP provides a systematic conceptual framework for identifying hazards and focusing efforts on the proper functioning of key food production, processing and marketing steps. HACCP cannot be expected to control unknown hazards, such as emerging food-borne pathogens. There is a need to re-examine how food is produced, processed, marketed and prepared to identify conditions that contribute to emergence. The changing epidemiology of food-borne disease calls for improved surveillance, including rapid sub-typing methods, cluster identification and collaborative epidemiological investigation (including case-control studies). The new problems of food-borne disease require new control and prevention strategies to ensure that food in both domestic and international trade is safe. Topics included a need for multidisciplinary teams that can provide 'just in time' research; for basic research to explain factors associated with food production and processing that contribute to new food-borne microbial threats; for prompt evaluation and implementation of innovative preservation methods (e.g. food irradiation) to meet consumer demand for fresh foods; for the use of emerging molecular methods (e.g. polymerase chain reaction and molecular typing) to examine emerging food-borne disease organisms; and for models to predict the probability of a particular microbial event (e.g. growth and death), which may be useful in the design of HACCP programs and in defining processes, formulations and storage conditions to yield foods with acceptable shelf-life and safety characteristics.

Conclusions

There is a long and multi-step path taken by food of animal origin from the farm to the consumer's plate. There are a lot of circumstances and potential hazards, which may or may not constitute a risk to humans. As a consequence, measures should be taken, especially where the prevalence of pathogens has been high, i.e. hygiene in primary production, immunisation, logistical slaughter or measures in cleaning and disinfection of the food processing site. It is obvious that the inspection service by the authorities cannot afford the total of surveillance in every production stage. The hygiene status of intermediate products and end products is particularly dependent on the circumstances of previous stages of production. In consequence, hygiene is an issue of day-to-day practices, and checks must be carried out frequently. Here, the authorities have to rely more on the responsibility of the processing plant. The role of the authorities is currently being reconsidered in order to focus the available resources on the essentials of surveillance. This is true also with respect to future additional tasks of surveillance in animal husbandry, which will possibly demand more personnel in the future. It should be emphasised that the producer is responsible for the product and should do everything to guarantee it. Consumers are important stakeholders in the food chain and as such they share equal responsibility. Among other things, all consumers should learn to understand and apply the basic rules of food hygiene. They should be able to discriminate between hygienic and unhygienic practices, and participate in improving food safety in the community.

Drinking Water

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Ari Hörman
Finnish Defence Forces, Helsinki, Finland
Mati Roasto
Estonian University of Life Sciences, Tartu, Estonia
Marja-Liisa Hänninen
University of Helsinki, Helsinki, Finland

Introduction

Drinking water is the single most important source of gastroenteric diseases, mainly due to faecal contamination of raw water, failures in the water treatment process or recontamination of the treated drinking water (World Health Organization, 2003; Medema et al., 2003a). Two-thirds of the drinking water consumed world-wide are derived from various surface water sources (Annan, 2000), which may easily be contaminated microbiologically by sewage discharges or faecal loads by domestic or wild animals, while microbial quality may be endangered by various weather conditions. Surface water is an important water source for agriculture and animal breeding. Surface waters are also used to a great extent for leisure and recreational activities, and thus unintended ingestion of microbiologically contaminated water poses a potential health risk (Cabelli et al., 1982; Schönberg-Norio et al., 2004).

International concepts of Hazard Analysis of Critical Control Points (HACCP) (Dewettinck et al., 2001; Howard, 2003) and Water Safety Plans (WSP) by the WHO (World Health Organization, 2004) have been introduced to enable the improvement of drinking water safety and security. WSP include health-based targets, which mean that the microbial risks and adverse health effects in a population through drinking water should be minimised.

Drinking Water-borne Enteric Diseases in Humans

Significant Drinking Water-borne Enteropathogens

Water-borne gastrointestinal infections remain one of the major causes of morbidity and mortality world-wide (World Health Organization, 2002; World Health Organization, 2003). The most important microbes causing infections or epidemics through drinking water include bacteria: *Campylobacter* spp., *Escherichia coli*, *Salmonella* spp., *Shigella* spp., *Vibrio cholera, Yersinia enterocolitica,* viruses such as: adeno-, entero-, hepatitis A- and E-, noro-, sapo- and rotaviruses and protozoa: *Cryptosporidium parvum*, *Dracunculus medinensis*, *Cyclospora cayetanensis*, *Entamoeba histolytica*, *Giardia duodenalis* and *Toxoplasma gondii* (World Health Organization, 2004).

Historically, large water-borne cholera epidemics with numerous casualties in the mid-1800s and the early investigations of cholera epidemics in London by John Snow (1813-1858) and the works of Robert Koch (1843-1910) on *V. cholerae* have remarkably assisted the understanding of the epidemiology and prevention of water-borne diseases (Brock, 1999). World-wide, *V. cholerae* is still a significant cause of water-borne infections, especially in developing countries where most of the victims are children under five years old (World Health Organization, 2002, 2003).

Epidemiological studies of water-borne outbreaks in Finland have indicated that the most important water-borne pathogens in that country are noroviruses (NV; formerly referred to as the Norwalk-like viruses) and campylobacters (Miettinen et al., 2001). During 1998-1999, eight of a total of 14 reported water-borne outbreaks were caused by NV and three by campylobacters (Miettinen et al., 2001). Noroviruses are also the leading cause of gastroenteritis elsewhere in the Western world, causing 60-80% of gastroenteritis outbreaks (Fankhauser et al., 2002; Lopman et al., 2003). *Campylobacter* spp. are the most common bacterial cause of gastroenteritis in the Nordic Countries (Rautelin and Hänninen, 2000).

Enteric parasites such as *Giardia* spp. and *Cryptosporidium* spp. are well recognised as emerging pathogens in drinking water with the ability to cause severe water-borne enteritis even in small doses, especially in immunocompromised individuals (Franzen and Muller, 1999; Szewzyk et al., 2000). *Giardia* spp. and *Cryptosporidium* spp. are common causes of human diarrhoeal diseases in the developed and developing countries (Marshall et al., 1997; Clark, 1999). Outbreaks associated with contaminated drinking water have occurred especially in the USA and the UK. *Cryptosporidium parvum* infected 403,000 people in one of the largest water-borne epidemics ever in Milwaukee, WI, USA in 1993 (Mac Kenzie et al., 1994). During the 1990s, *Cryptosporidium* became one of the most important pathogenic contaminants found in drinking water, due to its low infective dose (Dillingham et al., 2002) and high resistance to the common water disinfectant, chlorine, and environmental factors such as low temperature (Rose, 1997; Fayer et al., 1998; Payment, 1999).

Noroviruses

Human noroviruses, formerly described as Norwalk-like viruses, belong to the genus *Caliciviridae,* together with Sapoviruses. NV are small ribonucleic acid (RNA) viruses, the RNA genome being approximately 7.5-7.7 kb, which provides them with a high degree of genomic plasticity and the capability to adapt to new environmental niches (Radford et al., 2004). The NV were recently divided into five genogroup, genogroups I and II being associated mostly with human infections. Within genogroups, there is a large amount of inherent genetic variability and at least 20 genotypes have been recognised (Radford et al., 2004).

NV infection is typically a violent vomiting disease with a sudden onset, incubation period normally 1-3 days. Besides vomiting, symptoms may include high fever, diarrhoea and headache. The symptoms are considered to be self-limiting and last 2-3 days (Kaplan et al., 1982a). The infective dose for man is very low, as 10-100 virus particles can cause a clinical infection (Schaub and Oshiro, 2000). The virus is excreted in faeces and vomit and the patient may be infective from the incubation period and stay infective for up to 2-3 weeks after symptoms have ended (Okhuysen et al., 1995). NV gastroenteritis is rapidly and effectively spread from person-to-person, especially in near contact (Koopmans et al., 2002). In most cases the NV infection does not require medication, but some severe cases may need hospitalisation and fluid therapy (Kaplan et al., 1982b).

The detection of NV in faecal samples has developed remarkably after the application of molecular methods in virus detection (Koopmans and Duizer, 2004). The most sensitive method to detect NV is reverse-transcriptase polymerase chain reaction (RT-PCR), which is able to detect 1-1,000 virus particles per gram, although less sensitive electron microscopy and enzyme-linked immuno sorbent assays (ELISA) are still also utilised (Koopmans and Duizer, 2004). Before the specific detection methods became available, the causative organism of the most viral epidemics and infections remained unspecified or unsolved. The molecular methods are helpful tools in epidemiological investigations and in tracking infection routes (Maunula et al., 1999).

***Campylobacter* spp.**

Campylobacter enteritis in man is caused mainly by *Campylobacter jejuni* or *C. coli*, which are zoonotic and carried by wild and domestic animals, especially by birds and poultry (Blaser, 1997). The pathogenic potential of *C. jejuni* and *C. coli* was not discovered until the 1970s (Szewzyk et al., 2000). Campylobacters are microaerophilic and survive only few hours at high temperatures (37°C) but can survive at low temperatures (4°C) for several days (Szewzyk et al., 2000). The infective dose of Campylobacters is relatively low, with 800-100,000 ingested organisms needed to cause illness in man (Black et al., 1988). During the 1990s, *Campylobacter*-like organisms, *Arcobacter* spp., were described. These occur in the environment and possess pathogenic potential (Szewzyk et al., 2000).

Campylobacter infection is usually self-limiting and is characterised by diarrhoea, fever and abdominal cramps. The incubation time can vary from 1 to 10 days, but is usually 2-5 days. Diarrhoea may last for 3-5 days, although abdominal pain and cramps may last after that (Blaser, 1997). Campylobacter infection may lead to severe but rare sequelae, reactive arthritis (Hannu et al., 2004), Guillain-Barré syndrome (Kuwabara, 2004) or even myocarditis (Cunningham and Lee, 2003). The risk of developing Guillain-Barré syndrome is very low, less than 1 per 1,000 infections (Kuwabara, 2004). *Campylobacter* gastroenteritis does not usually require any other medication than oral fluid therapy, but antibiotic therapy may be indicated in severe cases.

Diagnosis of *Campylobacter* gastroenteritis is traditionally made by bacterial culture of faecal samples at selective media and isolation and detection of typical colonies. Positive isolates can be further subtyped in various serotypes according to the various antigens, while tests for antibiotic resistance can be applied for subtyping. During recent years, pulsed-field gel electrophoresis has been utilised in typing of *Campylobacter* strains and this has increased the accuracy of epidemiological investigations (Hänninen et al., 1998; Moore et al., 2001; Hänninen et al., 2003).

Giardia spp. and *Cryptosporidium* spp.

The genus *Giardia* comprises six species that can infect several hosts. *Giardia duodenalis* (also referred to as *G. intestinalis* or *G. lamblia*) is infectious for humans but can also cause infections in other hosts (Monis et al., 2003). The spectrum of clinical giardiasis varies from asymptomatic to severe diarrhoea and malabsorption. Acute giardiasis develops after an incubation period of 1 to 14 days (average of 7 days) and usually lasts 1 to 3 weeks. Symptoms include watery, foul-smelling diarrhoea, abdominal pain, bloating, nausea and vomiting. In chronic giardiasis the symptoms are recurrent and malabsorption and debilitation may occur. Occasionally, the illness may last for months, or even years, causing recurrent mild or moderate symptoms such as impaired digestion, especially lactose intolerance, intermittent diarrhoea, tiredness and weakness, and significant weight loss. Giardiasis is diagnosed by the identification of cysts or trophozoites in the faeces, using direct microscopy as well as concentration procedures. Repeated samplings may be necessary; it is sometimes necessary to collect samples for four to five weeks to get a positive laboratory diagnosis. In addition to faecal samples, samples of duodenal fluid or duodenal biopsy may demonstrate trophozoites. Alternative methods for detection include antigen detection tests by enzyme immunoassays and detection of cysts by immunofluorescence. Both methods are available in commercial kits.

The genus *Cryptosporidium* was recently suggested to comprise over 20 species based on morphological, biological and genetic studies (Xiao et al., 2004). These species have several mammalian and non-mammalian hosts and cross-infections may occur between various host species (Dillingham et al., 2002). In humans, cryptosporidiosis was first diagnosed in the late 1970s in immunocompromised individuals, in which *Cryptosporidium* can cause severe, even fatal disease (Marshall et al., 1997). Later, the causal agent *Cryptosporidium parvum* was noted as a global human enteropathogen. *Cryptosporidium parvum* has been genetically divided into human genotype 1 (*Cryptosporidium hominis*) and genotype 2, which also infects cattle (Dillingham et al., 2002). The life cycle of *Cryptosporidium* is more complex than that of *Giardia* and includes an asexual and a sexual stage inside the host's intestine and an infective stage outside the host, the oocyst stage (CDC Division of Parasitic Disease, 2001, 2003; Dillingham et al., 2002).

Symptoms of cryptosporidiosis include diarrhoea, loose or watery stools, stomach cramps, upset stomach and a slight fever (CDC Division of Parasitic Disease, 2003). Some infected individuals have no symptoms. Symptoms generally begin after the 2-10 day incubation period. In individuals with an average immune system, symptoms usually last approximately two weeks. The symptoms may go in cycles in which the individual may seem to recover for a few days, then feel worse, before the illness ends. Although *Cryptosporidium* can infect all people, some groups are more likely to develop more serious illness. People that have a severely weakened immune system, those with human immunodeficiency virus

(HIV), acquired immunodeficiency syndrome (AIDS) or cancer, transplant patients who are taking certain immunosuppressive drugs, and those with inherited diseases that affect the immune system are at risk of more serious disease (Gerba et al., 1996). Symptoms may be more severe and can lead to serious or life-threatening illness.

Testing for *Cryptosporidium* can be difficult and several stool specimens over several days may be needed to detect the parasite. Acid-fast staining methods, with or without stool concentration, are most frequently used in clinical laboratories for detection of *Cryptosporidium* oocysts. For increased sensitivity and specificity, immunofluorescence microscopy and enzyme immunoassay have been taken into use in some clinical laboratories. Molecular methods are mainly applied for research purposes. However, tests for *Cryptosporidium* are not routinely carried out in most clinical laboratories. There is no established specific therapy for human cryptosporidiosis (Marshall et al., 1997). Rapid loss of fluids because of diarrhoea can be managed by fluid therapy.

Surveillance for Drinking Water-related Enteric Diseases

Surveillance of water-borne outbreaks in European countries is generally the responsibility of the local municipal or regional public health authorities. The executive holder of an office is usually a veterinary hygienist or veterinary surgeon specialised in environmental health, while the decision-making multi-member authority is a municipally elected board. The health authorities have a duty to investigate suspected water-borne outbreaks and report them further to provincial and state authorities. When the outbreak has been investigated, the outbreak report is sent to national food or health authorities. The assumption is that large community-based drinking water-borne epidemics are reliably notified and reported, even with delay. However, mild, single or obscure water-borne infections can remain undetected and unreported.

To be able to notify and recognise a water-borne disease, the requirement is that the infected individual develops symptoms and contacts the healthcare provider or seeks medical care. The symptoms and anamnesis may then guide the medical personnel to suspect a water-borne disease and necessary faecal, vomit or other samples are taken. Possible other patients with similar symptoms and anamnesis (time and place of exposure) provide valuable information for outbreak investigation. The World Health Organization defines a water-borne outbreak as an episode of two or more individuals experiencing a similar illness after ingestion of the same type of water from the same source and when the epidemiological evidence implicates the water as the source of the illness (Schmidt, 1995). Adequate samples collected from consumed drinking water at an early stage of the investigation are essential in linking the exposure and outbreak. Obtaining representative samples may be difficult or even impossible after the individual has developed symptoms and contacted healthcare personnel (Hunter et al., 2003a).

One clear reason for underestimation of water-borne epidemics is that not all patients have severe symptoms and seek medical care. Clinical symptoms may be masked by other causes and thus faecal samples will not be analysed for the presence of protozoa. Laboratory analysis may also fail to detect these parasites in faecal samples. Underreporting has also been estimated for viruses (Kukkula et al., 1999; Koopmans and Duizer, 2004). Thus the subclinical, asymptomatic or undetected cases may play a significant role in infection transmission and epidemiology in the general population.

In analysing the epidemiological data from Nordic countries, the estimation is that there will be 4670 (95% CI: 4300-5060) symptomatic cases of *Giardia* and 3340 (95% CI: 3110-3580) symptomatic cases of *Cryptosporidium* annually per 100,000 general population in the Nordic countries (Hörman et al*.,* 2004b). The vast majority of cases will remain unrecorded in the national registers of infectious diseases, since for single recorded cases there will be 254-867 undetected/unregistered cases of *Giardia* and 4072-15,181 cases of *Cryptosporidium*.

Enteropathogenic and Indicator Microbes in Surface Water

Enteropathogens in Surface Water

Enteropathogen microbes are usually adapted only to multiply in humans and other animals, and surface water is only a niche on their circulation pathway in the environment and human or animal population (Medema et al., 2003a).

The occurrence of water-borne enteropathogenic microbes in surface water is associated with faecal contamination of surface water sources (Ashbolt, 2004). Environmental factors have an influence on how enteropathogens survive and move in surface water. Faecal contamination can originate from municipal or domestic sewage discharge or from direct release of faecal material into surface water by domestic or wild animals. Enteropathogenic and other microbes can adhere on soil particles and be carried on them. Exceptional weather conditions such as heavy rains and flooding may increase the faecal load in surface water, lakes and rivers, by moving e.g. sewage, other waste or contaminated soil into water (Kistemann et al., 2002; Auld et al., 2004). Surface run-off after snowmelt can also have an impact on surface water quality. Diffuse and single point pollution sources at the catchment area have a heavy influence on surface water quality in densely populated areas, but remote wilderness waters can also be faecally contaminated and contain human enteropathogens (Welch, 2000; Boulware et al., 2003).

A few systematic studies have been undertaken on the simultaneous prevalence of various enteric pathogens in surface waters. In one systematic study carried out in Finland (Hörman et al., 2004a), a total of 41.0% (57/139) of surface water samples tested positive for at least one of the pathogens analysed: 17.3% positive for campylobacters (45.8% *Campylobacter jejuni*, 25.0% *C. lari*, 4.2% *C. coli*, and 25.0% *Campylobacter* spp.), 13.7% for *Giardia* spp., 10.1% for *Cryptosporidium* spp., and 9.4% for noroviruses (23.0% genogroup I and 77.0% genogroup II). During the winter season the samples were significantly ($P<0.05$) less frequently positive for enteropathogens than during other sampling seasons. No significant differences were found in prevalences of enteropathogens between rivers and lakes.

Possible seasonal or time-related variations in the occurrence of various groups of enteric pathogens in surface water appear to be dependent on the source of contamination and conditions facilitating contaminant discharge into surface water. If the major sources are effluents from sewage water plants that treat human wastes, seasonal patterns similar to those found in human infections for a particular pathogen would be detected in effluents and downstream water samples (Kukkula et al., 1999; Nylen et al., 2002; Hänninen et al., 2005). If the watershed is contaminated from discharges stemming from agricultural runoffs, the highest numbers of zoonotic enteric pathogens would be found during the pasture season after snowmelt, floods and heavy rainfall (Bodley-Tickell et al., 2002).

The major dissimilating factors between seasons in watersheds in Finland and regions with similar climatic conditions are temperature, ice cover and solar radiation (Järvinen et al., 2002). Low temperatures (< 5-10°C) in water during winter and high solar radiation during the summer months (June, July and August) are known to have impact on the survival and recovery of *Campylobacter* spp. In studies in Norway (Kapperud and Aasen, 1992; Brennhovd et al., 1992) and Finland (Korhonen and Martikainen, 1991), campylobacters in natural water exhibited seasonal patterns, the number of positive samples being highest in winter and lowest in summer. *C. jejuni* and *C. coli* survive in cold water, below 10°C, much longer than in water with temperatures exceeding 18°C. A confounding factor in the assessment of seasonality of campylobacters in natural water sources is the faecal load caused by wild birds living in watershed areas and known to be carriers of *C. jejuni*, *C. lari* and *C. coli* (Waldenström et al., 2002; Hänninen et al., 2003).

Recent data reveal that protozoa, *Giardia duodenalis* and *Cryptosporidium parvum* occur in Nordic surface water sources in rivers and lakes and can pose a potential biohazard for drinking water supplies (Robertson and Gjerde, 2001; Rimhanen-Finne et al., 2002). In Norway the prevalence of *Giardia* was found to be 7.5% and that of *Cryptosporidium* 13.5% in water samples taken from water plants and 9.0% and 13.5%, respectively, in raw water samples (Robertson and Gjerde, 2001). The occurrence of *Giardia* and *Cryptosporidium* was significantly correlated with water sample turbidity values ≥ 2 nephlometric turbidity units (NTU) and with a high number of domestic animals in the catchment area (Robertson and Gjerde, 2001).

A few studies are available on the possible seasonality of the intestinal parasites *Giardia* spp. and *Cryptosporidium* spp. in surface waters. Lower numbers of positive samples with these parasites during the cold winter months compared with other seasons have been found in some studies (Wallis et al., 1996). In one study the highest frequency of positive samples for *Giardia* spp. and *Cryptosporidium* spp. was found during autumn and winter in surface waters affected by agricultural discharge due to heavy rains (Bodley-Tickell et al., 2002), but no clear seasonality has been found in some other studies (Robertson and Gjerde, 2001).

Indicator Microbes and Water Quality

Since the analysis of various enteropathogens can be laborious and can require special analytical techniques, there have been strong efforts to find or develop an overall indicator of hygiene quality. Already in the late 1800s, a concept of a total heterotrophic plate count was used to assess drinking water quality and >100 bacteria in a 1 ml sample was noted as unacceptable (Bartram et al., 2003; Medema et al., 2003a). The United States Environmental Protection Agency (US EPA) has suggested that the HPC should not exceed 500 cfu/ml, although it has been estimated that HPC bacteria do not represent a significant exposure of total bacteria in the average diet in the US (Stine et al., 2005). The absence of a correlation between heterotrophic plate count (HPC) and pathogenic microbes has been found in most studies (Edberg and Smith, 1989; Bartram et al., 2003) and HPC is no longer used as a faecal indicator of drinking water quality (World Health Organization, 2004).

To reliably assess faecal contamination of water and thus the possibility for occurrence of enteropathogenic microbes other indicators have been proposed, amongst the earliest *E. coli* (Ashbolt et al., 2001). The criteria for a microbial indicator of drinking water quality and faecal contamination include: 1) the indicator should be absent in unpolluted water and present when a source of pathogenic microorganisms is present, 2) the indicator should not multiply in the environment, 3) the indicator should be present in greater numbers than the pathogenic microorganisms, 4) the indicator should respond to natural environmental conditions and water treatment processes in a manner similar to the pathogens, and 5) the indicator should have methods for isolation, identification and enumeration (Medema et al., 2003a).

Total coliform and *E. coli* counts are used world-wide as indicators of faecal contamination of drinking and recreational bathing water (Edberg et al., 2000; Havelaar et al., 2001; Rompre et al., 2002). The focus of debate has concerned the suitability of these organisms as indicators of water quality and contamination, since pathogens may be present in drinking water without the presence of coliforms or *E. coli* (Payment et al., 1991). In addition, some *E. coli* strains have been isolated from surface and industrial waste water without any connection to faecal contamination (Niemi et al., 1987). The correlation between the actual counts of coliforms or *E. coli* and the presence of pathogens has been studied extensively and any direct correlation is weak or non-existent (Grabow, 1996).

In addition to coliforms and *E. coli*, other organisms have also been proposed as indicators of the hygiene quality of drinking and bathing water, e.g. faecal enterococci, sulphite-reducing clostridia, *Clostridium perfringens* and bifidobacteria (Barrell et al., 2000; Ashbolt et al., 2001). Bacteriophages such as somatic coliphages, F-RNA bacteriophages, or phages of *Bacteroides fragilis* have also been proposed as indicator organisms especially suitable for assessment of viral contamination (Payment and Franco, 1993).

To obtain reliable data on a specific enteropathogen in a surface water source, this enteropathogen has to be specifically investigated with adequate sampling and analysis. Inadequate sampling may lead to failure to detect otherwise present pathogenic and indicator organisms, e.g. lack of sodium thiosulphate has been reported to cause false negative *Legionella* and HCP results in chlorinated water samples (Wiedenmann et al., 2001). The ecology and environmental survival characteristics of bacterial, viral and parasitic enteropathogens vary, revealing that most probably no single indicator organism can predict the presence of all enteric pathogens. Furthermore, whether a true correlation exists between the indicator organisms generally used and pathogens, and the extent and circumstances in which these organisms can be used as reliable determinants in water hygiene, have been discussed (Edberg et al., 2000; Tillett et al., 2001; Leclerc et al., 2001). However, *E. coli* is still considered to be superior as an indicator of faecal contamination and the hygiene quality of drinking water (Edberg et al., 2000). *E. coli* is abundant in human and animal faeces – in fresh faeces it can occur at concentrations of 10^9 colony forming units (cfu)/g (Payment et al., 2003). To some extent coliforms or *E. coli* can also be used as process indicators when water treatment processes and water purification devices are tested (Grabow et al., 1999).

Coliform bacteria are defined as Gram-negative, non-spore forming, oxidase-negative, rod-shaped facultatively anaerobic bacteria that ferment lactose with β-galactosidase to acid and gas within 24-48 h at 36 ± 2°C (Ashbolt et al., 2001). Thermotolerant coliforms are coliforms that produce acid and gas from lactose at 44.5 ± 0.2°C within 24 ± 2 h and *E. coli* are thermotolerant coliforms that produce indole from tryptophan at 44.5 ± 0.2°C (Ashbolt et al., 2001). *E. coli* have also been defined as thermotolerant coliforms producing indole and as coliforms producing β -glucuronidase (Ashbolt et al., 2001).

Detection and counting of total coliforms and *E. coli* have traditionally been based on the multiple tube fermentation method, using the most probable number (MPN) estimation of the bacterial count or membrane filtration (MF) methods (Ashbolt et al., 2001; Rompre et al., 2002). The reference method used in the European Union (The Council of the European Union, 1998) for detection of *E. coli* in drinking water samples is MF method ISO 9308-1:2000 (International Organization for Standardization, 2000) based on cultivating the membrane filter on lactose Tergitol-7 (LTTC) agar.

Since traditional cultivation-based methods require a minimum of 24 h of incubation followed by a confirmation procedure lasting 24-48 h, the need for rapid test methods has increased, especially in the water industry and in emergency situations (International Water Association, 2000). During recent decades new chromogenic or fluorogenic, defined substrate-based methods on β-galactosidase (total coliforms) or β-glucuronidase (*E. coli*) and ready-made culture media have been introduced and numerous comparative studies have shown these tests to give results comparable to those of the MF LTTC or m-Endo Agar LES methods (Edberg and Edberg, 1988; Clark and el Shaarawi, 1993; Ashbolt et al., 2001). Due to differences in the test principles, the outcome of different test methods may vary in the numbers of organisms detected and the tests may also detect metabolically different types of organisms (Ashbolt et al., 2001). One explanation for this is the apparent difference in sensitivity and specificity due to various selective or confirmation components used in test media or procedures, e.g. the production of indole versus β-glucuronidase used in *E. coli* detection.

The occurrence of enteropathogens in surface waters is linked directly to possible contamination sources, while environmental conditions affect only the survival of these microbes in water. The presence of traditionally used faecal indicators, including thermotolerant coliforms and *E. coli*, has significant predictive value for the presence of the enteropathogens studied but no significant correlation has been found between a certain cfu level of indicators and the presence of pathogens. Microbial monitoring of raw water using only faecal indicator organisms is not sufficient for assessment of the occurrence of a particular enteropathogen (Hörman et al., 2004b).

Microbiological Requirements for Drinking Water Quality

Drinking water or water intended for human consumption is defined in the European Union legislation as all water intended for drinking, cooking, food preparation or other domestic purposes or water used in food production (The Council of the European Union, 1998). In most developed countries, drinking water is ranked as food, and high standards are set for its quality and safety (Szewzyk et al., 2000)

The WHO has established revised guidelines for drinking water quality (World Health Organization, 2004). These guidelines can be applied into national standards and legislation taking into account e.g. the national climatic, geographical, socio-economic and infrastructural characters, as well national health-based targets. The national legislation regulating the drinking water quality in member states of the European Union is implemented from EU Directive 98/83/EC (The Council of the European Union, 1998). This Directive and national legislation follow the guidelines given by the WHO. In general, water intended for human consumption "must be free from any micro-organisms and parasites and from any substances which, in numbers or concentrations, constitute a potential danger to human health" at the point of compliance (The Council of the European Union, 1998). Although not stated in the EU Directive, to fulfil this requirement a risk assessment for microbiological hazards must be carried out in a particular drinking water production process or plant. The specific parametric values for microbiological quality require that *E. coli* or *Enterococci* may not be detected in a 100 ml sample using the accepted detection methods. Similar requirements are in effect outside the EU (Havelaar et al., 2001).

European legislation sets requirements for the quality of surface water intended for the abstraction of drinking water (The Council of the European Communities, 1975). The legislation give instructions for the minimum treatments required for production of drinking water from surface water according to the surface water quality. Surface waters are divided into three quality categories judged by various microbiological and physico-chemical parameters. Microbiological parameters include numbers of coliforms and thermotolerant coliform bacteria, faecal *streptococci* and *Salmonella* spp. in water samples.

Water Treatment

General

The general purpose of water treatment is to make water potable by removing or inactivating the pathogenic organisms and toxins from drinking water entirely or to a level where no harmful effects will occur to the consumer (Backer, 2002). In terms of terminology, disinfection is a process where harmful microbes are inactivated, chemically or physiologically, and purification refers to removal of harmful substances from drinking water. The terms treatment, disinfection and purification are commonly used interchangeably. In general, the purpose of drinking water treatment is not to sterilise the water but only to destroy or remove harmful microbes and substances (Backer, 2002).

The concept of multiple barriers is essential in water treatment, since a single treatment method is only capable of removing or inactivating all different types of pathogenic microbes under all conditions in exceptional cases (Stanfield et al., 2003; LeChevallier and Au, 2004). In practice, the multiple barrier concept means a combination of two or more treatment methods or steps in drinking water production. Having multiple barriers limits the possibility of harmful microbes or toxins entering drinking water due to a failure in one of the treatment steps (World Health Organization, 2004). The multiple barrier concept can also contain steps beyond the actual treatment process, e.g. selection of the best possible raw water source and protection of the treated water (LeChevallier and Au, 2004).

Thermal Treatment

Thermal treatment, in practice letting the water come to a rolling boil (at 100 °C) for 1-3 minutes, is the oldest means of disinfecting water and is a simple way to treat smaller (less than few tens of litres) amounts of water in the field and emergency conditions where a heat source is available (Backer, 2002). The 'boil water' advice is also common practice in communities where treated drinking water is suspected to have been contaminated or is experiencing temporary quality problems. Intervention studies at population level in developing countries have shown that boil water campaigns improve the quality of drinking water and reduce the incidence of childhood diarrhoea (Mclennan, 2000). Heating water until 'too hot to touch', which is approximately 60°C or less, is inadequate for safe drinking water purposes (Groh et al., 1996).

The destructive effect of heat on microbes is based on the irreversible denaturation of genetic DNA or RNA molecules and intracellular proteins. In practice, all vegetative bacteria, protozoa and viruses start to be inactivated at temperatures above 50-60°C, with the final inactivation depending on the given temperature and its duration. Heat inactivation of microbes is exponential and thermal death is reached in less time at higher temperatures (Backer, 2002). Some mathematical models are designed to estimate the thermal inactivation (Lambert, 2003). At a temperature of 100°C, all pathogenic vegetative bacteria, protozoa and viruses are destroyed and only microbial spores, e.g. spores of *Clostridium* and *Bacillus*, and some heat-resistant toxins, e.g. some cyanobacterial toxins, survive or maintain their toxicity (Backer, 2002).

Studies on coliform and thermotolerant coliform bacteria, *E. coli*, *Salmonella* Typhimurium and *Streptococcus faecalis*, in water have shown that 1000-fold (3 $\log_{10}$ units) inactivation is reached once water is heated to 65°C (Fjendbo Jorgensen et al., 1998). *Vibrio cholera* is inactivated at 60°C in 10 min and at 100°C in 10 s (Rice and Johnson, 1991). Another experiment showed no inactivation of *E. coli* viability when the temperature was 50°C but 5 min at 60°C, 1 min at 70°C and any time at 100°C destroyed *E. coli* totally (Groh et al., 1996). *Giardia* is reported to be destroyed when water is heated at 72°C for 10 min (Ongerth *et al.*, 1989) and *Cryptosporidium* at 72°C for over 1 min (Fayer, 1994). Hepatitis A virus is inactivated totally at 98°C for 1 min (Krugman et al., 1970) and caliciviruses by 3 $\log_{10}$ units at 71.3°C for 1 min (Duizer et al., 2004). Inactivation of a heat-sensitive BoNT has been shown to be effective when water is heated at 80°C for 30 min (Josko, 2004).

For some purposes water may be needed to be distilled, i.e. the water molecules are transformed at boiling temperature from the liquid into the gaseous phase and separated from the remaining liquid and substances. This is a method to produce pure water and the temperature at normal air pressure is also effective against microbes and heat-sensitive toxins. Vacuum distillation is a method to distil the water under negative pressure; the temperature needed to boil the water may be as low as 50°C. This method is used especially to produce drinking water from salty sea water, but it is not considered as effective against pathogenic microbes.

Chemical Treatment

Chemical treatment of drinking water includes the use of various forms of halogens, chlorine or iodine, silver or ozone. All of these can be used in field conditions, although the generation of ozone requires technical equipment. Chemical treatment is the only method that ensures some protection for treated drinking water after the treatment. The efficiency of chemical treatment is a function of dose, contact time, temperature and pH (Stanfield et al., 2003). The practical application of this is the concentration-time (CT) concept, which is a product of the residual chemical concentration in mg/l and the contact time in min (Stanfield et al., 2003). The antimicrobial effect of a chemical depends on microbe susceptibility; a given CT value can be applied when a required inactivation of a certain microbe in log10 units is estimated. The treatment efficiency of all chemicals is reduced by solvents in the water, since a proportion of the added chemical (also referred to as chemical demand) is bound to solvents and cannot act against microbes; only the free residual chemical is effective in microbe inactivation. All chemicals have their best efficiency at moderate temperatures, at 15-20°C and at pH 6-9 (Backer, 2002). Besides their antimicrobial effects, chemicals can also oxidise and remove some harmful chemicals from drinking water.

The use of chlorination of drinking water was first invented at 1800 but it took several decades until chlorination came into widespread use for water treatment in the early 1900s, after which it dramatically reduced waterborne outbreaks (Beck, 2000). Today chlorination is the most widely used water chemical treatment for inactivation of pathogenic microbes. Chlorination can be applied in the form of liquefied chlorine gas, sodium hypochlorite solution or calcium hypochlorite granules, sodium dichloroisocyanurate, chloramines and chlorine dioxide, each of which have different disinfection properties (Stanfield et al., 2003; World Health Organization, 2004). Chloramine has a lower disinfection activity than chlorine but is more stable. Chlorine dioxide has greater effectiveness against protozoa but is not as stable as chlorine. The wide use of chlorination has raised the question of possible side-effects and chlorine has been shown to form mutagenic compounds when reacting with organic material, especially with humic acids. However, the benefits of chlorination of drinking water have been estimated to greatly exceed the negative side effects of by-products (Ashbolt, 2004). The formation of by-products can be minimised by filtering cloudy water before chlorination and using adequate rather than excessive, concentrations of chlorine chemicals (World Health Organization, 2004).

In general, chlorination is effective against bacteria and viruses but less effective against protozoa and algae in concentrations normally used in drinking water, e.g. 1-5 mg/l. Besides the use for drinking water treatment, chlorination can also be used as shock chlorination at high doses, 10-50 mg/l, for disinfecting drinking water pipelines or storage tanks. Another halogen, iodine, can also be used as a water treatment chemical and it acts mainly similarly to chlorine but has some physiological concerns, e.g. effects on thyroid, potential toxicity and allergenity (Backer and Hollowell, 2000). However, in short-term use iodine is considered to be safe except for individuals with thyroid dysfunction or iodine allergy or pregnant females. One of the benefits of iodination is its more acceptable taste compared with chlorination. Iodine, like chlorine, is also applied to products for use in emergency and field conditions (Gerba et al., 1997). Some CT values of chlorination and iodination against

Table 35.1. CT values for halogens of chlorine and iodine to achieve a 2 $\log_{10}$ unit (99%) reduction in counts of various microbes or concentration of various chemicals in water at pH 6-9.

Halogen type, organism	*Conditions during experiment*				Reference
	Concent-ration mg/l (ppm[a])	Contact time min	CT-value min*mg/l	Temperature °C	
Chlorine dioxide					
Escherichia coli			0.18	20	LeChevallier et al., 1988
			0.38	15	LeChevallier et al., 1988
Chlorine					
Bacillus antracis	1.02	60	60	25	Rose et al., 2005
	0.88	216	190	5	Rose et al., 2005
Campylobacter spp.	0.3	0.5	0.15	25	Blaser et al., 1986
E. coli	0.1	0.16	0.016	5	White, 1992
Calicivirus (CaCV48)	300	10	3,000[b]	20	Duizer et al., 2004
Hepatitis A virus	0.5	5	2.5[b]	5	Sobsey, 1975
Human norovirus	5-6	30	150-180[c]	25	Keswick et al., 1985
Giardia spp. cysts	2.5	60	150	5	Rice et al., 1982
Cryptosporidium spp. oocysts	10	720	1,440	20	Carpenter et al., 1999
	80	90	7,200		White, 1992
Clostridium botulinum neurotoxin	5	30	150[b]		Wannemacher et al., 1993
Iodine					
E. coli	1.3	1	1.3	2-5	Backer, 2002
Giardia spp. cysts	4	120	480	5	Fraker et al., 1992
	13	20	260	20	Gerba et al., 1997
Cryptosporidium spp. oocysts	13	240	3,120[d]	20	Gerba et al., 1997

[a] ppm, parts per million of free residual halogen.
[b] End point > 3 or 4 $\log_{10}$ units reduction.
[c] Result obtained from clinical responses on volunteers, one of eight developed clinical symptoms.
[d] End point < 1 $\log_{10}$ units reduction, iodine considered not effective against *Cryptosporidium* spp. oocysts.

various microbes and chemical compounds are presented in Table 35.1.

Silver ions have bactericidal effects at low doses (≤ 100 parts per billion), but the effect is strongly affected by adsorption onto the surface of any container, as well as by any substances in water. Data on the effect of silver ions on viruses and cysts is scant (National Academy of Sciences, 1980). Therefore the use of silver ion products is better suited as a water preservative for previously treated water, not for disinfection of surface water (Backer, 2002). Silver ions are used in many filter devices as a coating to reduce bacterial growth on filter media (Backer, 1995).

Ozone is a powerful oxidant and effective against bacteria, viruses and even protozoa. In general, CT values needed to reduce microbes are much lower than those of chlorine or iodine, e.g. the CT value for reducing *Giardia* cysts by 2 $\log_{10}$ units at 5°C is 0.5-0.6 (LeChevallier and Au, 2004). Besides microbes, ozonation is also effective against cyanobacterial toxins, such as microcystins, at concentrations of 1.5 mg/l for 9 min (Hoeger et al., 2002; LeChevallier and Au, 2004). However, the production of ozone requires special technical equipment and therefore ozonation is not readily available for small-scale water treatment in field conditions, but large-scale mobile water treatment plants

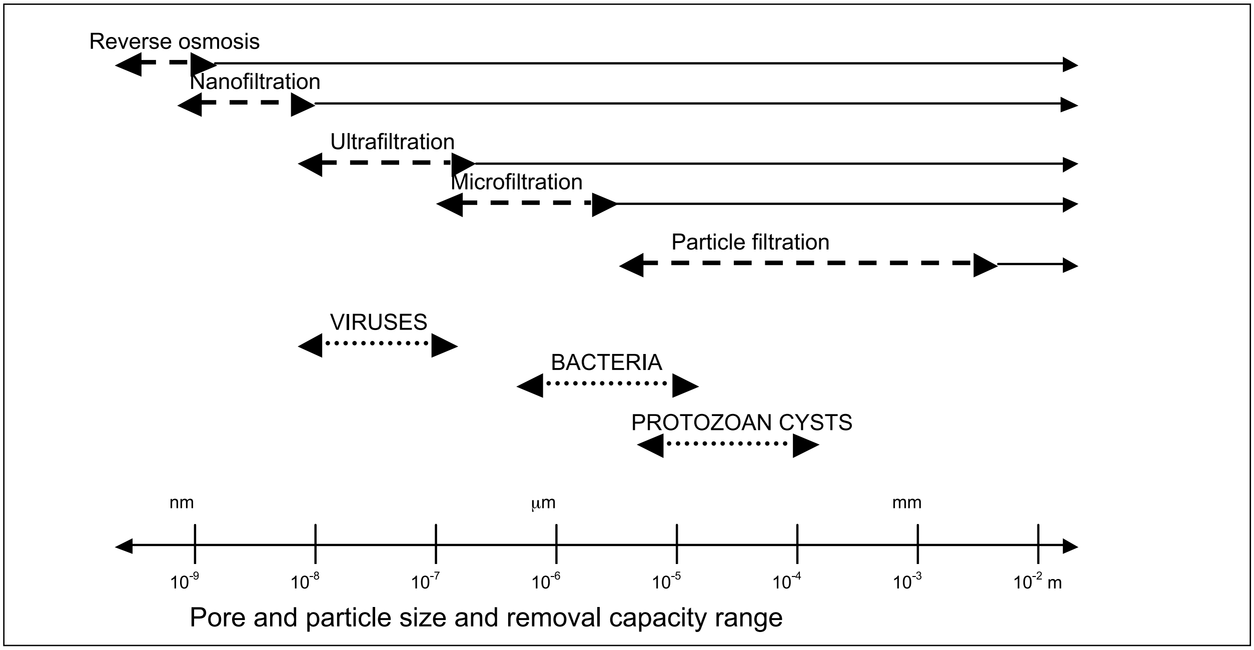

Figure 35.1. Pore size (dashed lines) and range of removal capacity (solid lines) of various filtration methods and general size range of microbial particles (dodded lines).

have been constructed. Another limitation for ozonation in field conditions is that to achieve the full effect the ozone needs a prolonged reaction time besides the actual contact time (Hoeger et al., 2002).

Filtration

Filtration is a physical removal method of organisms and other particulate matter from drinking water based on particle and sieve size. The various filtration methods (Figure 35.1) have their own effective removal range according to the pore size of the filter medium (Stanfield et al., 2003; LeChevallier and Au, 2004). Particle (also referred as granular media) filtration is the most widely used filtration process in drinking water treatment, usually combined with coagulation, flocculation and sedimentation. The filter medium is usually fine-grained sand or other similar material. Sand filtration has been shown to be effective in removal of bacteriophages and the cyanobacterial toxin microcystin in some experiments (Rapala et al., 2002). For field conditions there are no commercial sand filters but an experimental sand filter can easily be constructed from e.g. a bucket and fine-grained, heated and washed sand.

A primitive filter can be made e.g. from woven fabric, the removal efficiency being dependent on fabric layers, density and material. In India, four-times folded sari fabric was shown to reduce the *V. cholera* counts in water by 99% (2 $\log_{10}$ units), probably because *V. cholera* was attached to plankton particles (Huo et al., 1996). Simple particle filters are especially useful in primitive conditions where cloudy, organic material-containing surface water is treated to reduce solvents and turbidity, e.g. prior to the chemical treatment.

Besides particle granular media, the filtration medium can be ceramic or a special membrane. Ceramic filters, the first of which were invented during the late 1800s (Beck, 2000), are usually filter devices which with a mechanical pressure or gravity force the water through a porous filter medium. The removal capacity of ceramic filters is usually ≥ 0.2 µm according to the material. A smaller pore size and thus removal of smaller particles can be achieved using membrane technology. The pore size utilised in ultra- and nanofiltration and reverse osmosis (RO) is such that passage of water molecules and severance from remaining substances is achieved by high pressure in the range 15-50 atm (1.5-5 x 10^3 kPa) (World Health Organization, 2004). The number of sporadic cryptosporidiosis cases was observed to decline in 1996-2002 in England in two districts where membrane filtration was installed (Goh et al., 2004) Various membrane filtration methods are effective in removal of micro-organisms but contamination and microbial fouling of filter media can lead to a breakthrough of organisms and failure in water treatment (Daschner et al., 1996).

The RO technique is effective for removal of monovalent ions and organic compounds of size molecular weight > 50 (World Health Organization, 2004). Reverse osmosis is the most common application for desalination of seawater. There are multiple membrane filtration devices commercially available for field conditions.

Other Treatment Methods

Ultraviolet (UV) radiation can be categorised as UV-A, UV-B, UV-C or vacuum-UV, with wavelengths between 40-400 nm. UV-B and UV-B are effective against microorganisms and the maximum effectiveness is approximately 265 nm (LeChevallier and Au, 2004). The permeability of the UV radiation is reduced by substances, e.g. organic material and humic acids (Huovinen et al., 2000), in water and for cloudy waters treatment with UV radiation is considered not effective (LeChevallier and Au, 2004). The dose of UV radiation is calculated as the total amount of UV energy incident on a certain area in a certain period of time. The units of UV dose are joules per unit area (J/cm^2 or J/m^2), which is defined as the irradiance rate of the UV radiation (in Watts) multiplied by the time the material is exposed to such radiation (in seconds) per unit area. The limiting factor for use of UV radiation in field conditions can be the lack of a source of electricity. In primitive conditions solar UV radiation can be utilised for drinking water treatment, e.g. by exposing water bottles to direct sunlight (McGuigan et al., 1998).

Typical UV (250-275 nm) doses for a 4 $\log_{10}$ unit reduction in bacteria in laboratory experiments with clear water (turbidity < 1 NTU) range from 30 J/m^2 for *Vibrio cholera* to 80 J/m^2 for *E. coli* (LeChevallier and Au, 2004). For virus inactivation higher doses are required; e.g. for animal caliciviruses 340 J/m^2, human rotavirus 500 J/m^2 and human adenovirus 1210 J/m^2 (Duizer et al., 2004). For inactivation of *Giardia* cysts a UV dose of 10 J/m^2 has been shown to be effective (and 20 J/m^2 for *Cryptosporidium* oocysts; Linden *et al.*, 2001, 2002), although in one surface water pilot study 500 J/m^2 was needed to destroy 3.9 $\log_{10}$ units *Cryptosporidium* oocysts (Betancourt and Rose, 2004).

Activated carbon is used as a compound e.g. in water filters, usually in either powdered or granular form (World Health Organization, 2004). Activated carbon is produced by the controlled thermal treatment of carbonaceous material, e.g. wood. The activation produces a porous material with a large surface area and a high affinity for organic compounds (World Health Organization, 2004). The activated carbon loses its ability to absorb compounds once saturated; the carbon can be reactivated by thermalisation. Activated carbon is used for removal of taste and odour compounds, cyanobacterial toxins and other organic chemicals (World Health Organization, 2004). The removal of microbes is only minimal through adhesion on the surface of activated carbon (Backer, 1995).

Concepts for the Microbial Risk Assessment and Management of Drinking Water

Quantitative Microbial Risk Assessment

The aim of the quantitative microbial risk assessment (QMRA) approach is to calculate the risk of disease in the population from what is known, or can be inferred, about the concentration of particular pathogens in the water supply and the infectivity of those pathogens to humans (Hunter et al., 2003a). The formal steps involved in QMRA are: 1) problem formulation and hazard identification, 2) dose-response analysis, 3) exposure assessment and 4) risk characterisation. Making a reliable QMRA for a certain pathogen in a certain drinking water supply and for a given population requires knowledge of the concentrations of the pathogen in source water and the removal or inactivation efficiency of the treatment process and consumption of drinking water and any special characteristics in a population. Some published QMRA studies have estimated quantitative data for *Cryptosporidium* in surface water treatment (Medema et al., 2003b).

QMRA in relation to drinking water has several practical benefits: 1) It can predict the burden of water-borne diseases in the community, under outbreak and non-outbreak conditions; 2) it helps to set microbial standards for drinking water supply; 3) it can identify the most cost-effective option to reduce microbial health risks; 4) it helps to determine the optimum treatment of water; and 5) it provides a conceptual framework to understand the nature and risk from water and how those risks can be minimised (Hunter et al., 2003a).

Hazard Analysis of Critical Control Points (HACCP) and Water Safety Plans (WSP)

The Hazard Analysis of Critical Control Points (HACCP) approach was first introduced for food production with the aim of producing safe food for astronauts, but the framework has also been found to be acceptable for the risk management process in the water supply (Dewettinck et al., 2001; Howard, 2003). The formal principles for HACCP are: 1) Identification of hazards and preventive measures; 2) identification of critical control points; 3) establishment of critical limits; 4) identification of monitoring procedures; 5) establishment of corrective action procedures; 6) validation and verification of the HACCP plan; and 7) establishment of documentation and record-keeping.

The approach to assess and manage the risks in drinking water production related to HACCP consists of the water safety plans (WSP) introduced by the WHO (World Health Organization, 2004). WSP draw on many of the principles and concepts from other risk management approaches, in particular from the multi-barrier approach and from HACCP. The general principles of WSP should be developed and implemented for individual drinking water systems. The key steps of WSP are similar to those of HACCP. For drinking water supply in emergency or field conditions, the principles can be applied but establishment of a full-scale WSP may not be realistic (World Health Organization, 2004). However, the principles provide a suggestive framework for assessing and managing microbial risks in every circumstance.

Acceptable Risk

The purpose of drinking water treatment and drinking water hygiene is to minimise the adverse health effects of hazards on the consumer. However, in practice it is completely unachievable to reduce the risks to zero in all circumstances (Hunter and Fewtrell, 2001). Therefore some risk must be accepted or tolerated and several approaches can be applied to estimate what the acceptable risk is in a given situation. A risk may be acceptable when: 1) It falls below an arbitrary defined probability; 2) it falls below some level that is already tolerated; 3) it falls below an arbitrary defined attributable fraction of total disease burden in the community; 4) the cost of reducing the risk would exceed the costs saved or also the costs saved when the costs of suffering are also factored in; 5) the opportunity costs would be better spent on other, more pressing, public health problems; 6) public health professionals say it is acceptable; 7) the general public say it is acceptable (or they not say it is not);or 8) politicians say the risk is acceptable (Hunter and Fewtrell, 2001). Each of these approaches could lead to a different definition of the acceptable risk, even in the same population.

The acceptability of a risk is dependent on the given population, circumstances and time; a risk accepted somewhere is not necessarily acceptable everywhere else. In field conditions it can be acceptable e.g. to take a risk of getting diarrhoea from unsafe drinking water if there is a risk of getting even more severe health effects due to lack of water. An opposing example could be the near zero tolerance to getting a disease from drinking water for astronauts or strategic pilots during field operations.

The United States Environmental Protection Agency (US EPA) requires the use of *Giardia* as a reference organism, with a microbial risk of less than one infection per 10,000 persons annually (Macler and Regli, 1993). The logic behind this requirement is that *Giardia* is more resistant to drinking water disinfection than other microbial pathogens. The US EPA is so far the only authority in the world to have quantitatively defined the acceptable microbial risk for drinking water (Hunter and Fewtrell, 2001).

Agricultural Terrorism
The US Perspective

36

John A. Herrmann
University of Illinois at Urbana-Champaign, Urbana, IL, USA

Introduction

Throughout history, natural disasters ranging from droughts and floods to untimely frosts and unmitigated heat have routinely reduced production of the safe and inexpensive food that is taken for granted in the United States. However, the terrorist attacks on the World Trade Center and other American targets in 2001 caused policy-makers to consider whether American agriculture was also susceptible to malicious acts of terror.

American agriculture could be a particularly inviting target. Not only could intentional contamination of the food supply or a natural disaster affect the health of humans, plants and animals, it could also have profound and lasting effects on the agricultural economy and compromise consumer confidence. In 2006, an outbreak of human illness caused by *E. coli* O157:H7 associated with spinach grown in the state of California led to 183 cases and one death in 26 states (MMWR, 2006). The spinach disaster was followed by an outbreak of *Salmonella* traced to contaminated tomatoes, which affected almost 200 people in 21 states (Wood, 2006). Another *E. coli* outbreak associated with shredded lettuce at fast food restaurants affected 152 individuals a few months later (USFDA, 2008).Following national recalls by the United States Food and Drug Administration (US FDA), consumer confidence in leafy greens plummeted, sales dropped, shares of publicly traded grocery store stock declined in value, produce had to be destroyed and losses to growers and processors approached an estimated $50-100 million.

Two months after these outbreaks, consumer purchasing of leafy greens was still significantly below pre-outbreak levels (Cuite et. al, 2007) If consumer confidence is shaken by food-borne illnesses not associated with intentional contamination of food, then it is reasonable to assume that a directed attack against agriculture, well planned and executed, would cause significant and long-lasting economic and social impacts in addition to any ensuing human, plant and animal diseases.

This chapter briefly discusses the history of agroterrorism in America, the potential impact of agroterrorism on the US economy and how policy-makers have responded to the likelihood of an attack on US agriculture.

What is Agroterrorism?

Agroterrorism has been described as:

> *"The deliberate introduction of a disease agent, either against livestock or into the food chain, to undermine socioeconomic stability and/or generate fear."(Chalk, 2003)*

and:

> *"Agroterrorism is a subset of bioterrorism, and is defined as the deliberate introduction of a plant or animal disease with the goal of generating fear, causing economic losses, and/or undermining social stability."(Monke, 2007)*

Although these and similar definitions have been accepted for some time in topical discussions, both descriptions accentuate plant and animal disease but do not mention water, human health or food safety. A more inclusive definition of agroterrorism proposed in this chapter is the "*intentional introduction or threat of introduction of biological, chemical or physical agents into agricultural production or processing systems that diminish human, animal or plant health, compromise public confidence in the safety of food and water, generate economic losses for the agricultural sector or undermine social stability*."

The Importance of US Agriculture

The agriculture sector of the United States economy directly employs only 2% of the nation's workforce but, overall, approximately one in six US workers, ranging from producers to processors, shippers and grocery shops to restaurant workers, are associated with food and fibre production. (USDA, 2005) Estimates from 2002 indicate that agriculture contributes $1.3 trillion dollars (USD), or 11%, to the gross domestic product. Gross farm sales exceed $200 billion dollars annually, with production almost evenly split between livestock and plants.

As of 2003, the United States produced more than 42% of the world's maize, 35% of the world's soybean and 12% of the world's wheat. A significant proportion of US agricultural production (21%) is exported, with crops (22%) outpacing livestock (10.5%). These products made up 8% of all US exports ($60 billion USD) in 2003, which more than balanced the agricultural imports of 4% ($47 billion USD)(US Census Bureau, 2004).

Because of production surpluses and government policy supports, the average family in the United States spends only an estimated 5.8% of disposable household income on food, whether it is prepared at home or eaten away from home. Estimates from the European Union suggest that 7.2% of disposable household income is spent on food in its member countries. The world average for household spending for food is 15% for most developed nations and over 30% for many developing nations (USDA, 2006). Because US citizens are used to having access to safe, inexpensive food and water, any disruption in US agricultural productivity would severely shake consumer confidence and could lead to significant social unrest.

The cost of recovering from natural disasters or intentional agricultural terrorism directed at plants, animals or production and distribution systems would be much higher than that associated with the loss of product alone. Agroterrorism, because it is a directed and purposeful activity designed to create maximum damage to targets, could have much higher costs than randomly occurring natural outbreaks. The immediate loss of tangible goods and consumer confidence in safe and wholesome food and declines in domestic food purchases would be followed by declines in the value of publicly traded stocks in agriculture and food-based industries and loss of domestic and international markets, agricultural and non-agricultural alike. The Foot and Mouth Disease (FMD) outbreak in the United Kingdom in 2001 illustrated the extent of collateral economic damage associated with an agricultural disaster. Due to that episode, costs to agriculture were estimated at £3.1 billion. Additional costs to the tourism and hospitality industries were estimated at between £2.7 and £3.2 billion (Thompson et al., 2002).

Effects of Agroterrorism, Natural Disasters and Disease Outbreaks on Agriculture in the United States

Agroterrorism

Attacks against agricultural production systems and water supplies are not new and have been employed throughout the world for many years. Over the past forty years, the United States has experienced numerous incidents of intentional contamination of food, feed and water. Although the following list is comprised of events that did not involve recognised terrorist groups or states, the events indicate how easy it has been to contaminate food, feed or water.

In 1970, 30 cows were found dead on a farm owned by a Black Muslim group in Ashville, Alabama. Analysis of a pinkish-white material found on rocks in the stream from which the cows drank was identified by a local veterinary surgeon as cyanide. It was alleged that the local racist group, the Klu Klux Klan, might have been responsible (New York Times, 1970).

One of the earliest proven attempts to disrupt civil society with bioterrorism in the United States occurred in 1972, when a group of college students who belonged to a neo-nazi group apparently cultured and then planned to put 30-40 kg of typhoid bacteria into Chicago's drinking water system. Fortunately, had the group been successful in delivering the bacteria into the water system, the bacteria would have been killed by normal water chlorination and no illnesses would have been expected. In less developed countries, however, the plan, if carried out, could have been effective in inducing human illness. In addition, the incident made authorities aware of how easy it was to culture pathogens in a school laboratory (Kellman, 2001).

In 1984, upset about local zoning board decisions that had gone against them, followers of the Indian guru Bhagwan Shree Rajneesh tested a plan to influence a local election by spiking salad bars with *Salmonella* Typhimurium at 10 restaurants and one grocery store in the town of The Dalles, Oregon. The cult members had hoped to incapacitate enough voters through food poisoning the day before a local county election so that their own candidates would win and would eventually reverse the previous zoning board decisions. At least 751 persons became sick with gastrointestinal illness from what local public health officials thought was a naturally occurring contamination. It was only when the cult leader admitted one year later to purposefully contaminating the salad bars that the plan was revealed. Fortunately, the exercise did not make enough people sick, so the plan to influence the November election was abandoned (Carus and Tucker, 2001).

In 1999, a Wisconsin man was indicted for contaminating, on two occasions, consumer products intended as ingredients in animal feed processed by National By-Products, Inc. The contamination with the pesticide chlordane forced the company to recall products and to destroy raw material, leading to losses exceeding $2.5 million. Chlordane, a potent pesticide, is linked to cancer in humans. On 13 April 2000, a federal jury returned a guilty verdict to the charge of tampering and sentenced the defendant to three years' incarceration (USFDA, 2000). In early 2003, a disgruntled employee at a supermarket in Michigan intentionally contaminated over 200 pounds of minced beef with a nicotinic acid-based insecticide. Approximately 100 people in the community eventually became ill with symptoms of nicotine poisoning. The perpetrator was indicted after an investigation by the Federal Bureau of Investigation and the United States Department of Agriculture (MMWR, 2003a).

In 2007, a number of pet dogs and cats were reported to have become ill with symptoms consistent with kidney failure. Many of the pets were diagnosed with abnormal crystals in their urine samples and impaired kidney function. Many deaths of dogs and cats were alleged to have resulted. Epidemiological investigation strongly associated the ill pets with various pet foods that were processed by a Canadian pet food manufacturer, which used imported wheat gluten and rice middlings from China in the pet food formulations. Analysis of these raw products from China indicated that the wheat and rice products were contaminated with the synthetic polymer melamine to try to increase crude protein levels. The price of the wheat and rice was directly tied to crude protein concentrations and the contamination was a deliberate attempt to increase the value of the product.

An April 2007 hearing in the US Senate led to legislation requiring the Secretary of the Department of Health and Human Services to establish: "(1) processing and ingredient standards for feed, pet food, animal waste, and ingredient definitions; (2) update standards for pet food labelling that includes nutritional information and ingredient information; and (3) an early warning and surveillance system to identify contaminations of the pet food supply and outbreaks of illness from pet food". (US Senate, 2007) Although the melamine incident was a case of manipulating agricultural production for financial gain and not a deliberate form of agroterrorism, it illustrates how easy it was to contaminate foodstuffs for companion animals and humans.

Natural Disasters and Disease Outbreaks

The recent history of natural disasters and disease outbreaks that resulted in widespread agricultural losses in the United States is impressive. Hurricanes Floyd (1999) and Katrina (2005) left agricultural and environmental devastation in their wake. Total damage estimates as a result of hurricane Floyd were approximately $6 billion. The estimated livestock damage was estimated at over $13 million. Hurricane Floyd destroyed a total of 2,504,161 acres (1.0 million hectares) of crops, with damage estimates exceeding $543 million. Portions of ten states were declared major disaster areas, from Florida north to Connecticut (NWS, 1999 and NCFMP, 2007).

The effects of Katrina reached far beyond the immediate geographical area that felt her winds. The Mississippi River serves as the major shipping route for maize, soybean, rice and wheat produced in the nation's major grain-producing states from Louisiana to Minnesota, a distance of over 1200 miles. New Orleans, at the terminus of the Mississippi River, is a major port for US oil imports and for export of agricultural products and is the site of considerable oil processing. Hurricane damage brought a halt to the flow of agricultural trade through New Orleans, which resulted in a domino effect of negative consequences. Grain producers were not able to move their harvested grain to the port of New Orleans, requiring them to seek storage facilities at high storage fees. Commodity prices were affected due to excessive stockpiles and there was eventual loss of product that could not be stored properly (Schnepf and Chite, 2005). Hurricane damage to farm-related industries was estimated at more than $2 billion. The estimate included $1 billion in direct losses, as well as $500 million in higher fuel and energy prices (AFBF, 2005).

Officials suspected that the outbreak of Exotic Newcastle Disease that occurred in California in 2002 was introduced into the United States by birds smuggled into California for illegal cock fighting. Although that incident was relegated to 50 small backyard producer premises and only 5,700 birds were euthanised, it demonstrated how easily a potentially devastating disease agent could be introduced into the country (Nolen, 2002).

The rapidly expanding trade in exotic pets was associated with the first outbreak of human monkeypox seen in the United States. In June 2003, 71 people in several Midwestern states were afflicted with pox type symptoms after handling prairie dogs that they had purchased as pets. Monkeypox, native to parts of central and western Africa, is common in rodents. The prairie dogs had been housed with African Gambian rats at exotic pet supply warehouses, contracted the disease from the rats and transmitted monkeypox to humans (MMWR, 2003b).

In November 2003, The Center for Disease Control and Prevention (CDC) and the Food and Drug Administration (FDA) issued interim final rules to establish new restrictions on the import, capture, transport, sale, exchange, distribution and release of African rodents, prairie dogs and certain other animals to prevent the introduction and spread of monkeypox in the United States. The government response was swift but this incident yet again pointed out how easily exotic diseases could be introduced into the country (USFDA, 2004).

Agents and Toxins with Agroterrorism Potential

Agents and toxins that can cause disease in plants and animals are often easy to employ, spread rapidly through the environment, are generally safe for humans to handle and have a potential for great economic damage. Terrorists would not only want to create as much damage as possible, but would also want to take credit for the attack in the ensuing media coverage. Pathogenic organisms and toxins that affect animals might be more likely to be used as agents in an agroterrorism attack. There are a number of reasons for this:

- Animal pathogens can be easily imported, concealed and introduced into production animal herds.
- Many animal pathogens are zoonotic, i.e. they can cause disease in both humans and animals and can also be transmitted between humans and animals.
- Animal pathogens can multiply within the target species, create a carrier state and thereby serve as vectors of the disease, prolonging the outbreak.
- Many animal diseases are highly contagious and have the potential to cause a rapidly progressing, explosive epidemic.
- Due to very effective eradication programmes in the United States over the past century, production animals in the United States are immunologically naïve to many foreign animal diseases.
- Many animal pathogens can spread to wildlife, creating a reservoir that would be difficult, if not possible, to control.
- In addition, due to historically effective eradication programmes, veterinary professionals and producers, the first line of surveillance for foreign diseases, are inexperienced in their early recognition.

- In an outbreak requiring the widespread destruction of infected animals, the disposal of dead animals would become a challenge to public health agencies. Media coverage of the burning or burial of dead animals would be distressing to consumers and might cause them to question the government response.

In contrast, plant pathogens and toxins may be less likely to be employed as agents of agroterrorism because their effects might be more protracted and less dramatic, resulting in a diminished 'shock value' to consumers. Other reasons why plant agents would be less likely to be employed in an agroterrorism attack include:

- Plant pathogens are as easily imported into the country as animal pathogenic agents but difficult to disseminate over a large area due to weather, temperature, humidity, particulate binding, dilution and air currents.
- Many plant pathogens require months to show their effects. The time lag between introduction and noticeable disease would reduce the shock value of a terrorist act.
- Given the lack of immediate effect after introduction of a plant pathogen, it would be difficult for terrorists to 'take credit' for an event.
- Many plant pathogens mimic naturally occurring diseases, further reducing the shock value.
- Plant pathogens have very little, if any, direct effect on human health.
- There is not as much consumer distress over media coverage of destroyed crops as there is over destruction of animals.

Through its Select Agent Program, the United States Center for Disease Control and Prevention (CDC), regulates all laboratories and other entities that possess, use and transfer pathogenic agents and toxins that could be used in a terrorist attack. These select agents were designated as such because they have the potential to create a severe threat to human, animal and plant health, either through direct effects on humans, plants or animals or on their products, or because they are particularly virulent, easily introduced into populations and difficult to prevent, recognise and treat (Table 36.1) (CFR, 2005).

Current Susceptibility of US Agriculture to Terrorism

Consolidation and Vertical Integration

A trend in American agriculture over the past 50 years has been toward larger farms and less plant and animal diversification on each farm. Large farms have become so prevalent that according to the National Agricultural Statistics Service of the USDA, 6.7% of farms (143K/2.1 M) account for 75% of the value of agricultural production in the United States and average 2000 acres in size.

In the beef, pork and poultry industries, consolidation among meat packers to form mega corporations has followed and perhaps even directed the same trend. During the past 30 years, concentration within the meat packing industry has tripled and now just four beef packing companies control more than 83% of the industry. Similar trends have occurred in pig and poultry processing (Farm Bill, 2007 and USDA, 2002).

From a bio-surveillance standpoint, large consolidated farms can be either very good or very bad. If adequate surveillance and response systems are in place, large farms can recognise, control and mitigate breaches of bio-security rapidly and effectively. However, because of their sheer size, an unrecognised outbreak of animal or plant disease can affect many more acres of crops and more individual animals, resulting in significantly more financial loss than on small diversified farms.

Concurrent with the change from small diversified farms to larger, more specialised farms has been the vertical integration of agriculture. In a vertically integrated system, producers, shippers, processors and often retailers are all part of the same company. There are economic advantages to this business model as well as greater control of product quality and uniformity. Vertical integration provides an opportunity to develop and implement effective biosecurity programs such as Hazard Analysis of Critical Control Points (HACCP), since one management team has control of all aspects of production, processing and retailing to consumers, essentially the 'farm to fork' concept.

However, there are risks with vertical integration. From an agroterrorism perspective, the danger of vertical integration is that a breach in any part of the system could conceivably affect the entire system, resulting in

Table 36.1. Select Agent and Toxin List. Source: USCDCP, 2005.

USDA only agents and toxins	USDA/HHS overlap agents and toxins
Livestock • African horse sickness virus • African swine fever virus • Akabane virus • Avian influenza virus (highly pathogenic) • Blue tongue virus (exotic) • Bovine spongiform encephalopathy agent • Camel pox virus • Classical swine fever virus • Cowdria ruminantium (Heartwater) • Foot-and-mouth disease virus • Goat pox virus • Japanese encephalitis virus • Lumpy skin disease virus • Malignant catarrhal fever virus (exotic) • Menangle virus • Mycoplasma capricolum /M. F38/M. mycoides capri (contagious caprine o pleuropneumonia) • Mycoplasma mycoides mycoides (contagious bovine pleuropneumonia) • Newcastle disease virus (VVND) • Peste des petits ruminants virus • Rinderpest virus • Sheep pox virus • Swine vesicular disease virus • Vesicular stomatitis virus (exotic)	• Bacillus anthracis • Botulinum neurotoxins • Botulinum neurotoxin-producing species of Clostridium • Brucella abortus • Brucella melitensis • Brucella suis • Burkholderia mallei • Burkholderia pseudomallei • Clostridium perfringens epsilon toxin • Coccidioides immitis • Coxiella burnetii • Eastern equine encephalitis virus • Francisella tularensis • Hendra virus • Nipah virus • Rift Valley fever virus • Shigatoxin • Staphylococcal enterotoxins • T-2 toxin • Venezuelan equine encephalitis virus
Plants • Candidatus Liberobacter africanus • Candidatus Liberobacter asiaticus • Peronosclerospora philippinesis • Ralstonia solanacearum, race 3, biovar 2 • Sclerophthora rayssiae var. zeae • Synchytrium endobioticum • Xanthomonas oryzae pv. Oryzicola • Xylella fastidiosa (citrus variegated chlorosis strain)	

widespread exposure of consumers to tainted produce and massive financial loss to the company.

Other Potential Vulnerabilities of U.S. Agriculture

With the development of the international economy, humans, animals and consumer products can travel across the globe within 14 hours. In today's world, extremism and the frequency of international travel make the intentional introduction of foreign disease agents a possibility and unintentional introduction a probability.

Farms in the United States are geographically dispersed and are typically grouped in remote locations. Livestock, especially cattle, are often born and raised at one site and then transported to another site to be fattened to market weight. Ultimately they are transported once again to a processing facility at yet another location. Commingling of livestock, especially if surveillance is deficient, could allow easy and rapid transmission of disease among animals and across large regions of the country. A national identification system for all individual animals or shipments of livestock would greatly improve traceability. However, without a national capacity to arrive at an early and definitive diagnosis of foreign plant or animal diseases, spread of an introduced pathogen could occur unabated for too long to prevent widespread damage to the industry.

Although measures have been enacted to improve border security, experts warn that there is still much work to be done. In addition, there is little biosecurity at processing and distribution centres and even less on most farms. A recent study found that livestock producers fattening beef cattle did not have high implementation of biocontainment, biosecurity or security practices. The authors found that, in general, larger cattle feedyards had better biosecurity practices than smaller facilities (Brandt et al., 2008).

Since the United States has been relatively successful in eliminating many internationally important animal diseases such as foot and mouth disease and classical swine fever, there are few producers, extension agents and veterinary surgeons, conceivably the first line of surveillance, who are adept at recognising foreign animal and plant diseases. Furthermore, since eradication programmes have been so successful, there has been little need to fund federal indemnity programmes for livestock losses associated with disease surveillance and eradication programmes. Many livestock producers see no financial incentive in reporting odd disease symptoms for fear that an investigation by animal disease authorities will result in economic losses to them with little chance of regaining the value of animals or their products. In these circumstances, passive surveillance for animal diseases could be severely affected. This was apparent when the ban on processing downer cattle (cattle unable to rise) for slaughter took effect right after the first case of bovine spongioform encephalopathy (BSE) was discovered in the United States in 2003. (USDA, 2003) BSE surveillance was centred on testing at processing centres by teams from the Food Safety Inspection Service (FSIS) of the USDA. The highest BSE risk cattle, those that were non-ambulatory or showing neurological signs, were often lost to surveillance because, with the downer cow ban in place, they never made it to slaughterhouses where they could be tested.

The longer the time between recognition of symptoms and actual diagnosis, the greater the chance of an explosive outbreak occurring. Besides the lack of trained personnel on most farms, there are few on-site, rapid diagnostic tests for most agents liable to be used in an attack on the nation's food production systems. At a United States Senate Government Affairs Committee hearing 2003, a simulation of an intentional introduction of foot and mouth disease (FMD) into the country was described. Since the incubation period for FMD is three to five days and the disease is not usually diagnosed until day five, and due to the customary movement of cattle throughout the country, FMD would have already spread to 23 states by the time it was diagnosed. By day nine, it would have been in 29 states and would require an estimated 700,000 people to aid in response and recovery operations and to deal with the estimated 23 million head of cattle destroyed. Estimated losses ranged from \$30.4 billion to \$83.6 billion, not including loss of export markets. These estimates did not include the possibility of FMD spreading to deer and elk and becoming endemic in the wildlife population (US Senate, 2004).

A terrorist attack on the transportation infrastructure could also have profound effects on agriculture. Similar to the consequences seen with hurricane Katrina, any disruption of producers' ability to get their products to terminal markets would dramatically affect the agricultural economy, as well as food and feed supplies. Similarly, since agriculture is very labour-intensive despite advances in mechanisation, any intentionally introduced epidemic affecting the labour force would necessarily affect the harvesting, processing and marketing of food, feed and fibre.

Legislative and Presidential Responses

Legislative Responses

During recent years in the past, the United States Congress has held four hearings on agroterrorism, three in the Senate and one in the House of Representatives. Congressional hearings are formal events during which invited witnesses are asked to offer their expertise on the topic of interest. Congressional hearings serve to focus legislators' attention on particular issues and often result in the eventual introduction of legislation to address needs discovered during the hearings. The four hearings were entitled, 'Agroterrorism: The Threat to America's Breadbasket', 'Evaluating the Threat of Agroterrorism', 'Bio-security and Agroterrorism', and 'Biosecurity Coordination'.

Three major pieces of legislation were passed into law, two in 2002 and one in 2006, and focused on the nation's security with specific provisions to safeguard agriculture, the food supply and public health. The Bioterrorism Preparedness Act (PL 107-188) was passed in 2002 and expanded FDA authority over the registration of food manufacturing and required prior notice from importers within a specified period prior to product arrival at the US borders. It also created the Select Agent List and the policies tightening the control over possession, transport and uses of those agents. In addition, the Act expanded agricultural security at USDA facilities and defined criminal penalties for violation of the Select Agent rules and for terrorism activities against animal production and processing enterprises.

The massive Homeland Security Act (PL 107-296) created the Department of Homeland Security (DHS) by combining parts of many agencies throughout the federal government into one Cabinet level agency. Two major changes directly affected agriculture: 1) responsibility for agricultural border inspections was transferred from the USDA to the US Customs and Border Patrol (CBP) while retaining scientific expertise from the USDA; and 2) responsibility for Plum Island Animal Disease Center, the lead foreign disease unit in the United States, was transferred from the USDA to the DHS. The reason for consolidation of these activities within the DHS was to coordinate activities and intelligence to prepare the agricultural industry for disaster prevention, planning, response and recovery.

The third law, the Animal Enterprise Terrorism Act (PL 109-374) was enacted to expand criminal penalties for terrorism activities against animal production and processing enterprises.

Presidential Responses

Presidential directives have been issued throughout US history to establish national policies, goals and objectives, especially for administrative agencies within the Executive Branch of government. These directives can be challenged by Congress if they are thought to violate the statutory process or conflict with the Constitution. Presidential directives often carry the force of law but can be modified extensively by Congress through its legislative oversight. For example, Congress can authorise and appropriate funds for only some provisions of executive directives, thereby dramatically altering their function and effectiveness in achieving the original intent of the President (Relyea, 2007)

Three Presidential Directives concerning homeland security were issued from December 2003 to December 2004. The first was the Homeland Security Presidential Directive-7 (HSPD-7), which designated critical infrastructure that was vulnerable to terrorist attack and was essential to the operation of the economy and government.

Presidential Directive-9 (HSPD-9) was issued a month later and was entitled 'Defense of United States Agriculture and Food'. This directive established a national policy to protect against attacks on agriculture and food systems. The directive specifically instructed federal agencies to develop prevention and surveillance systems to monitor plant and animal health, food quality and public health through an integrated diagnostic system. It also recommended the establishment of a National Veterinary Stockpile (NVS), which would be able to deploy vaccines and therapeutic products within 24 hours of an attack, and a National Plant Disease Recovery System (NPDRS), which was charged with developing disease- and pest-resistant plant varieties within one growing season of an attack.

A previous Presidential Directive-5 (HSPD-5) called for the development of a National Response Plan to coordinate federal agencies as they plan prevention, response and recovery capabilities. Agricultural components of the plan provided nutrition assistance for disaster areas, control and eradication of animal and plant diseases and pests, assurance of food safety and protection of natural and cultural resources and historic properties.

The Executive Branch has also been active in promoting the development of public-private partnerships to protect critical infrastructure. Examples of public-private partnerships engaged in agricultural security activities include the:

- National Infrastructure Protection Plan, which shares best practices, identifies deficiencies in biosecurity, and improves communication between government agencies and private sector agriculture.
- Strategic Partnership Program Agroterrorism, which determines critical aspects of agriculture that may

Table 36.2. Appropriations for funding of activities direct against agroterrorism. Source: Monke, 2007.

YEAR (millions)	2002	2003	2004	2005	2006	2007	2008 (requested)
USDA	552	416	412	596	598	523	719
DHS	NA	69	227	211	264	295	300
TOTAL	552	485	639	807	862	818	1,019

be vulnerable to attack, enhances intelligence and surveillance, and develops mitigation strategies in case of an attack.
- Information Sharing and Analysis Center, which assesses information shared with the law enforcement and intelligence communities

Private, university and governmental laboratories and research centres have also been asked to coordinate activities so that a seamless laboratory system would have the surge capacity needed to respond in the case of an attack. in addition, public and private research centres have been asked to coordinate research activities to provide the scientific basis for response and recovery phases.

Funding of Government Anti-agroterrorism Activities

Legislative and executive branch responses to the threat of agroterrorism have little impact unless they are accompanied by appropriate levels of funding. The US Congress has oversight of the national budget and has, for the most part, complied with the President's funding level requests for agroterrorism activities.

The bulk of federal funding to combat the potential of agroterrorism proceeds to the Departments of Agriculture and Homeland Security. The agricultural biosecurity functions of these two agencies are funded primarily through direct appropriations from Congress and, to a lesser degree, through user fees generally charged to food importers. The leading USDA recipients are the Agriculture Research Service (ARS), the Animal and Plant Health Inspection Service (APHIS), the Cooperative State Research, Education and Extension Service (CSREES), the Food Safety Inspection Service (FSIS), the Economic Research Service (ERS) and various administration and executive operations.

Funds appropriated to the Department of Homeland Security are used to support Customs and Border Protection (CBP) and to support research and training in science and technology.

Federal funding of activities directed toward agricultural biosecurity has increased by $262 million from the 2002 budget though to the 2007 budget. The requested budget for 2008 would increase funding by another $201 million over the 2007 budget, which would be almost a 100% increase in funding from the 2002 baseline level (Table 36.2).

Federal funding for agroterrorism activities described by function is depicted in Figure 36.1. The bulk of funding goes to the US Customs and Border Patrol. The second largest amount of funding is spread across the USDA and DHS and directed to address catastrophic threats. Ninety three percent (93%) of funding for agroterrorism activities is directed to border security, identifying and researching catastrophic threats and preserving critical infrastructure.

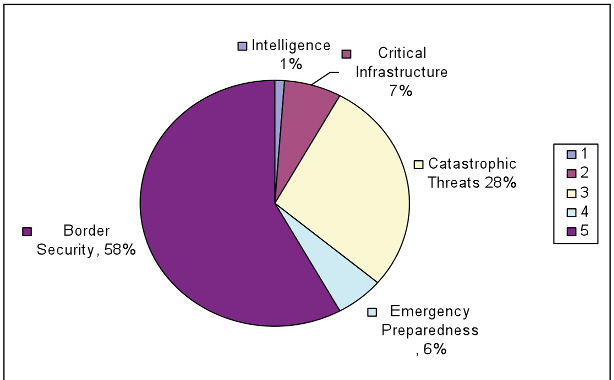

Figure 36.1. Homeland security funding for agriculture by function. Source: Monke, 2007.

Conclusions

Agriculture is a significant component of the US economy, providing safe and inexpensive food for people and feed for animals at an affordable cost and helping to balance trade with international markets. It is fundamental to the health of humans, animals and the ecosystem.

Previous natural disasters and incidents of deliberate contamination of food, feed and water have shown that agricultural systems in the US are susceptible to intentional and malicious attacks. The ineffective state preparations and federal responses to hurricane Katrina in 2005 highlighted the necessity for a well-funded, directed and coordinated response to help communities in need.

The US government has taken numerous measures to counteract the threat of agroterrorism based on the principles of primary, secondary and tertiary prevention. Primary prevention focuses on eliminating or reducing individual and community exposures through enhanced surveillance and communication at borders, at production and processing facilities, and at importing entities, whether the importers are countries or international corporations. The bulk of the federal budget allocated to agricultural biosurveillance is directed at border security and keeping foreign pests and pathogens out of the country.

Secondary prevention centres on recognising exposures and preventing disease through better training of individuals in the diagnosis of foreign diseases, better diagnostic tests and coordinated and timely laboratory response capabilities. These areas have also received considerable federal financial support over the past five years. A recent example of renewed interest in developing a cadre of professionals adept at foreign disease diagnosis is the mandatory training in exotic and emerging animal diseases taken by students in the professional veterinary medicine programmes at 23 colleges in the United States. However, such efforts are offset by the unfulfilled need for veterinary surgeons in production medicine and public health practice. Legislation to encourage more veterinary medical students to enter food animal and public health careers has also been introduced in Congress.

Tertiary prevention concentrates on mitigating the effects of disease outbreaks through rapid response and mobilisation of necessary funds and personnel. Over the past seven years throughout the country, there have been, and continue to be, many simulations and tabletop exercises that link local, state and federal agencies in a coordinated response to natural disasters and intentional exposures. All three levels of government have been energised to develop coordinated and workable plans to respond efficiently to threats to public health.

Public health and agricultural planners recognise that the damage associated with an outbreak is proportional to the time elapsed between the index case and diagnosis. They also recognise that as efforts progress through primary to tertiary prevention, attendant costs rise dramatically. Planning must be all-inclusive and bring to the discussion the intelligence and science communities, the public and private sectors and risk communicators who can inform consumers about the nature of threats.

Bioterrorism and Intentional Contamination of Drinking Water

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Ari Hörman
Finnish Defence Forces, Helsinki, Finland
Mati Roasto
Estonian University of Life Sciences, Tartu, Estonia
Marja-Liisa Hänninen
University of Helsinki, Helsinki, Finland

Biohazardous Agents

Water sources, drinking water supply systems and treated drinking water can become contaminated with naturally occurring microbes or toxins, but may also be targets of bioterrorism, sabotage and intentional contamination (Burrows and Renner, 1999; Khan et al., 2001; McAvin, 2004). The United Nations Biological Weapons Convention of 1972 prohibits member states from developing, producing and using bioweapons, but monitoring compliance with this convention has proven difficult. Even the definition of a bioweapon or bioagent (B-agent) has proven to be elusive, since the Convention prohibits only the use of bioweapons and not the B-agent itself. The intentional destruction of drinking water resources leading to water-borne diseases without a committed microbial contamination with a certain B-agent can be viewed as a type of biological warfare (White, 2002). Moreover, the bioweapons are most probably used by individual terrorist organisations and disturbed individuals rather than governments or nations.

B-agents which may be utilised in intentional contamination of drinking water include naturally occurring human enteropathogenic microbes, eradicated or uncommon pathogens, genetically modified organisms or microbial toxins (McAvin, 2004). In theory, any microbe or microbial toxin possessing the potency to cause illness or disorder in man can be used as a B-agent against a target population through the drinking water supply.

The most frightening B-agents include microbes and microbial toxins, which have low infective, incapacitating or lethal dose, high contagiousness, no acquired immunity in the population and no medication or preventive means available. To be able to infect or intoxicate through drinking water, the organism or toxin should survive in the aquatic environment and tolerate other unfavourable environmental conditions. Intentional contamination of drinking water with microbes or toxins that are colourless, odourless and tasteless presents a serious threat and this threat cannot be assessed by sensory testing of water. Botulinum neurotoxins (BoNTs) produced by *Clostridium botulinum* present this kind of severe threat and are the most potent biotoxins known (Gill, 1982a; Schechter and Arnon, 2000).

Detection of Bioterrorism

Distinguishing between a naturally occurring disease or outbreak and intentionally spread disease may be extremely difficult. The uncommon symptoms, high infectivity, severity or other abnormal factors may direct suspicions towards B-terrorism. Surveillance for infectious diseases and early notice of single cases and outbreaks of emerging diseases are essential for the prevention of further infection (Hugh-Jones, 2003). Unfortunately, surveillance systems have been found to be insufficient to detect possible intentional release of B-agents (Ashford et al., 2003)

Rapid and sensitive tests are needed for detection of B-agents and biotoxins, as well as for rapid screening for susceptive samples. Some devices are already developed for use and an expanding market is predicted for detection industry (Alocilja and Radke, 2003). Real-time PCR methods have provided some promising results for detection of *Fransiella tularensins* (McAvin et al., 2004), *E. coli* and *Bacillus antracis* (Higgins et al., 2003). Some multiplex diagnostic platforms have also been described (Cirino et al., 2004). Fundamental differences in detection principles between microbes and inorganic chemicals probably mean that similar easy real-time detection of microbes, similar to that for chemicals, will be not be available in the near future (Green et al., 2003).

Protection Against Bioterrorism

All actions taken to ensure the drinking water safety and security are also actions against bioterrorism and vice versa. There are only limited means to specially protect against B-agents used to intentionally contaminate the drinking water apart from controlling all critical control points, guaranteeing the treatment efficiency and securing the treated drinking water from manipulation. The most effective general protection against B-agents is to maintain high general and drinking water hygiene and use adequate treatment and common sense. Probiotic microbes, e.g. *Lactobacillus* spp. are well known to have beneficial effects on stabilising intestinal disturbances (Isolauri, 2001; Ried, 2004) but may provide only some protection for less severe B-agents.

Vaccination and immunisation can be implemented in the event of a special threat, e.g. against *B. antracis,* smallpox or BoNT (Arnon et al., 2001; Grabenstein, 2003). If exposure or suspected infection has occurred or clinical disease has developed, specific medication with appropriate antibiotics or antidotes can be initiated after consulting the medical experts.

Protecting the population against bioterrorism or intentional contamination of drinking water is a multidisciplinary challenge in which close collaboration and cooperation between veterinary, public health and medical professionals together with experts on security, water engineering and communication are essential (Mossel, 1990; Hartung, 1992; de Balogh et al., 2002; Rose, 2002).

Part I

Impact of Climate Change on the Health of Wildlife, Domestic Animals and Ecosystems

Authors: Ann Albihn, Hans Gustafsson and Marilyn O'Hara Ruiz

Coordinating Author: Ann Albihn

Preparing for Climate Change

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Ann Albihn
National Veterinary Institute, Uppsala, Sweden
Hans Gustafsson
Swedish University of Agricultural Sciences, Uppsala, Sweden
Marilyn O'Hara Ruiz
University of Illinois at Urbana-Champaign, Urbana, IL, USA

Climate change (CC) in terms of global warming and sea level rise is an ongoing process and a topic of immediate interest. Extreme weather events such as heat waves, storms, heavy precipitation, floods and droughts, are expected to become more frequent as CC progresses; whereas snow cover and the coverage of land and sea ice are predicted to contract. According to a report from the UN Intergovernmental Panel (IPCC, 2007) the main underlying reason for CC is anthropogenic greenhouse gas emissions causing a global increase in average temperatures. Even though this report is based on a huge amount of scientific evidence, it has been criticised for underestimating certain effects of CC, such as the extent of loss of Arctic and Antarctic ice-cover. A recent Swedish review of scientific literature published 2007-2009 concluded that continued global warming is more severe than previously thought and that future effects can be even more far-reaching than previously estimated (Rummukainen and Källén, 2009).

The mathematical modelling of predicted CC is becoming increasingly advanced and the possibility to assess changes in the future climate is improving continuously. However, climate models, just as any other kind of mathematical models, have their limitations considering the possible amount of parameters included and their interactions and flexibility. Therefore any predictions of CC should be considered indicative rather than accurate.

Depending on the viewpoint, CC may cause positive effects for certain regions and species, for example for crop production in some regions. However, in general, as a consequence of CC, many severe problems for society and agricultural production are expected to surface or, in some cases, are already evident (SOU, 2007:60). Increased shortage of water is anticipated to become an increasing problem for society. As CC influences health in many ways, the well-being of humans and domestic animals as well as of wildlife and ecosystems is expected to be affected. In the case of extreme weather events the health effects may be drastic and acute. Other effects from extreme weather events are that lack of feed and water may follow or there may be outbreaks of epizootic disease. Heat stress may cause suppressed production and decreased reproductive performance in farm animals (Sartori et al., 2002; Huynh et al., 2005). The impact of vector-borne diseases (VBD) that are transmitted between individuals via insect or other arthropod vectors on human and animal health is expected to increase. Several of these VDB are zoonoses and can therefore be transmitted between humans and animals (for example West

Nile fever and Lyme borreliosis); others involve a severe impact ‘only’ for animal husbandry (for example blue tongue disease).

Mitigation of the above-mentioned effects is central in today’s discussion of CC, but adaptations and animal health effects would benefit from greater attention. For example heat stress among domestic animals may be less severe in temperate climate zones if animal breeds genetically adapted to a warmer climate are used (Setchell, 2006) and if farmhouses are constructed in a way that protects animals from high temperatures and humidity. However, adaptations to protect for infectious diseases, which often appear as a surprise in new regions, may be rather difficult to achieve, and prediction and response to future problems is demanding. To understand and facilitate adaptation to CC – as far as possible – increased knowledge is needed about changes in ecosystems, including interactions between vector animals, wildlife, domestic animals and humans. For certain diseases vaccination of a population of animals or humans may be an alternative when the risk of introduction of a new disease is high or when the disease has already been introduced. However, effective vaccines are not always available or affordable. Vaccination of livestock in northern Europe against blue tongue disease has been extensive since the disease was introduced in 2006 (Purse et al., 2008).

Potential for Animal Production will Change

Today’s agriculture is under high pressure to increase its productivity, which would compensate for food scarcities and an increasing human population (FAO, 2008). At the same time there is a demand for the negative impacts of agricultural production on the environment to decrease (Steinfeldt et al., 2006). This is a global dilemma of the highest priority which has yet to be solved. Healthy animals are a key factor in this regard, since they produce more food while consuming less feed (Wood and Lysons, 1988). In addition, healthy animals cause less greenhouse gas emissions in proportion to the amount of food produced (Flint and Woolliams, 2008). Even so, if the main purpose of agriculture is food production, it should not be forgotten that animal production includes promoting sustainable rural development and preservation of biodiversity and ecosystem health.

Climatic changes are expected to alter the potential for farming and animal production, but the consequences of these will depend on prevailing circumstances in a region or production system (de La Rocque et al., 2008). In certain areas, such as the temperate and cold northern latitudes, the consequences of CC may, to some extent, be positive. A prolonged grazing season for animals can lower the incidence of infectious respiratory and gastrointestinal diseases which are associated with the indoor period. On the other hand, the exposure to VBD and para-

Figure 38.1. Clean water in sufficient amount will be one of the challenges with climate change. Jumkilsån. Photo: Bengt Ekberg, SVA.

sites will be prolonged with extended outdoor periods. In addition, some parasitic diseases are suggested to become an increasing problem for animal production (Van Dijk et al., 2008; Mas-Coma et al., 2008). As the winters get warmer the ground may not freeze (IPCC, 2007) and may become wet and muddy. This can cause hygiene and health problems and infections in the skin and hooves of cattle and sheep. In addition, outdoor animals are more exposed to extreme weather events such as storms, fires, drought and flooding. In addition to the problems with VBD, dense populations of certain insects can become a serious nuisance for grazing animals.

In areas already classified as marginal today according to the possibilities for livelihood and food production, even a small change concerning temperature or precipitation events may be disastrous (FAO, 2008). Water supply concerning both quality and quantity may become a problem. Animal husbandry consumes large quantities of water for crop production, cleaning, drinking, etc. A high-yielding dairy cow consumes more than 100 litres of water per day, and thus a decrease in water quality may negatively affect animal health. A shortage of water may be one of the most severe and widespread health effects of CC. Today's dry areas are in general predicted to get dryer as CC progresses and the patterns of precipitation to become more irregular. An increased drought frequency can have catastrophic effects for agriculture. In production systems based on extensive grazing and relying on natural water sources, the distance for the animals to reach drinking water may become too long when water sources dry out. Heavy rainfall and flooding, on the other hand, may facilitate rapid transportation of diseases-causing pathogens into water supplies, which can potentially harm populations of both wild and domestic animals and humans. When animals drink water of low hygienic quality, they may acquire infections such as VTEC, salmonella and water-borne parasites.

A warmer and more humid climate may also create new demands for housing animals, especially under the intensive indoor production conditions used for pigs, chickens and eggs. Pigs and chickens cannot sweat and are therefore especially sensitive to heat if not provided with possibilities to seek shade or wallow in mud or water. The high-yielding dairy cow is also sensitive to heat stress (Sartori et al., 2002). In temperate and cold northern latitudes today, farm buildings are to a large extent constructed to protect against cold weather and may not be adjusted to protect against heat. As a consequence of global warming, the construction of suitable buildings and the provision of shade from direct sunlight on pastures may become necessary in new regions when attempting to maintain high production.


Figure 38.2. Climate change can lead to a prolonged grazing period, which both can enhance and decrease the infection risk for the grazing animals. Photo: Uffe Andersson, LRF.

In the temperate and cold northern latitudes a prolonged vegetation period may be beneficial in many ways. As a consequence of a prolonged grazing period, the need for winter feed decreases. The possibility to grow new feed crops may also arise, e.g. maize has already been introduced and established as a common feed crop in southern parts of Scandinavia. It may be possible to increase the crop yield and take an additional harvest per season. However, there may be a more insecure supply of feed. Droughts or heavy rainfall may be detrimental for the crop in some years or in some regions. There may also be a shortage of feed for grazing animals during the more frequent periods of drought and therefore a need for extra feed supply. A warmer and more humid climate makes storing feedstuffs more risky, as microbial growth may be enhanced and hygienic quality lowered. The presence of salmonella bacteria or mycotoxins in feed can be more common. Mycotoxin production may also occur even in the field and new species e.g. *Fusarium* spp. may emerge in new areas.

Impact on Health – Infectious Diseases

For both domestic animals and wildlife, new infectious diseases are a serious threat. Introduction of new infective agents may entail disastrous effects for a large proportion of individuals in an immunologically naive population and, in extreme cases, lead to extinction of entire populations. Globalisation with increased movement of animals, people and goods facilitates dispersal of disease agents far from their endemic regions (Moore et al., 1988; Harrington et al., 2005; Mintiens et al., 2008). As infectious diseases today are spreading geographically much faster than at any previous time, an epidemic or epizootic disease outbreak in any part of the world is not far away from becoming a threat somewhere else. Consequently, new diseases can arrive from 'anywhere', while CC may facilitate establishment of novel imported infectious diseases in regions that were previously unable to support endemic transmissions (Dufour et al., 2008). The mechanisms by which CC affects disease transmission have often been oversimplified, as many other factors also influence the environment and the behaviour of populations and individuals (McMichael et al., 2006; Randolph, 2008). For example, human activities such as change of land use and fragmentation of habitats can enhance the dispersal of disease agents. Extreme weather events such as flooding, storms, droughts, etc. are associated with an increased disease risk. Such events may influence infectious diseases more profoundly and acutely than the ongoing CC (Kinde et al., 1996; Kriz, 1998; Hubalek et al., 2004).

In terms of CC, VBD are a special concern (Gubler, 1998; Dufour et al., 2008; de La Rocque et al., 2008). The transmission of VBD is closely linked to nature and ecosystem structure and therefore to a large extent dependent on processes in a specific ecosystem (Hales et al., 2006). Most VBD that are expected to emerge because of CC are zoonotic diseases. Compared with 'human-only' diseases, zoonoses are in general more difficult to control with vaccination, education of populations and medical checks of travellers, etc. In an attempt to identify animal infectious diseases of importance whose introduction or distribution in France could be affected by CC, five out of six prioritised diseases were VBD and five of these six were also zoonoses (Dufour et al., 2008).

Although the introduction of exotic diseases to new regions is a concern, geographical and seasonal patterns of endemic infectious diseases may also be altered due to CC. It is possible that well-known, persistent diseases will change character in respect of severity, epidemiology, incidence, etc. The seasonality of outbreaks of a certain disease has long been commonly recognised. The same has been observed for periodic epidemics and disease outbreaks of less frequent or more irregular intervals. Knowledge of the mechanisms triggering epidemics with interannual cycles is in most cases sparse. Most epidemiological data concerning infectious diseases are restricted to certain periods of time or to specific countries or regions. The identification and relative importance of climatic factors for disease dynamics on a longer timescale is therefore still a controversial topic (McMichael et al., 2006). Other non-climatic aspects such as environmental disturbances and pollution, land use changes, habitat fragmentation, effects of altered behaviour, etc. also affect the incidence of diseases. These factors may have either cumulative or opposing effects on disease occurrence. For example, effects of CC in promoting the dispersal of 'human-only' VBD such as malaria and dengue fever from more tropical ranges to temperate areas have been observed (Epstein, 1995; Patz et al., 1996). Another interesting example of climate effects on the incidence of human infections is the El Niño Seasonal Oscillation effect (ENSO). This semi-regular climate cycle, although not perfect, can be used as an analogue for the effects of global CC (Cazalles and Hales, 2006). The ENSO event has been associated with an increase in diarrhoea in Peru, cholera epidemics in Peru and Bangladesh and dengue fever and malaria epidemics in several tropical and subtropical countries (Cazalles and Hales, 2006). ENSO-events are predicted to become a more frequent and severe phenomenon as global warming progresses (IPCC, 2007), which would further facilitate local epidemics of certain diseases.

Another aspect of CC is that an effect on the pathogen microorganism itself may be seen. This has been speculated to cause an increase in the virulence of pathogens in some cases (Marcogliese, 2008). To fit a dynamic environment such as that induced by CC, a generalist strategy is favourable. A generalist organism displays pioneer behaviour and is ready to encroach or occupy any novel ecological vacuum that may present itself. For example, the ability to infect multiple host species makes it possible for a pathogen to overcome temporary shortfalls in host availability. A pathogen with a flexible host-occupancy pattern is also an effective transmitter of infection to new species. Vector-borne viruses are often RNA viruses, which are known for their variability and limited specificity of hosts (de La Rocque et al., 2008).

Opportunistic Pathogens

Commensals or opportunistic microorganisms cause no harm when present in healthy animals. Stress caused by e.g. increased temperature, increased population density, high density of biting insects or lack of food may induce suppression of the immune response and lead to increased susceptibility of organisms to opportunistic pathogens. An example of this is an outbreak of fatal pneumonia caused by an opportunistic bacterium (*Pasteurellaceae* or *Mannheimia* spp.), which occurred in a musk ox population of Dovrefjell in Norway. A large proportion of the animals died during a period of extraordinarily warm and humid weather during early autumn 2006 (Ytrehus et al., 2008). Musk ox, like other Arctic species are adapted to extreme cold and therefore regarded as vulnerable to the impacts of CC (ACIA, 2005). Dovrefjell is a rather southerly habitat for this cold-adapted species and during early autumn the animals have a well-developed winter coat and a thick layer of subcutaneous fat, which makes them especially vulnerable to heat stress (Ytrehus et al., 2008).

Parasitic Infections

For domestic animals a prolonged grazing period has several advantages, but a negative health effect may follow from prolonged exposure to parasite infections. Warmer and more humid climates, especially in temperate and colder northern latitudes and in areas of high altitude, may shorten generation time and increase the survival and population density of parasites (Feachem et al., 1983; Mas-Coma et al., 2008; van Dijk et al., 2008). On the other hand, e.g. a sparse protective layer of snow during the winter or less protective vegetation during drought may reduce the survival of free-living stages of the parasite or of intermediate hosts. Intermediate hosts are common in the development cycle of many parasitic organisms and play an important role in disease transmission dynamics (Feachem et al., 1983). Therefore an expansion of suitable habitats, e.g. damp areas, for invertebrate hosts may favour the existence of parasitic organisms.

Zoonotic parasites such as *Cryptosporidia, Giardia* and *Toxoplasma* can be water-borne and are suggested to cause increasing problems in the developed world, in part due to CC (Gajadhar and Allen, 2004; Mas-Coma et al., 2008). The ability of these parasites to survive for long periods of time in the environment and resist many natural and artificial conditions makes them most difficult to control (Feachem et al., 1983).

In a British study, 430 000 samples of faeces from sheep with gastroenteritis symptoms were analysed for the presence of parasites during the period 1975-2006 (van Dijk et al., 2008). A highly significant increase in the rate of parasitic gastroenteritis was observed during this period and CC was suggested to be the most likely explanation for this increase. Possible sources of bias such as reporting bias, changes in husbandry patterns and antihelminthic resistance were also evaluated (van Dijk et al., 2008).

Under optimal conditions for an individual, many parasitic infections may not provoke any negative effect on the physiology of the host. However, under certain environmental conditions involving stress, the effects may become negative. Intestinal parasites may cause protein losses in the intestine, resulting in decreased digestive ability and increased nitrogen in the manure. Parasitic diseases, especially helminthiases, have a great impact on animal health. Under certain conditions they can also influence wild animal populations and in that way affect whole ecosystems (Mas-Coma et al., 2008). CC may cause amplification of the parasite population and have profound effects on the host-parasite assemblages. This in turn can have large impact on the health and survival of wild animal populations. Kutz et al. (2001) gives an example of this effect on a Canadian musk ox population. This population was decimated to about half during a six-year period due to an unusual genus of a protostrongylid lung nematode. Even though this parasite did not kill the animals, it lowered their lung capacity and made them more vulnerable to predators.

Impact on Health – Heat Stress

During the process of domestication many animals have become adapted to our temperate climate, with temperatures seldom above 25°C. As global warming progresses, periods with temperatures above 30°C are expected to become more frequent. Dairy cows in particular are expected to be affected by these increased temperatures since they are to a large extent out on pasture in the summer, in many cases without shade. Moreover, the high-yielding dairy cow is more sensitive to heat stress than other farm animals due to her high feed intake and metabolic rate as a result of a progressively increased milk yield (Sartori et al., 2002).

One of the highest physiological priorities for warm-blooded animals is the maintenance of body temperature. The ideal ambient temperature for dairy cows is between 5 and 25°C, which makes it possible for the cow to maintain a body temperature of approx. 38.5°C. At temperatures above 25°C cows have to use energy to cool themselves through heat loss via surface skin and the respiratory tract. As ambient temperature increases, it becomes more difficult for cows to adequately cool themselves. Temperature-humidity index (THI) is most commonly used to measure the combined effect of air temperature and moisture in the air (Thom, 1959). It is generally accepted that when THI exceeds 74, heat stress effects on reproduction begin to be manifested (USDC-ESSA, 1970).

Domestic pigs are also sensitive to high surrounding temperatures as they are not able to sweat and therefore cannot utilise evaporative cooling of their skin. On the other hand, the lack of substantial hair coat in pigs facilitates some heat loss from the skin surface. Under hot conditions, pigs usually attempt to increase heat transmission by increasing contact between their body and a cooler surface (floor) or wallowing in mud or water. The first physiological indicator that pigs are reacting to high ambient temperatures is an increase in respiration rate, which occurs at temperatures above approximately + 22°C for pigs of around 60 kg (Huynh et al., 2005). Extremely warm conditions can result in the death of pigs if the possibility for supplemental cooling is not provided. However, reduced growth resulting from decreased feed intake and decreased reproduction are more common effects of increased temperatures (Quiniou et al., 2000; Suriyasomboon et al., 2006).

Figure 38.3. A pig covered in mud, an important way for the non-sweating pig to cool down through evaporation. Photo: Bengt Ekberg, SVA.

Figure 38.4. Pig huts may protect pigs from trying weather conditions. Photo: Bengt Ekberg, SVA.

Heat Stress and Reduced Reproductive Performance

Heat stress has long been recognised as reducing reproductive efficiency in domestic animals in tropical areas. In particular, the impacts of heat stress on reproduction in dairy cows have been well documented (Jordan, 2003). The question is whether and to what extent the expected CC in temperate climate zones such as Northern Europe will affect the reproductive efficiency of our farm animals.

There are marked depressions in pregnancy rates per insemination during the warm months. For example, an investigation in Florida showed that the pregnancy rate in cows in July was around 10%, compared with 35% in February (Cavestany et al., 1985). In a recent study of insemination outcomes of 16,878 services performed during 21 months in Australia, pregnancy rates were related to when THI exceeded 72 during the first 24 hours post-service. Conception rates were reduced when lactating cows were exposed to an elevated heat load (> 24 h with THI > 72) from the day of service until six days after. Weeks preceding service with an elevated heat load were also associated with reduced pregnancy outcome (Morton et al., 2007).

Declining Oestrous Symptoms

Heat stress has been shown to reduce reproductive efficiency in e.g. dairy cattle and domestic pigs by reducing oestrous expression and the duration of oestrous behaviour. A reduction in oestrous detection leads to a declining ability to perform insemination. Reduced oestrous behaviour is in part related to reduced general activity in heat-stressed cows. Cows become sluggish in a hot climate and are not willing to move or show mating attempts (Pennington et al., 1985). It has also been reported that high temperatures cause reduced peripheral concentrations of oestradiol 17-β and a decreased gonadotrophin response to injection of gonadotrophin-releasing hormones (GnRH), pointing at possible endocrine effects of heat stress on oestrous expression (Gilad et al., 1993).

Effects on Fertilisation and During Gestation

Early studies showed that uterine temperature and average ambient temperature on the day of insemination were inversely related to pregnancy rates in dairy cows (Gwasdauskas et al., 1973). Several recent studies indicate that disruption of the reproductive process can occur quite early – during oocyte growth or maturation, since oocytes are temperature-sensitive. According to Al-Katanani et al. (2002), oocytes collected from cows during the warm season were less likely to give rise to an embryo capable of development to the blastocyst stage than embryos collected during the cold season. Similar results were reported from Israel, where oocyte quality gradually improved as the season progressed from early to late autumn. However some studies have also shown a delayed effect of heat stress on oocyte quality, leading to a prolongation in decreased fertility even when the cows are no longer under heat stress (Roth et al., 2001). Furthermore, heat stress has been shown to change follicular dynamics in the ovary. The duration of dominance in the ovary of the preovulatory follicle has been shown to increase in hot weather and is negatively correlated to fertility (Wolfenson et al., 1995).

Embryos seem to become more resistant to heat stress as pregnancy proceeds. However heat stress has been reported to reduce uterine blood flow, which could potentially reduce the delivery of nutrients and hormones to the uterus and placenta. In a study by Collier et al (1982), the secretion of the placental hormone oestrone sulphate, placental size and calf birth weight were shown to be reduced in heat-stressed cows. In a another more recent study in north-east Spain by Garcia-Isperto et al. (2006), which involved 1391 pregnancies in Holstein cows with a mean annual production of 10 890 kg per cow, a clear relationship was found between maximum THI on days 21-30 of gestation and subsequent foetal loss.

Early research in pigs showed that heat stress had a negative effect on embryo survival during the first 30 days of pregnancy, whereas during mid-pregnancy it did not influence the number of pigs born per sow. However, when pregnant gilts were stressed during the last two weeks of pregnancy, an increase in the number of stillborn pigs per sow was observed. Further research has shown that the first five days of pregnancy are especially critical for sows to become heat-stressed (Suriyasomboon et al., 2006). Heat stress during lactation causes reduced feed intake, leading to reduced milk production and weight loss. Pig producers, especially in tropical areas, utilise several methods to minimise heat stress e.g. water sprinkling, ventilation, increased floor space and shaded areas.

Effects on Male Fertility

In male mammals high temperatures impair spermatogenesis. The effects of heat on sperm production include decreased sperm numbers, decreased sperm motility and increased numbers of abnormal sperm. Primary (immature) spermatocytes are especially susceptible to elevated temperature. Consequently, males of many species have evolved anatomical structures for local thermoregulation that involves placement of the testes at the periphery of the body. As a result, testicular temperature is approx. 2-4°C below body temperature. A negative effect of temperature on male fertility has been known for centuries. Furthermore cryptorchids, animals whose testes do not descend into the scrotum, are infertile. However in some animals e.g. elephants and dolphins, the testes remain inside the abdomen. Cooling of testes in these species is often achieved by alternative physiological solutions: for example in dolphins the testes appear to be cooled by venous blood from the tail running into the abdominal cavity next to the arterial supply for the testes. Spermatogenesis in many animals is disrupted when either the thermoregulatory system of the testes is defective or when body temperature becomes elevated in general because of fever or heat stress (Cameron and Blackshaw, 1980). A well-known case is the summer sterility affecting rams in Central Australia, which may also occur in natural populations of wild animals. There is a delay of about two weeks between heat stress and the first alterations visible in the semen. It is also notable that the effects of heat stress on sperm output in the bull persist for 7-8 weeks after the end of heat stress. Some recent studies suggest that the effects of heat on the testes are never fully reversible. However, there is a considerable variation between individuals and breeds and their response to heat exposure. For example, *Bos indicus* bulls are less sensitive to the effects of high temperature than *Bos taurus* bulls (Setchell, 2006).

Impact on Ecosystem Health

Changes in ecosystems as a consequence of CC will have a profound impact on ecosystem health. Today an obvious effect of CC has been observed on the distribution of vertebrates, invertebrates and plant species, on timing of seasonal activities of species and on physiological responses in both terrestrial and aquatic organisms (Marcogliese, 2008). Ecosystems with a low biodiversity, e.g. in Arctic and sub-Arctic climate zones, are more sensitive to changes (ACIA, 2005) that the more diverse systems (Parmesan, 2006). Continuous and systematic assessment of wildlife health can provide a means to estimate ecosystem health, e.g. infectious diseases are one manifestation of deteriorating ecosystems. New diseases may cause epizootics, especially if the native biota is immunologically naive to the new agent.

Predicting how ecosystems are likely to respond to further CC is complicated. Multiple factors and changes may be of importance for ecosystem health and in some cases other factors may play an even greater role than CC. Human modifications of landscapes such as deforestation, urbanisation, agriculture, coastal zone management and also consequences of pollution are already threatening many different kinds of environments. Complex ecological interactions result from several factors working in unison or in tandem. Sudden, fundamental changes (threshold changes) in ecosystems can already be observed today and further anthropogenic changes can magnify the health effects of CC and extreme weather events. Taken together, the present and coming changes in ecosystems will have profound implications for epidemiology and animal health.

Species Distribution and Importance of Biodiversity

In general, biodiversity is highest in the tropics and lowest in the Arctic and sub-Arctic climate zones. Loss of biodiversity increases the vulnerability of the ecosystem and may tip the ecological balance. For example, declines in predator populations can disrupt the natural biological control system of prey species, which can, as a consequence, become pests and carriers of pathogens (Epstein, 2002). At present, a marked change in the existence and geographical distribution of many species can be observed (Parmesan, 2006). It has been predicted that as many as 30% of animal species may disappear (www.iucnredlist.org) because of recent changes. In retrospect, during past periods of CC the survival of species has been relying on different strategies. A shift in distribution by migration to areas of more suitable climate has been a common historical way of survival. Temporary establishment in refugee areas has often been practised, together with downsizing in population numbers and in distribution area. However, a higher density of an animal population, as may follow migration, can create an opportunity for density-dependent pathogens to cause disease outbreaks. Today, many examples of species shifting upward in altitude and northward in latitude can be found (Lovejoy, 2008). Species in the Arctic/Antarctic and high latitude locations will be particularly vulnerable, as there will be no possibility for a climatic shift in terms of altitude or latitude (ACIA, 2005). In addition, the expected average temperature increase will be highest close to the poles (IPCC, 2007). Another survival strategy during periods of CC is adaptation by genetic selection, but examples of this strategy are rare and often debated (Parmesan, 2006).

In general, VBD today are more prevalent in the tropics, since the warm and humid climate there provides ideal conditions for vectors. However, the large diversity and abundance of different infections in the tropics is most likely buffered by the large biodiversity, causing a 'dilution effect'. Through this, the presence of incompetent disease reservoirs decreases the impact of highly competent reservoirs and reduces the disease risk (Marcogliese, 2008). Vectors can only bite a limited number of times during their lifetime. If some bites are spent on individuals that are non-competent to either amplify or transmit the pathogen, those bites are wasted. If VBD disperse to higher latitudes and altitudes under warmer climate conditions, they will invade ecosystems in which the natural level of biodiversity is relatively low. If the most competent vector or reservoir host becomes dominant, the risk for a high prevalence of the infection will increase (Schmidt and Ostfeld, 2001; Marcogliese, 2008). Furthermore, some host species may also spread from the tropics into the temperate zones. However, larger species typically spread at a slower rate than smaller, so for a significant time VBD will be moving down a gradient of biodiversity (Dobson et al., 2006). Therefore it may be wise to conserve biological diversity for the purely selfish reasons of protecting human and domestic animal health.

Figure 38.5. Wildlife can act as a reservoir for infectious diseases. Photo: Roland Mattsson, SVA.

Impact on Health of Wildlife

From one point of view wildlife may act as a disease reservoir, transmitting infections to domestic animals or humans. On the other hand, as mentioned above, wildlife and the ecological communities play a major role in regulating the natural abundance of zoonotic and animal pathogens that may infect humans and their domestic livestock (Dobson et al., 2006). When wildlife acquires a pathogen, the actual species may act as a reservoir for a certain disease for a prolonged time without displaying clinical symptoms. Since CC may alter the presence of wildlife species, the presence of several diseases may also be altered. Usutu virus (USUV) is an example of an infection causing mass mortalities of birds and it was diagnosed in Austria in 2001, which was the first time outside Africa (Brugger and Rubel, 2009). Blackbirds (*Turdus merula*) and great grey owls (*Strix nebulosa*) around Vienna were particularly affected. The virus was most probably transferred from Africa by migratory birds. The transmission cycle between birds and the mosquito vector (*Culex pipiens* spp.) depends strongly on the environmental temperature. USUV is a rather unknown virus since it is mainly a problem in wild birds, although it has shown zoonotic potential (Brugger and Rubel, 2009).

It has been noticed that hunting and fishing have been much harmed by recent CC in some regions, through stresses on animals driven by ecosystem shifts as sea ice retreat or warming of air and sea (ACIA, 2005). In food-insecure human populations, this alteration may already be contributing to malnutrition. There is an inability of many species to adapt to the relatively rapid ongoing changes in the ecosystems, resulting in a marked decline in population number or even to species extinction (Lovejoy, 2008). The decline or extinction of a key species can have severe effects on the whole ecosystem.

Habitat fragmentation, introduction of new species, scarcity of feed and water, deforestation and many other anthropogenic activities will stress wildlife populations. The impact of CC will be superimposed onto the effects of other stressors in ecosystems. This combination may work cumulatively or synergistically to exacerbate negative effects on populations or ecosystems (Marcogliese, 2008; Ytrehus et al., 2008).

CC affects both water availability and quality. Shortage of water may cause wildlife density to increase in the neighbourhood of a water source and this in turn could enhance the spread of infectious diseases, both within and between species. In the same way, forced migration could enhance transmission of disease due to intermingling of populations with introduction of novel diseases into non-immune populations.

Ecosystem Interactions and Dynamics

An ecosystem is a naturally occurring collection of organisms, plants and animals co-existing in a certain area and mainly dependent on the same environment. Different ecosystems are more or less sensitive to disturbances depending on environmental factors and inhabiting species. In general, more sensitive ecosystems are found in the Arctic and sub-Arctic climate zones than in temperate or tropical zones, largely due to lower biodiversity in the former.

Climate change directly affects ecosystems by changes in snow and ice cover, precipitation and temperature, etc. A secondary effect may be seen on biodiversity, population dynamics, home range size, migration patterns, habitat use, etc. The occurrence of new species may have effects on introduction of new infectious diseases, predation, vegetation and also on their competitive relationships. The possibilities for adaptation to a changed environment vary from one species to another. Effects may also arise from dense populations of insects, ticks, etc. considerably disturbing mammals and birds and also acting as vectors. For example, rodent breeding increases during mild weather and decreases in times of drought or heat. However, drought may also drive rodents to seek indoor sources of water, increasing the risk of disease transmission if humans and domestic animals come in contact with rodents or their secretions or excrement. Rodents play an important role as a reservoir host for many tick-borne diseases (Hartelt et al., 2008).

Changes in the vegetation may result from changed temperature and precipitation and thus indirectly affect the animal community at all ecosystem levels. A changed vegetation cover will affect the quantity, quality and accessibility of feed plants. Warm winter events in the Arctic and sub-Arctic zone will affect winter grazing conditions by the creation of ice cover on the ground (ACIA, 2005). Animals can exert further feedback on the vegetation and fundamental ecosystem processes such as soil nutrient recycling.

Species may be forced to migrate due to extreme weather events, changing ecosystems due to ongoing CC or other reasons causing unfavourable changes in habitat. Invasive exotic species may compete for feed and space but can also bring disease agents into a vulnerable host population, causing severe outbreaks of disease. Human penetration into remote areas may bring new pathogens to the ecosystem, but may also bring humans or domestic animals into contact with previously 'isolated' pathogens.

Marine Ecosystem

While temperature and moisture are the most important environmental parameters for land-living animals, temperature and pH are the two most important for aquatic species (Lovejoy, 2008). Ocean temperature increases while absorbing most of the heat added to the climate system. Aquatic life is often strictly adapted to a certain temperature range and will therefore be exposed to a significant level of stress at temperatures outside this range.

The uptake of carbon dioxide will make sea water more acidic. This profound change in oceanic chemistry has worrisome implications for species constructing their shell and skeleton out of calcium carbonate, including those that exist in untold numbers at the base of marine food chains (Lovejoy, 2008). In marine environments, plankton species as well as fish species have been shifting geographically (ACIA, 2005; Lovejoy, 2008).

Changed salt levels are another part of the problem. Depending on the different IPCC emissions scenarios based on different data and assumptions, the Baltic Sea is predicted to warm up as much as 4°C during the present century (IPCC, 2007). A higher temperature lowers the maturation of oxygen, which increases the stress already present in this marine ecosystem. High temperatures in surface water and greater hypoxia in bottom waters will confine fish to narrower bands of tolerable conditions, where they will seek refuge at higher densities. This in turn may serve to increase disease transmission (Marcogliese, 2008). Aquatic organisms under stress are more sensitive to infections than land-based animals and perform less well in terms of growth and reproduction (Marcogliese, 2008). Diseases may be introduced with invading exotic species or opportunistic pathogens, but diseases already present in the population may also cause disease outbreaks in aquatic species immune-suppressed due to stress.

Altered Persistence of Pathogens in the Environment

Although VBD are the main focus when concerning the impact of CC on infectious diseases, infections transmitted through the environment, e.g. water and soil, are also of interest. Infections may be transmitted to grazing animals or by feed and water. Increased air and water temperatures may improve the survival and proliferation of some pathogens (Mitscherlich and Marth, 1983). Although pathogens occur naturally in the environment, in many cases there is a man-made cause for the environmental presence of many pathogen microorganisms. Pathogens may be introduced into ecosystems when biological waste of agricultural, municipal or industrial origin is recycled in agriculture or deposited in the environment in other ways (Albihn, 2009). Once pathogen pollution has taken place, it is often impossible to control the spread of the infective agents either in time or space.

Some vegetative bacteria may survive in the environment for over one year if temperature, humidity, type of soil, etc. are favourable (Mitscherlich and Mart, 1983). Different soil types affect microbial survival in different ways owing to their texture, pore space, surface activity, moisture-holding characteristics, presence of essential nutrients and type of autochthonous microorganisms, etc. Some pathogenic bacteria such as *Salmonella* and VTEC can also multiply in favourable circumstances, such as in warm and humid weather (Mitcherlich and Marth, 1983). To enhance persistence, different *Salmonella* serovars adjust themselves to the surrounding circumstances in different ways (Mitcherlich and Marth, 1983).

Figure 38.6. Blood agar with *Salmonella Dublin* growth. Salmonella is a bacterium which epidemiology can change as a response to climate change. Photo: Bengt Ekberg, SVA.

Spore-forming bacteria such as *Bacillus* and *Clostridia* species are extremely persistent in the environment and may stay viable for decades buried in dry soil, where microbial activity is minimal. There are several reported examples of unexpected infections of cattle after digging in places where anthrax-infected carcasses had been buried decades previously. In the surface layer of soil, in competition with other organisms, anthrax spores can disappear within a few years. In Germany, it was noted that the anthrax losses of cattle on pastures drastically diminished during World War I. These pastures were flooded annually by water from a river and tanneries had drained their spore-contaminated sewage into this river (Mitcherlich and Marth, 1983). Cattle losses due to anthrax increased subsequently once contaminated hides were again imported after the war. Other interesting observations concerning the potential of *Bacillus anthracis* to multiply in soil have been made during periods of major climatic and ecological changes in which the soil microenvironment is altered in such a way that spores start to germinate and multiply (Mitscherlich and Marth, 1983).

Altered Presence and Epidemiology of Vector-borne Infectious Diseases

VBD have become an increasing problem in recent years and are of special interest in relation to CC and ecosystem changes (Gubler, 1998; Dufour et al., 2008). These disease agents are transmitted from one vertebrate host to another in a transmission cycle that includes a biological vector, which might be an insect, such as mosquitoes or fleas; or other arthropod, such as ticks or mites. In a broad sense, birds (Hubalek, 2004) and mammals (Randolph, 2008) may act as vectors, and sometimes the disease agent is transmitted physically or mechanically (e.g. on the feet of flies) but in this chapter the emphasis is on infectious agents carried by arthropod vectors and transmitted biologically. Some VBD often mentioned in connection with CC are Eastern and Western equine encephalitis, West Nile fever, blue tongue and Lyme disease (Epstein, 1995; Rogers and Randolph, 2006; Takken and Knols, 2007; Dufour et al., 2008; Gould and Higgs, 2009).

Historically, both animal and human populations in Europe and North America suffered from numerous pests and VBD. Introduction of hygiene measures, drugs and vector control caused the disappearance of many of these diseases from more economically developed and climatically temperate nations (CDC, 2004). In warmer climate zones, many diseases no longer seen in Europe and North America still thrive, causing serious health problems for humans and animals. Today we are aware of the risk of introduction and spread of exotic VBD to new regions and the persistent presence of such diseases in these new regions. This has been seen e.g. for West Nile fever introduced into the USA in 1999 (Blitvich, 2008) and for blue tongue disease in Northern Europe from 2006 (Gould and Higgs, 2009).

Most species, including humans, may act as a host for some VBD agent. However, only a few VBD (e.g. dengue fever and malaria) are strictly dependent on humans as a host, whereas most pathogens have the ability to infect a variety of hosts. A host might play the role of reservoir, where the microorganism matures or increases in number so that it can be picked up through ingestion by a vector and further transmitted, or it might be a dead-end host. In this case, the host might itself carry the agent and eventually become ill, but some characteristic of the host, agent, vector or environment prevents the agent from being further transmitted through the vector. A third type of host is known as an intermediate host, whereby an agent is present in the host during an intermediate stage of development.

Climate Change and Vector-borne Disease – How does it fit Together?

The concern today is for increased incidence of VBD is due to CC. Environmental conditions are important for the presence of many vectors (ticks, mosquitoes, sandflies, etc.), as well as the mammalian or avian host species they use to feed on or which act as reservoir species for the disease agent. The transmission cycle for infectious agents carried by vectors can be complex, and it is especially important to understand this cycle for any given agent and its geographical context in order to understand how climate plays a role. In general, vectors are favoured by increased temperature and high humidity. However, increasing precipitation is not always favourable for vectors. Heavy rainfall may wash away breeding sites, while drought, on the other hand, may slow rapid streams and create pools of stagnant water.

To a large extent, other non-climatic causes of ecosystem changes, such as pollution, land use changes, population increases and fragmentation of habitats, may also play an important role in changing vector transmission cycles. A particular concern as regards CC is the potential for temporary introductions of diseases supported by increased populations of vectors during exceptionally warm summers. While temporary outbreaks of most exotic VBD will most probably die out during the following winter, the impact on human and animal health may be considerable.

It is the complexity involved with the life cycles of the different hosts and vectors and the microorganism that make VBD difficult to predict and control, and this complexity also results in a disease ecology that can be modified due to CC. The distribution and size of populations of vectors and hosts will respond to changes in temperature and precipitation and these can often produce an outcome of changing disease patterns (Daniel et al., 2003).

An attempt has been made to simulate the influence of CC on replication and population size of ticks and the secondary effect on disease transmission of three different pathogens by using a mathematical model (Ogden et al., 2008). The seasonal synchrony of different development stages of ticks is important for the population biology and this is to a large extent temperature-dependent and therefore may be altered by CC. However, the results from the mathematical modelling were difficult to interpret, since the influence of CC was different between different tick species and geographical locations and even amongst different populations of the same species, as an effect of evolutionary processes (Ogden et al., 2008). In addition, the relationship between ambient weather conditions and vector ecology is complicated by the natural tendency for arthropod vectors to seek out the most suitable microclimates for their survival. Example of this are resting under vegetation or in pit latrines during dry or hot conditions or in culverts during cold conditions (Gubler et al., 2001).

Three important and contrasting examples of VBD agents for which the range has changed recently are West Nile fever virus (VNFV) in the United States, blue tongue virus (BTV) in Northern Europe and *Borrelia burgdorferi*, the bacterial agent of Lyme disease, in both the US and Europe. The three disease agents in question are spread to different hosts through different arthropod vectors: mosquitoes for WNFV, midges for BTV, and ticks for *B. burgdorferi*. Each disease system responds differently to ecosystem changes and changes in temperature, humidity, etc. Below we consider the disease transmission for each agent to illustrate the varied dynamics of VBD and CC.

West Nile Fever Virus

West Nile fever virus (VNFV) is a flavivirus and was identified in 1937 in the West Nile region of Uganda. After a period of mostly sporadic outbreaks reported in humans, geese and horses in Africa, southern Europe and the Middle East, it made a surprise jump across the Atlantic Ocean, to New York City, in 1999 (Hayes, 2001). From there it spread rapidly across North America, with more than 27,000 human cases and 25,000 equine cases of illness in the period 1999-2007 (Blitvich, 2008; Reiter, 2008). The transmission cycle of the virus involves avian reservoir hosts and a mosquito vector, most often from the genus *Culex* (Ebel et al., 2005; Reisen et al. 2005; Hamer et al., 2008).

Seasonality is a key component of climate. Many VBD are seasonal, and in the northern latitudes they are most often seen in the warmer months, tied to the life cycle of the vector. West Nile fever in North America peaks in mid-August, and nearly all cases reported occur between July and October (CDC, 2008). Now consider the interaction between climate and WNFV in the United States Great Lakes region. The *Culex* mosquito that transmits WNFV in this region is affected in several ways by increased temperatures. A female *Culex* mosquito deposits eggs about 5-6 times during her life. With a longer period in

Figure 38.7. Grazing horses can be infected with Wets Nile fever virus from bites of *Culex vectors*. Photo: Bengt Ekberg, SVA.

Figure 38.8. A mosquito from the genus *Culex*, which is one of the vectors that can spread West Nile fever virus. Photo: Anders Lindström, SVA.

which warm temperatures prevail, more opportunities for depositing eggs are available (Reiter, 2008). Mosquitoes develop from larvae to pupae to adult. They develop faster when temperatures are warmer. When temperatures reach about 17°C and daylight is reduced, mosquitoes either die off or enter their winter state of inactivity, called diapause (Dohm et al., 2002). Currently in this region, there are about 18 weeks a year in which temperatures are, on average, above 17°C. An increase in overall temperature would extend this time period, allowing more vectors to be produced. During the spring, the active season for mosquitoes could start earlier too, since water temperatures greater than 15°C allow their proliferation.

In addition to the potential for more vector mosquitoes to be present, those mosquitoes may transmit the virus more efficiently at warmer temperatures due to a shorter extrinsic incubation period (the time it takes from when a mosquito receives the virus until it is able to transmit the virus to its next host). For example, at 18°C, this period can be more than three weeks, while at 30°C, the incubation period can be as short as just four days. Overall, the replication of the virus is temperature-dependent and warmer temperatures increase the replication rate (Saegerman et al., 2008).

It is interesting to note that both the epidemics and clinical symptoms in different host species are totally different in Europe and Africa compared with North America (Reiter, 2008). The reason for this may in part be the development of immunity among wildlife in Europe, but to a large extent this difference is unknown (Reiter, 2008). Climate change is clearly very important in determining whether or not WNFV is efficiently transmitted between vertebrates and mosquitoes, but CC has not played an obvious role in the epidemic outbreaks seen in North America. More important factors have probably been the availability of competent vector species, the wide range and large numbers of susceptible species of migratory birds and different human activities (Reiter, 2008; Gould and Higgs, 2009).

Figure 38.9. The Chicago suburbs grew quickly in the 1950s. The neighborhoods that developed have, today, proven to have ecosystem characteristics that are conducive to the transmission of West Nile virus (WNV) to avian and human hosts. Railroads cutting through the neighborhoods are one of the primary semi-natural areas, with good bird habitat for many species. Here, the photograph is tagged with information related to a research project on WNV transmission dynamics. Photo: Marilyn Ruiz.

Lyme Borreliosis

Lyme borreliosis is caused by *Borrelia burgdorferi*, a spirochete bacterium carried by ticks largely from the genus *Ixodes*. It emerged in the 1980s as an important illness of dogs, humans and some other species in North America and in Europe (Vrbova and Middleton, 2006). The *Ixodes* tick goes through three stages over two years as the larval tick matures to a nymph and then becomes an adult. The tick requires a blood meal to mature to each stage. The tick poses the greatest risk to humans and dogs in the spring and summer of its second year, when it is a nymph. At that point, it may have become infected as a larva and can now infect its new host (Hubalek, 2009). In the later summer and autumn, in the adult stage, it prefers a deer host but can also infect humans during that period. In North America, the reservoir host for *B. burgdorferi* is the white-footed mouse (*Peromyscus leucopus*) (LoGiudice et al., 2003).

While reduced summer precipitation leads to fewer ticks due to desiccation, increases in winter temperatures may allow for an extension of the northern boundary of the tick (Jones and Kitron, 2000; Lindgren et al., 2000). For example, the tick *Ixodes ricinus* was not found above 61°N prior to 1980, but has now found its way to the Baltic region, at 66°N (Gray et al., 2009). Increased precipitation appears to play a role in tick abundance, although not in excess (McCabe and Bunnell, 2004). It is also possible that the population of white-footed mice will

Figure 38.10. Tick on grass. A tick is a common vector for the spirochete bacteria Borrelia burgdorferi, the causing agent of Lyme borreliosis. Photo: Bengt Ekberg, SVA.

be enhanced with an increase in important food sources that can vary due to climatic factors. Ostfeld et al. (2006) noted that acorn (*Quercus* spp.) abundance two years prior led to higher abundance of white-footed mice and this was linked to *B. burgdorfer* disease. While this empirical analysis may not be directly generalised to other regions, it illustrates that an indirect path of climate that affects the acorn crop also affects the number of rodents (and deer) in an area, which in turn influences the number of ticks at various stages.

Tick-borne disease systems are very sensitive to CC through the impact of temperature and moisture stress on population size. However, it is not reasonable to expect tick abundance or seasonal activity patterns to respond to CC in ways that inevitably increase the risk of tick-borne infections (Dufour et al., 2008; Randolph, 2008).

Blue Tongue Virus

Blue tongue virus (BTV) recently emerged into northern Europe. Before 2006, BTV was not noticed further north than around the Mediterranean Sea (Mehlhorn et al., 2008; Purse et al., 2008; Saegerman et al., 2008). This has raised great concern as to the mechanism and implications for the rapid change in the range of the virus. The vector of BTV is *Culicoides* biting midges, found very commonly in and around cattle and sheep farms (Zimmer et al., 2008). BTV is an Orbivirus and can infect almost any ruminant. Wild ruminants, including red deer and roe deer, have been found to have antibodies to BTV, and the potential for these to serve as reservoirs is an important consideration in control efforts (Ruiz-Fons et al., 2008). However, the recent experience with BTV-8 in Europe suggests that 'spillover' into deer occurs only when there are high infection levels in farmed ruminants. In England, at present, wild ruminants do not appear to be an important reservoir for BTV-8 (VLA, 2007).

When illness is seen, the virus causes haemorrhagic disease, especially in sheep. BTV is a common and important disease of domestic livestock around the world, and there are 24 different serotypes and numerous strains. The remarkable increase in northern Europe of the serotype BTV-8 has caused millions of cases of illness, with disruptions of trade. Control of this disease through vaccines is still associated with several problems and vaccines need further development (Purse et al., 2008; Saegerman et al., 2008; Gould and Higgs, 2009).

Recent CC in Europe are being strongly considered as the driving factor behind the northern expansion and effectiveness of *Culicoides* vectors in transmitting BTV (Purse et al., 2008). These insects are small and not strong fliers, but can be carried as aerial plankton for distances up to 700 km (Sellers, 1992). Drivers of the expansion of BTV in Europe include several environmental conditions such as a minimum temperature during a minimum period of time. Temperature seems to play the largest role (Purse et al., 2008). In particular, the warmer winters allow more adults to survive into the spring. Then, during the warm months, the extrinsic incubation period of the virus in the vector is shorter at higher temperatures and viral replication is also enhanced. Transmission conditions are ideal when temperatures are not so warm so that survival of the adult vector is threatened (above about 30°C) but when other conditions are optimised. Other effects of a higher ambient temperature are an increased biting rate and the possibility of non-vector competent *Culicoides*

Figure 38.11. Biting midges, *Culicoides*, are vectors that can spread Blue tongue virus. Photo: Anders Lindström, SVA.

species and individuals becoming vector-competent and the *Culicoides* population increasing in density due to the higher reproduction rate (Wittmann and Baylis, 2000). The 'baton effect' is also dependent on a warmer climate. Hence, when a vector-competent species expands its distribution northwards, it may bring BTV into the range of other *Culicoides* species that occur much further north and are, or become, vector-competent. This species may then spread the virus over a large geographical area in a short time (Wittmann and Baylis, 2000). The vector capacity for BTV should be considered as temporally and geographically variable within and between *Culicoides* species, due to genetic, environmental and unknown factors (Purse et al., 2008). This may well be one of the reasons for the present northward spread of BTV in Europe.

The impact of CC is obvious on vector range expansion and the northerly establishment of BTV-8 in Europe but the transportation of infectious ruminants and the wind-borne dispersal of infected midges are believed to be highly significant contributing factors to the introduction of BTV (Purse et al., 2008; Gould and Higgs, 2009).

Vector Control Efforts

In developed countries, the systems currently used to control exotic animal infectious diseases are based mainly on monitoring animal movements and on detecting and destroying infected animals. These methods may be inadequate for VBD when a reservoir is established in vectors and/or in reservoir species among wildlife. Different control strategies for vectors may help in disease prevention and control, especially at times with high vector population density or when new vector species, e.g. the Asian tiger mosquito (*Aedes albopictus*), is detected in a new region.

Great vector control programmes using insecticides/biocides in the environment are currently in place in several continents to fight major human infections such as malaria and leishmania. However, the lesson that has been learned from most of these efforts is that as soon as the treatment is finished, the insect problem becomes just as bad as before (WHO, 2004). Due to CC, the problems concerning a number of VBD may increase. This may cause the use of pesticides to be intensified, both in the environment and directly on animals and humans, in barns and houses, etc. For domestic animals insect repellents may be an option, but are not always practical or realistic. Impregnated ear tags or regular dipping of production animals in insecticide pharmaceuticals are a routine in certain areas. Such treatment, especially against ticks, may become more widespread in the future. Here organic production is facing a special problem, since this kind of treatment is not allowed.

There are several problems associated with chemical control efforts. Firstly, it is not uncommon for the target organism to adapt or develop resistance to the pesticide. Secondly, a substance may be harmful to the health of the humans who have to handle it and there may also be residues of the substance in food. In the UK the use of pyrethroids on livestock has raised environmental issues and in the US pyrethroids used to control WNFV have come under scrutiny for possible carcinogenicity in humans (Gammon, 2007). In Europe, it has been suggested that chemical vector control should be limited to application in indoor conditions (WHO, 2004), due to possible impacts on public health and the environment.

The use of biological control for the management of pest insects pre-dates the modern pesticide era. The first major successes in classical biological control occurred with exotic pests in plants controlled by natural enemy species collected from the area of origin of the pest (Bale et al., 2008). Biological control also includes the introduction of enemy microorganisms for pests and vectors. The use of environmental risk assessment before introduction of non-native control agents is essential.

A bacterial toxin (Bti) produced by *Bacillus thuringiensis israelensis* is commonly used in a variety of habitats against a multitude of species of mosquitoes in several countries (Lacey, 2007). Factors that influence the larvicidal activity of Bti include species of mosquito and their respective feeding strategies, rate of ingestion, age and density of larvae, habitat factors, formulation factors (type of formulation, toxin content, how effectively the material reaches the target) and means of application and frequency of treatments. Bti is reported to be very specific to mosquitoes and therefore only the target organisms are affected (Lacey, 2007). However, unwanted side-effects have been reported for Bti and there is discussion as to whether this is a true biological treatment.

With mosquito vectors, environmental control, including reduction of oviposition sites, and larvicide application are important aspects of mosquito control. Many vectors, e.g. several species of both *Culicoides* and *Culex*, are habitat generalists breeding in a range of moist microhabitats including man-made possibilities such as irrigation channels, drainage pipes, dung heaps, flooded cellars, etc. (Purse at al., 2008; Reiter, 2008). Such spots may be virtual factories for vectors. Effective sewage disposal and the elimination of standing polluted water may limit the risk by reducing population of *Culicoides* (Purse et al., 2008) and *Culex* mosquitoes (Reiter, 2008).

Integrated pest or vector management is an approach combining biological control agents, environmental management, personal protection and the judicious use of pesticides, etc. Here it is necessary to carefully consider the transmission cycle and then focus on several methods to reduce the opportunities for host-vector interaction and to reduce the proportion of infectious vectors (Ginsberg, 2001; Peter et al., 2005). For example, keeping stock indoors at times of high vector activity may reduce transmission of an infection. Vaccination of animals or humans, when an effective vaccine is available, may reduce the proportion of infectious vectors. In the best case scenario vaccination of a reservoir species may eliminate the infection from a vector population. However for e.g. BTV, where 24 different serotypes have been identified, a multivalent vaccine protecting against several of these serotypes would be complicated and costly to produce (Gould and Higgs, 2009). However, when wildlife species act as reservoirs vaccination may be more complicated. Integrated vector management, when successful, may counteract pesticide-resistant vectors, enhance the withdrawal of chemicals and minimise the usage of pesticides.

For adaptation to CC it is vital that more effective, sustainable approaches for vector control are developed. Such methods may be based on genetics, spread of sterile males, etc. In addition, public education and preventive advice are important in protecting humans from VBD. Protective clothing, insect repellents and mosquito nets may be efficient for protecting humans from bites, when available and affordable.

Conclusions

Nature is on the move in response to climate change. A great deal of change lies ahead, with a negative global impact predicted on food and feed production. Shortage of water will become an increasing problem. At present ecosystem change and biodiversity decline is a fact. However, there are many other changes that also influence ecosystem health and these may even be of higher importance than climate change. New health problems will emerge, affecting domestic animals, wildlife and humans. A special focus is needed on vector-borne diseases; Blue tongue, West Nile fever and Lyme borreleosis are concrete examples. Their presence and epidemiology are largely dependent on ecosystem changes and since they involve wildlife, vector-borne diseases may be very hard to control. Heat stress may cause suppressed production and lowered reproduction performance in domestic animals. Another effect of heat stress is a suppressed immune response, which may result in infectious diseases being more easily acquired. However, positive effects of climate change may also be seen, especially in the temperate and cold northern latitudes. These may come from a prolonged vegetation period, leading to improved feed production and a prolonged grazing period for domestic animals.

We have to consider – and handle – both known and unknown health effects. To develop adaptation strategies concerning health effects, a multidisciplinary and deep understanding is needed of the complex and dynamic interactions between climate change, ecosystem health, wildlife, domestic animals and humans. Increasing knowledge concerning infectious diseases can be used to forecast, and in the best case prevent, some effects of climate change. World-wide, intensive and effective surveillance of domestic and wild animal health is essential and efficient international networking should be compulsory. Early detection and action, before an infectious disease has been widely spread, are essential for successful control and eradication. So for some diseases and in some cases, action must be taken without complete knowledge. Last but not least, vector control efforts must be further developed and the reliance on chemical control must be reduced.

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Chapter 18

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List of Authors

Name of author	University/Affiliation	Email adress
Adriane M Adkesson	College of Veterinary Medicine, University of Illinois at Urbana-Champaign, Urbana, IL, USA	animaldoc2@yahoo.com
Ann Albihn	National Veterinary Institute, Uppsala, Sweden	ann.albihn@sva.se
Magnus Appelberg	Swedish Board of Fisheries, Öregrund, Sweden	magnus.appelberg@fiskeriverket.se
Rodney B Baker	Population Medicine and Veterinary Public Health, College of Veterinary Medicine, North Carolina State University, Raleigh, NC, USA	butch_baker@ncsu.edu
Val R Beasley	Department of Comparative Biosciences, College of Veterinary Medicine, University of Illinois at Urbana-Champaign, Urbana, USA	val@illinois.edu
Björn Bengtsson	National Veterinary Institute, Uppsala, Sweden	bjorn.bengtsson@sva.se
Anders Bignert	Swedish Museum of Natural History, Stockholm, Sweden	anders.bignert@nrm.se
Jeffrey D Brawn	Department of Natural Resources and Environmental Sciences, College of Agricultural, Consumer and Environmental Sciences, University of Illinois at Urbana-Champaign, Champaign, IL, USA	jbrawn@illinois.edu
David B Bunnell	Great Lakes Science Center, United States Geological Survey, Ann Arbor, MI, USA	dbunnell@usgs.gov
Steve Bursian	Department of Animal Science, College of Agriculture and Natural Resources, Michigan State University, East Lansing, MI, USA	bursian@msu.edu
Bruce M Chassy	Department of Food Science and Human Nutrition, College of Agricultural, Consumer and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, IL, USA	bchassy@illinois.edu
Joanne C Chee-Sanford	Invasive Weed Management Unit, Agricultural Research Service, United States Department of Agriculture, Urbana, IL, USA	Joanne.CheeSanford@ars.usda.gov
Natalia Chukalova	AtlantNiro, Kaliningrad, Russia	
Scott B Citino	White Oak Conservation Center, The Howard Gilman Foundation, Yulee, FL, USA	scottc@wogilman.com
Elisabet Ekman	Swedish University of Agricultural Sciences, Uppsala, Sweden	elisabet.ekman@slu.se
Kelly A Estes	Illinois Natural History Survey, Institute of Natural Resource Sustainability, University of Illinois at Urbana-Champaign, Champaign, IL, USA	kcook8@illinois.edu
Sverker Evans	The Swedish Agency for Marine and Water Management, Göteborg, Sweden	sverker.evans@snv.se
Anne Fairbrother	EcoSciences, Exponent, Seattle, WA, USA	afairbrother@exponent.com
Guy Fitzgerald	Department of Clinical Sciences, Faculty of Veterinary Medicine, University of Montreal, Saint-Hyacinthe, QC, CAN	guy.fitzgerald@umontreal.ca
John Fitzsimmons	Great Lakes Laboratory for Fisheries and Aquatic Sciences, Fisheries and Oceans Canada, Burlington, ON, CAN.	John.Fitzsimons@dfo-mpo.gc.ca
Stephen Garbaciak Jr.	ARCADIS U.S., Chicago, IL, USA	Steve.Garbaciak@arcadis-us.com
Thomas R Gillespie	Department of Environmental Studies, College of Arts and Sciences, Emory University, Atlanta, GA, USA	thomas.gillespie@emory.edu
Lennart Gladh	World Wildlife Fund, Stockholm, Sweden	lennart.gladh@www.se
Hans Gustafsson	Swedish Dairy Association , Department of Clinical Sciences, Uppsala, Sweden	hans.gustavsson@svenskmjolk.se
Anders Hellström	National Veterinary Institute, Uppsala, Sweden	anders.hellstrom@sva.se
John A Herrmann	Veterinary Clinical Medicine, College of Veterinary Medicine, University of Illinois at Urbana-Champaign, Urbana, IL, USA	jah1110@illinois.edu
Marja-Liisa Hänninen	University of Helsinki, Faculty of Veterinary Medicine, Helsinki, Finland	marja-lisa.hanninen@helsinki.fi
Ari Hörman	European Commission, Directorate-General for Health and Consumer Affairs, Brussels, Belgium	ari.horman@milnet.fi
David J Jude	Great Lakes Fishery Laboratory, School of Natural Resources and Environment, University of Michigan, Ann Arbor, MI, USA	djude@umich.edu
Ivan G Krapac	Illinois State Geological Survey, Institute of Natural Resource Sustainability, University of Illinois at Urbana-Champaign, Champaign, IL, USA	krapac@isgs.illinois.edu
Richard L Lampman	Illinois Natural History Survey, Institute of Natural Resource Sustainability, University of Illinois at Urbana-Champaign, Champaign, IL, USA	richlamp@uiuc.edu
Jeffrey M Levengood	Department of Veterinary Biosciences, University of Illinois at Urbana-Champaign, Urbana, IL, USA	jleven@inhs.illinois.edu

David Macarus	Office of Research and Development, United States Environmental Protection Agency, Chicago, IL, USA. (deceased)	
Roderick I Mackie	Department of Animal Sciences, College of Agricultural, Consumer and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, IL, USA	r-mackie@illinois.edu
Charles P. Madenjian	Great Lakes Science Center, United States Geological Survey, Ann Arbor, MI, USA	cmadenjian@usgs.gov
Ulf Magnusson	Faculty of Veterinary Medicine and Animal Health, Swedish University of Agricultural Sciences, Uppsala, Sweden	ulf.magnusson@slu.se
Daniel Martineau	Department of Pathology and Microbiology, University of Montreal, Saint-Hyacinthe, QC, CAN	daniel.martineau@umontreal.ca
Philip S. Miller	Conservation Breeding Specialist Group, International Union for Conservation of Nature, Apple Valley, MN, USA	pmiller@cbsg.org
Torsten Mörner	National Veterinary Institute, Uppsala, Sweden	torsten.morner@sva.se
Leif Norrgren	Faculty of Veterinary Medicine and Animal Health, Swedish University of Agricultural Sciences, Uppsala, Sweden	leif.norrgren@slu.se
James E Pettigrew	Department of Animal Sciences, College of Agricultural, Consumer and Environmental Sciences, University of Illinois at Urbana-Champaign, Urbana, IL, USA	jepettig@illinois.edu
Kishore Rajagopalan	Illinois Sustainable Technology Center, Institute of Natural Resource Sustainability, University of Illinois at Urbana-Champaign, Champaign, IL, USA	kishore@istc.illinois.edu
Galina Rodjuk	AtlantNiro, Kaliningrad, Russia	rodjuk@atlant.baltnet.ru
Marilyn O'Hara Ruiz	Department of Pathobiology, College of Veterinary Medicine, University of Illinois at Urbana-Champaign, Urbana, IL, USA	moruiz@illinois.edu
Mati Roasto	Estonian University of Life Sciences, Institute of Veterinary Medicine and Animal Sciences, Tartu, Estonia	mati.roasto@emu.ee
Anna Roos	Swedish Museum of Natural History, Stockholm, Sweden	anna.roos@nmr.se
Froucke Vermeulen	Department of Animal Science, College of Agriculture and Natural Resources, Michigan State University, East Lansing, MI, USA	fverm500@msu.edu
Chip Weseloh	Environmental Conservation Branch, Canadian Wildlife Service, Environment Canada, Toronto, ON, CAN	Chip.Weseloh@ec.gc.ca
Martin Wierup	Faculty of Veterinary Medicine and Animal Health, Swedish University of Agricultural Sciences, Uppsala, Sweden	martin.wierup@slu.se
Martha Wolgamood	Wolf Lake State Fish Hatchery, Michigan Department of Natural Resources, Mattawan, MI, USA	wolgamom@michigan.gov
Anthony Yannarell	Department of Natural Resources and Environmental Sciences, College of Agricultural, Consumer and Environmental Sciences, University of Illinois at Urbana-Champaign, Champaign, IL, USA	acyann@illinois.edu
Matthew Zwiernik	Department. of Animal Science, College of Agriculture and Natural Resources, Michigan State University, East Lansing, MI, USA	zwiernik@msu.edu

Authors can be contacted through the main editor, Leif Norrgren: *leif.norrgren@slu.se*