Ecological Effects of the Use of Chemicals in Aquaculture

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ABSTRACT

Many aquaculture chemicals are, by their very nature, biocidal, and may be released to the surrounding environment at toxic concentrations either through misuse, or in some cases, even by following generally accepted procedures for use. Thus, there is a potential for mortality of non-target organisms. Illustrations are provided of three classes of aquaculture chemicals and their effects on non-target biota: 1) use of a carbaryl pesticide and mortality of non-target invertebrates; 2) use of an organophosphate parasiticide and suspected effects on nearby biota; and 3) effects of antibacterial residues in aquatic sediments on the associated microbial community. Efforts to assess the risks posed by aquaculture chemicals are often frustrated by a lack of information on environmental fate and effects, and data needs to resolve this situation are identified.

INTRODUCTION

Many aquaculture chemicals are, by their very nature, biocidal, and achieve their intended purpose by killing or slowing the population growth of aquatic organisms. Chemicals used in this manner include:

- antifoulants
- disinfectants
- algicides
- herbicides
- pesticides
- parasiticides
- antibacterials

For the purpose of this assessment, mortality of the target organism is accepted as a given, and, from the perspective of the aquaculturist, a desirable outcome. Mortality of a pest species is in itself an ecological effect with potential implications to the surrounding ecosystem, but the implicit assumption is that the commercial value of elimination of the pest outweighs the ecological value of its presence. The emphasis in this discussion is on effects on non-target species.

In order to illustrate the potential range of ecological effects, three general classes of aquaculture chemicals are discussed below: 1) pesticides, 2) parasiticides, and 3) antibacterials. This assessment does not consider human health aspects or stimulation of antibacterial resistance in natural microbial communities, as both these topics are discussed elsewhere in this volume.

DISCUSSION

Pesticides

Perhaps the greatest potential for ecological effects arises from the use of aquaculture chemicals to remove pest species from the surrounding environment. An example of such an approach is the use of the carbaryl pesticide Sevin to control ghost shrimp (*Callianassa californiensis*) and mud shrimp (*Upogebia pugettensis*) in oyster culture areas of Washington State, U.S.A. Oysters (*Crassostrea gigas*) are produced by bottom culture on intertidal mudflats in Willapa Bay and Grays Harbor, Washington. Dense infestations of burrowing shrimp reduce production, either by creating suspended sediments which smother oyster seed, or by softening the substrate to the point where it cannot support the adult oysters (WDF/WDOE 1985). Since 1963, the pesticide Sevin has been used to control burrowing shrimp, with application on exposed mudflats during low tide, either by hand or by aerial spraying from helicopters. Application requires approval of a state fisheries biologist, and is given only when shrimp burrow entrances exceed a density of 10 burrows m^2. Application rate is 8.5 kg active ingredient/ha, with no more than 20 ha sprayed in a single treatment, and no more than 320 ha sprayed statewide each year. A single oyster bed may be treated in two successive years, but on average, treatment of any given bed is required once every six years.

Immediately after spraying, maximum observed concentrations of Sevin in overlying water have been reported to range from 1-20 ppm (WDF/WDOE 1985, 1989 and references therein). Sevin concentrations in water are reduced to 0.1 ppm within hours, either by hydrolysis to 1-naphthol or adsorption to sediments. In sediments, however, residues may persist for a few weeks (WDF/WDOE 1985). Sevin is highly toxic not only to burrowing shrimp, but to non-target organisms. In plots sprayed at a dosage of 11.3 kg/ha, the densities of gaper clams (*Tresus capax*) and bent-nosed clams (*Macoma baltica*) were reduced 69% and 28%, respectively (Armstrong and Millemann 1974). The 24 hr LD_{50} for many arthropods, including the Dungeness crab (*Cancer magister*) is approximately 0.1 ppm (Table 1), suggesting that these species may experience mortality at the concentrations observed after application.

Effects of Sevin spraying on Dungeness crab are of particular concern because of the commercial importance of the Dungeness crab fishery. Crabs, particularly juveniles, are at risk not only because aqueous concentrations of Sevin may exceed lethal levels, but because of their behavior of ingesting the carcasses of burrowing shrimp and other invertebrates that have been killed by Sevin treatment. Invertebrates collected from treated areas contain 5-75 ppm carbaryl (Table 2). Ingestion of food containing 43-167 ppm carbaryl has been shown to result in 7-80% mortality within 24 h in Dungeness crabs (Tufts 1988). Dungeness crabs have also been shown to develop irreversible paralysis after ingesting bivalves that had been exposed to 1 ppm Sevin (Buchanan *et al.* 1970).

It has been estimated that during the period 1984-1988 an average of 9,300 juvenile and 73 adult crabs were killed per hectare treated with Sevin. With an estimated 0.4% of juveniles surviving to a harvestable size, the mortality rate in adult crab equivalents is 110 individuals/ha, or 35,100 individuals over the entire 320 ha treated each year (WDF/WDOE 1989). This mortality represents an annual loss of $47,000 to the Dungeness crab fishery, but this cost is overshadowed by the value of the $7 million/yr oyster industry, and the loss of a large proportion of the beds if Sevin treatment were not done. This comparison, however, fails to take into account the ecological value of the Dungeness crabs and the many non-commercial species that are adversely affected by carbaryl treatment. Lacking any mechanism to place a monetary value on these species, the ecological impacts of aquaculture chemicals are often ignored, and regulatory decisions based largely on commercial considerations.

Table 1. Carbaryl LC_{50} values for selected invertebrate species (modified from WDF/WDOE 1989).

Species	Exposure Time (h)	LC_{50} (ppm carbaryl)
Arthropoda		
Amphipoda (*Gammarus* spp.)	24	0.04
Mud shrimp larva	24	0.03-0.16
Ghost shrimp larva	48	0.17-0.47
Ghost shrimp adult	24	0.13
Dungeness crab larva	24	0.08
Dungeness crab juvenile	24	0.08
Dungeness crab adult	24	0.49
Fiddler crab larva	24	0.1
Mollusca		
Bay mussel larva	48	1.4-2.9
Pacific oyster larva	48	1.5-2.7
Adult cockle clams	24	7.3

Table 2. Average carbaryl residues (mg/kg wet weight) found in the tissues of invertebrates after pesticide application (from Tufts 1988).

Rate of Application (kg/ha)	Residues in Burrowing Shrimp	Residues in Annelids
11.3	13.8	75.7
8.5	8.7	57.0
5.6	5.3	58.6

Parasiticides

Organophosphate compounds are occasionally used in aquaculture for a wide variety of applications, including control of ectoparasitic crustaceans, treatment of trematode or ciliate infections in shrimp hatcheries, or removal of mysids from shrimp ponds. They are sold under a variety of trade names including Nuvan®, Neguvon®, Aquaguard®, Dipterex®, Dursban®, Demerin®, and Malathion®. Neguvon® (trichlorphon) and its degradation product Nuvan® (dichlorvos) are used for treatment of ectoparasitic crustaceans such as salmon lice, *Lepeophtherius salmonis*, on marine fishes, or *Argulus* sp. and *Lernaea* sp. on freshwater fishes. Their use has generated considerable controversy in the UK regarding toxicity to non-target species (Ross 1989). Egidius and Møster (1987) provide one of the few pieces of evidence, albeit anecdotal, of non-target organism toxicity associated with the use of these parasiticides. In Norwegian salmonid cage culture, growers surround the cages with tarpaulins, and add Neguvon® to achieve a concentration ranging from 10 up to 300 ppm. After completion of treatment, the tarpaulins are removed, allowing the solution to disperse into surrounding waters. As a result of unexplained mortality of lobsters (*Homarus gammarus*) held near a salmon farm, subsequent investigation showed the lobsters were extremely sensitive to these parasiticides, with mortality occurring after 24 h exposure to 0.5 ppm Neguvon or 0.1 ppm Nuvan (Egidius and Møster 1987).

Antibacterials

Antibacterials are commonly administered as a bath or as a feed supplement. As a bath, there is an obvious route of release of unabsorbed antibacterials to the surrounding environment via the effluent. Even as a feed supplement, however, this loss can occur either via uningested waste feed or through elimination in the feces or urine. Oxytetracycline, one of the most widely used antibacterials in aquaculture worldwide, is notorious in this regard. The vast majority of oxytetracycline supplied in medicated feed can be found in hatchery effluent at concentrations that account for nearly all of the drug supplied (Smith *et al.* 1994). It has been estimated that only 7-9% of the oxytetracycline ingested is absorbed during gut passage in freshwater rainbow trout (Cravedi *et al.* 1987). Thus, even if all medicated feed were ingested, >90% of the drug could leave the facility via fecal matter or dissolved in the effluent. Fecal sources are also important for oxolinic acid, where 62-86% of the ingested drug has been shown to remain in the feces of rainbow trout (Cravedi *et al.* 1987). Conversely, chloramphenicol is absorbed very efficiently in the gut, and <1% of drug ingested is released via the feces (Cravedi *et al.* 1987).

Antibacterial residues in the surrounding water are not a major concern, given the rapid dilution and susceptibility of many drugs to photodegradation. However, the sediments serve as a long-term reservoir for residues of many drugs (Table 3). Of the antibacterials examined, oxytetracycline is among the most persistent, and is lost from the sediment only by dissolution and diffusion into the overlying water (Samuelsen 1989). Under conditions of rapid sedimentation, as would be expected near many aquaculture facilities, sediment containing oxytetracycline residues may be quickly buried, and the drug may persist indefinitely.

The quinolones oxolinic acid and flumequine are also very persistent in sediments, and detectable residues may remain for 6 mo or more after drug treatment. Sulfadiazine and sulfadimethoxine have been shown to persist for many months in a sealed vial (Samuelsen *et al.* 1994), but are far less persistent in an open system with flowing water overlying the sediments (Capone *et al.* 1996).

Residues of trimethoprim, ormetoprim and furazolidone are all relatively short-lived in aquatic sediments, with concentrations becoming immeasurable within 2 mo or less. Furazolidone, in particular, is rapidly degraded microbially, and has a half-life of less than 1 d in aquatic sediments (Samuelsen *et al.* 1991).

The organic-rich sediments typical in the vicinity of aquaculture operations are characterized by intense microbial activity including high rates of oxygen consumption and sulfate reduction. Since antibacterial residues can be found in sediments surrounding aquaculture operations and persist there for a year or more in the case of some antibacterials, the question arises as to what affect these residues may have on natural microbial communities in the sediments. Despite the potential environmental effects, little work has been done on changes in sedimentary microbial abundance or biogeochemical processes following aquacultural use of antibacterials. The most extensive data on biogeochemical effects of antibacterial treatment come from work recently completed at a salmon net-cage farm in the United States and in laboratory microcosms (Capone *et al.* 1994, Herwig and Gray 1997, Herwig *et al.* 1997). Use of 186 kg oxytetracycline at a farm and the presence of residues of the drug in the sediments (1-4 mg/kg) had no measurable affect on total microbial density in the farm sediments, the flux of ammonium from the sediments, sulfate concentrations in pore water, or sediment oxygen consumption. Dosing of microcosms with oxytetracycline and Romet® 30 (sulfadimethoxine and ormetoprim) at a rate intended to mimic field conditions also had no affect on the same parameters. The lack of an effect on microbial density or activity was due to an observed increase in antibacterial resistance that compensated for the loss of more susceptible microorganisms, complexation of the oxytetracycline with divalent cations, and a rapid disappearance of the sulfadimethoxine.

Table 3. Persistence of antibacterial residues in sediments. Bold text indicates data from sediments beneath a farm; all other data from laboratory microcosms. (modified from Weston 1996).

Antibacterial	Half-life (d)	Persistence[1] (d)	Farm or Lab. Data	Reference
Flumequine	155	>185	L	Hansen *et al.* 1992
		>180	L	Samuelsen *et al.* 1994
Furazolidone	0.75	9	L	Samuelsen *et al.* 1991
Ormetoprim		>2-<23	L	Capone *et al.* 1996
		<30	L	Samuelsen *et al.* 1994
Oxolinic acid		**6**	**F**	Björklund *et al.* 1991
		>77	L	Björklund *et al.* 1991
	165	>185	L	Hansen *et al.* 1992
	48	>210	L	Samuelsen 1992
		>180	L	Samuelsen *et al.* 1994
Oxytetracycline	**9-419**	**>7->308**	**F**	Björklund *et al.* 1990
		>12	**F**	Björklund *et al.* 1991
		>77	L	Björklund *et al.* 1991
		>300	**F**	Capone *et al.* 1996
	36	>60	L	Capone *et al.* 1996
	16	**>33-<71**	**F**	Coyne *et al.* 1994
	125	>185	L	Hansen *et al.* 1992
		>84	**F**	Jacobsen & Berglind 1988
	70	>84	L	Jacobsen & Berglind 1988
	32	**>39**	**F**	Samuelsen 1989
	30-64	>220	L	Samuelsen 1989
	55	>210	L	Samuelsen 1992
	87-144	**>550**	**F**	Samuelsen *et al.* 1992
		>180	L	Samuelsen *et al.* 1994
Sulfadiazine		>180	L	Samuelsen *et al.* 1994
Sulfadimethoxine		>2-<22	L	Capone *et al.* 1996
		>180	L	Samuelsen *et al.* 1994
Trimethoprim		>30-<60	L	Samuelsen *et al.* 1994

[1]Length of time after medication during which detectable residues were present in the sediment.

In the only other study on the topic, Hansen *et al.* (1992) reported a 40-50% reduction in microbial density and a >90% decrease in the rate of sulfate reduction in sediments dosed with either oxytetracycline, oxolinic acid or flumequine. The disparity between these results and the lack of microbial response observed in Capone *et al.* (1994) is probably due to differences in the concentrations of antibacterials used. Hansen *et al.* (1992) dosed the sediments with 400 mg/kg oxytetracycline and 100 mg/kg of the other drugs. However, Capone *et al.* (1994) used 5-15 mg/kg oxytetracycline in the microcosms and reported <4 mg/kg in the farm sediments.

The reported failure of antibacterial residues in sediment to alter microbial densities or biogeochemical processes, except under extraordinarily high dosing rates, should not be considered justification to eliminate this possibilty. These results are very likely a function of the particular drugs used and the marine environments examined. All three of the drugs that have been examined (oxytetracycline, oxolinic acid and flumequine) form complexes with divalent cations, principally calcium and magnesium (Lunestad and Goksøyr 1990). In the marine systems studied (Hansen *et al.* 1992, Capone *et al.* 1994, Herwig *et al.* 1997, Herwig and Grey 1997), approximately 95% of the drug residues were in the complexed form, which has no antimicrobial activity (Lunestad and Goksøyr 1990). Much more of the drug is likely to be in the free, bioavailable form in fresh waters. The calcium and magnesium content of sea water is 400 and 1,272 mg/L, respectively (Sverdrup *et al.* 1942). In fresh water, the calcium and magnesium concentrations are about 2-60 mg/L and 2-50 mg/L, respectively, depending upon water hardness (EPA 1989). With one to two orders-of-magnitude less complexing cations in freshwater systems, it is very likely that more of these drugs will be in the microbially-active free form, and effects on the microbial community more evident. No work has been published, however, on the environmental effects of these drugs in freshwater systems, nor is any information available on other antibacterials which do not form cation complexes.

If antibacterial residues were to reduce bacterial numbers and slow the rate at which they degrade organic matter, the effects are likely to be of limited spatial extent since, in net-cage culture at least, antibacterial residues are only elevated within tens of meters of farm sites (Coyne *et al.* 1994, Capone *et al.* 1996). There could, however, be adverse consequences to the farm aside from the potential stimulation of antibacterial resistance. Aerobic degradation of organic matter results in the production of non-toxic products such as carbon dioxide and nitrates, while anaerobic degradation yields toxic products such as sulfides and ammonium. If the presence of antibacterial residues reduced the extent of aerobic degradation of organic matter, more labile organic carbon would be incorporated into the anaerobic portion of the sediment column. Subsequent anaerobic degradation could result in an increased production of toxic end products. Such a scenario is entirely speculative, but illustrates the need for more information on antibacterial effects on sediment biogeochemistry.

CONCLUSIONS

There is remarkably little information available on the effects of aquaculture chemicals on non-target organisms. Emphasis has been on the efficacy of the chemical on the target species, and there has been little consideration of the environmental effects of any chemical residues remaining in wastewaters from the culture facility. There are rarely data available on the concentration of aquaculture chemicals in effluents. For most chemicals, there is little information on environmental persistence, obviously a key consideration in predicting ecological effects. Toxicity data are often lacking or minimal, and in any event, are difficult to relate to field conditions because of unknown dilution rates. Surveys are rarely conducted to document population or community-level effects due to release of aquaculture chemicals to receiving waters. Consequently, this evaluation is based on very limited data, and is biased towards temperate latitudes from which the majority of the data are available.

Ideally, information such as that listed in Table 4 is available to determine the likelihood that use of an aquaculture chemical will have unintended and adverse ecological effects. All the information listed may not be necessary for each compound; for example, rapid degradation would mitigate the need for chronic toxicity data. Nevertheless, for many aquaculture chemicals, we have very little of the data needs identified. In most developed countries, such information is now required before new aquaculture chemicals are approved for use. However, no comparable data are available for many chemicals already in widespread commercial use, and there is a particular lack of data for tropical environments where aquaculture chemical use is less strictly regulated.

Table 4. Primary data needed to assess ecological effects of aquaculture chemicals.

USE PATTERNS

- Dosage
- Frequency of use
- Wastewater management
- Solid waste management
- Dilution prior to release
- Rate of dilution in receiving waters

ENVIRONMENTAL FATE

- Absorption efficiency (if a feed supplement)
- Metabolism
- Chemical reactivity
- Microbial degradation rates
- Photodegradation rates
- Temperature/pH dependence of environmental fate
- Vapor pressure
- Aqueous solubility
- Degradation products (with same environmental fate and toxicology needs for all)

TOXICOLOGY

- Bioaccumulation potential (K_{ow}, BCF)
- Acute toxicity (multiple species, multiple life stages)
- Chronic toxicity (multiple species, multiple life stages)
- Stimulation of resistance (for antibacterials)
- Effects on sediment biogeochemistry

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