THE ASIATIC CLAM, *CORBICULA* SPP., AS A BIOLOGICAL MONITOR IN FRESHWATER ENVIRONMENTS

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Abstract. Asiatic clams, *Corbicula* spp., are filter-feeding freshwater bivalves that are widely distributed, abundant, and fast growing with a lifespan of 1-3 yrs. A review of the existing literature demonstrates that Asiatic clams can concentrate organic pollutants from both water and sediment and heavy metals from water. In conjunction with these traits, they exhibit a high tolerance for the effects resulting from exposure to toxic substances. While an organism must possess these traits to serve as an effective biological monitor, they have also permitted the Asiatic clam to rapidly colonize natural and industrial environments resulting in purported ecological disturbances and severe economic repurcussions, respectively. Its invasive biofouling attributes therefore restrict the use of Asiatic clams for biomonitoring purposes from *Corbicula*-free drainage systems.

Introduction

Biological monitoring uses living organisms as 'sensors' to detect changes in an effluent or water that may endanger aquatic life (Rand and Petrocelli, 1985). Organisms normally used as biological monitors are abundant, widespread, easy to collect, tolerant to collection, handling, and holding in the laboratory, possess multi-year life spans, exhibit non-migratory and sedentary life styles, and able to accumulate pollutants in their tissues without experiencing mortality (Phillips, 1977). One group of aquatic organisms that expresses these traits is filter feeding bivalves. These organisms are additionally suited to be biological monitors because they siphon large volumes of water on a daily basis providing an opportunity for concentrating a dissolved or suspended contaminant that may be present in low concentrations in the water column thereby facilitating detection (Simkiss *et al.*, 1982).

In the freshwater environment, bivalves have been used for the detection of a wide range of contaminants that include but may not be limited to heavy metals (Mathis and Cummings, 1973; Smith *et al.*, 1975; Renzoni and Bacci, 1976; Manley and George, 1977; Foster and Bates, 1978; Heit *et al.*, 1980; Cassini *et al.*, 1986; Hemelraad *et al.*, 1986a,b; Czarnezki, 1987), radionuclides (Nelson, 1962; Harrison and Quinn, 1972; Heit *et al.*, 1980), and pesticides (Bedford *et al.*, 1968; Fikes and Tubb, 1972; Leard *et al.*, 1980). Bivalve species used as biological monitors in the studies cited above and others include *Anodonta anatina, A. cygnea, A. grandis, A. nuttalliana, Amblema costata, A. plicata, Elliptio complanatus, E. crassidens* (Smith and Isom, 1967), *Fusconaia flava, Lampsilis anadontoides, L. clabornensis, L. radiata, L. siliquoidea, L. ventricosa, Lasmigona complanata, Megalonaias gigantea, Musculium transversum* (Zischke and Arthur, 1987), *Plectomerus dombeyanus, Quadrula quadrula, Unio elongatulus,* and *U. pictorum.*

While the objective of many of these studies was to analyze clam tissues for the presence

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of contaminants, exposure of an organism to a contaminant in its natural habitat may also be demonstrated through effects on biochemical events (e.g. enzyme activities and concentrations of marker proteins), physiological mechanisms (e.g. respiration and heartbeat rate), or behavior (e.g. individual reactions or intra- and interspecific interactions). Studies with marine bivalves have addressed the effects of contaminants on mixed function oxidases and cytochromes P-450 (Payne, 1984; Anderson, 1985; Livingstone and Farrar, 1985; Parke, 1985; Payne *et al.*, 1985), the presence of metal binding proteins in marine bivalves exposed to heavy metals (Hennig, 1986), and changes in the concentrations of free amino acids (Powell *et al.*, 1982) as indicators of exposure to contaminants.

In North America, the Asiatic clam *Corbicula* spp. appears to be a suitable candidate for biological monitoring in freshwater environments. It has migrated across the continent since its introduction to the Pacific northwest in the 1930s to its current distribution throughout 33 states of the United States (Counts, 1986). It is probably only limited in its northern migration by extremes in winter water temperatures (McMahon, 1982). Reproductively, individuals are hermaphroditic with the potential ability for self-fertilization (Kraemer and Lott, 1977; Kraemer *et al.*, 1986). Embryos are incubated within the inner demibranch (gill) and expelled through the exhalant siphon following development. Advanced pediveligers are the predominant stage released from gravid adults (Doherty *et al.*, 1987a). Growth rates of juveniles are high and individuals demonstrate multiyear life spans (Britton *et al.*, 1979; Mattice and Wright, 1986; McMahon and Williams, 1986). Taxonomically, the genus may be represented on this continent by two species, *C. fluminea* and a second unidentified species (McLeod, 1986). In addition to these geographic and life history traits, Asiatic clams are tolerant of exposure to a wide variety of aquatic contaminants (Doherty and Cherry, 1988).

While many taxa of freshwater bivalve are either endangered or not present in adequate numbers except in a few drainage systems, *Corbicula* are widespread, abundant, fast-growing with a lifespan of 1-3 yrs, and generally tolerant of exposure to toxic materials in less than optimal environments. The purpose of this review is to summarize the results of studies that assessed the sublethal responses of *Corbicula* exposed to contaminants or studies in which Asiatic clams were used as biological monitors to define the extent to which Asiatic clams can be used as biomonitors in freshwater environments.

Review of Methodologies and Results

Laboratory studies

Most studies conducted in the laboratory assessed the sublethal effects or bioconcentration potential of heavy metals and pesticides in Asiatic clams. Martin and Sparks (1971) examined the histopathologic effects of exposure of Asiatic clams to copper sulfate. They reported the presence of lesions in soft tissue of clams exposed to copper at concentrations (12-50 ppb) below that which produced changes in siphoning activity (250-500 ppb) (Table I). Harrison *et al.* (1984) exposed clams to copper chloride in 20-L plastic flow-through

TABLE I

Summary of studies assessing the bioconcentration of metals and trace elements from water and sediment by Asiatic clams and the sublethal effects of exposure of Asiatic clams to metals and trace elements

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Arsenic	IPS	F	NA	0.02 ppm (W), 6.24 ppm (SD)	TC	0.63 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.04 ppm (W), 5.0 ppm (SD), 0.01 ppm (W), 4.3 ppm (SD)	TC, SC TC, SC	4.6 ppm (T), 0.14 ppm (SH), 7.1 ppm (T), 0.18 ppm (SH)	Rodgers *et al.*, 1980
	IPS	F	NA	0.02-0.1 mg kg ^{1} DW-SD	TC	0.43-0.72 mg kg^{-1} WW (T) for samples collected from 5 different sites	USACE, 1981
	IPS	F	NA	2-20 μg g^{-1} (DW-SD)	TC	<0.1-0.36 μg g^{-1} DW	Elder and Mattraw, 1984
Beryllium	IPS	F	NA	2-10 μg g^{-1} (DW)	TC	BD	Elder and Mattraw, 1984
Cadmium	IPS	F	NA	0.13 ppm (W), 2.11 ppm (SD)	TC	1.3 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.45 ppm (W) 0.30 ppm (W)	TC, SC TC, SC	5.85 ppm (T), 0.87 ppm (SH) 6.60 ppm (T), 0.54 ppm (SH)	Rodgers *et al.*, 1980
	IPS	F	NA	<0.5-<1.30 mg kg^{-1} (DW-SD)	TC	0.42-0.57 mg /kg^{-1} WW (T) for samples collected from 5 different sites	USACE, 1981
	IPS	F	NA	ND	TC	0.01 to 0.10 μg g^{-1} WW in clams collected over a 5-month span in 1972	Caldwell and Buhler, 1983
	S	L	9 days	0.6, 4.8 ppm (W)	AEC	AEC was significantly reduced in clams exposed to cadmium after 4 and 9 days of treatment.	Geisy *et al.*, 1983
	S	L	9 days 16 days	0.5, 5.0 mg l^{-1} (W) 0.005, 0.05 mg l^{-1} (W)	AEC	AEC levels variable in cadmium-exposed clams relative to controls, AEC levels increased in cadmium-exposed clams during a cadmium-free recovery period.	Geisy *et al.*, 1983
	AS	F	28 days	0.023 and 0.055 mg l^{-1} (W)	TC	Calculated BCFs = 3,770 and 1,752, respectively (tested in presence of substrate)	Graney *et al.*, 1983

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	AS	L	14 days	0.05 mg l^{-1} (W)	TC	Accumulation of Cd in clams exposed to Cd at 9 °C was dependent on duration of exposure and significantly reduced in presence of organic substrate. Accumulation of Cd in clams exposed to Cd at 21 °C was reduced in presence of silt, clay, and organic matter, and elevated at pH 7.8 (vs. 5.0) and in the presence of food. Accumulation was enhanced in all tests conducted at 21 °C as opposed to 9 °C.	Graney *et al.*, 1984
	TP	F	9 weeks	ND	TC	Low variation among biweekly samples	Joy *et al.*, 1983
	IPS	F	NA	<0.2-3.0 $\mu g\ g^{-1}$ (DW-SD)	TC	<0.1-0.25 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
	AS	F	30 days	0.012, 0.025, 1.0 mg L^{-1} (W)	EA	Significant reduction in cellulase activity of clams exposed to ≥0.025 mg L^{-1} after 10 days and in all treatments after 20 days. EA in 0.012 and 0.025 mg L^{-1} Cd-exposed clams was greater than in controls by day 30.	Farris, 1986
	AS	F	30 days	0.016, 0.021, 0.026 mg L^{-1} (W)	EA	Significant reduction in cellulase activity of clams exposed to ≥0.016 mg L^{-1} after 10 days.	Farris *et al.*, 1988

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	AS	F	30 days	0.012, 0.025, 0.1 mg L^{-1} (W)	TC	Body burden levels of cadmium after 30 days were 366.1, 748.6 and 303.5 $\mu g\ g^{-1}$ DW, respectively.	Farris, 1986
	S	L	48 days	2.2 mg kg^{-1} (DW-SD)	TC	3.05 $\mu g\ g^{-1}$ DW; intermediate duration and sediment concentration results also reported.	Tatem, 1986
	S	L	24 hours	0.1, 0.2, 0.3, 0.4 mg l^{-1} (W)	VCB	Mean valve parting time was inversely related to cadmium concentration.	Doherty *et al.*, 1987b
	AS	L	30 days	0.1 mg l^{-1} (W)	HMBP	Significantly higher levels of HMBP in cadmium-exposed clams at all sample times after day 7 in one study, after day 15 in a second study.	Doherty *et al.*, 1988
Cesium	IPS	F	NA	<0.01 ppm (W), 1.80 ppm (SD)	TC, SC	1.80 ppm (T), 0.70 ppm (SH)	Rodgers *et al.*, 1980
				0.03 ppm (W), 0.70 ppm (SD)	TC, SC	3.50 ppm (T), 0.02 ppm (SH)	
Chromium	IPS	F	NA	1.15 ppm (W), 21.6 ppm (SD)	TC	5.25 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.37 ppm (W), 23.7 ppm (SD)	TC, SC	8.20 ppm (T), 0.49 ppm (SH)	Rodgers *et al.*, 1980
				0.26 ppm (W), 12.7 ppm (SD)	TC, SC	55 ppm (T), 0.73 ppm (SH)	
	IPS	F	NA	<1.2-19.3 mg kg^{-1} (DW-SD)	TC	0.46-4.60 mg kg^{-1} Ww (T) for samples collected from 5 different sites	USACE, 1981
	TP	F	9 weeks	ND	TC	Levels highly variable and occassionally fell below baseline levels	Joy *et al.*, 1983
	IPS	F	NA	30-100 $\mu g\ g^{-1}$ (DW-SD)	TC	<0.1-2.0 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
Cobalt	IPS	F	NA	10-30 $\mu g\ g^{-1}$ (DW-SD)	TC	BD	Elder and Mattraw, 1984

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Copper	NR	L	NR	12-50 ppb (W)	HP	Intracellular vacuolization of digestive diverticula, hemocytic infiltration and increased mucocyte production in gills, and fragmentation, necrosis, and tissue sloughing of mantle. Tissue recovery complete in clams exposed to 12 ppb then placed in copper-free water.	Martin and Sparks, 1971
	NR	L	NR	125-250 ppb (W)	HP	Gills first affected followed by digestive tubules and collecting ducts	Martin and Sparks, 1971
	NR	L	NR	>250 ppb (W)	HP	Digestive tubules first affected followed by gills and mantle epithelium	Martin and Sparks, 1971
	NR	L	>14 days	>250 ppb (W)	SB	Effects of starvation observed at 250 ppb, no clams observed siphoning at 500 ppb	Martin and Sparks, 1971
	IPS	F	NA	0.50 ppm (W), 69.9 ppm (SD)	TC	20 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.92 ppm (W), 150 ppm (SD)	TC, SC	6.32 ppm (T), 0.48 ppm (SH)	Rodgers *et al.*, 1980
				0.60 ppm (W), 160 ppm (SD)	TC, SC	12.1 ppm (T), 0.62 ppm (SW)	
	IPS	F	9 weeks	ND	TC	Low variation among biweekly samples	Joy *et al.*, 1983
	AS	F	28 days	0.016 and 0.057 mg l^{-1} (W)	TC	Calculated BCFs = 22,571 and 17.720, respectively (tested in presence of substrate)	Graney *et al.*, 1983
	IPS	F	NA	20-50 $\mu g\ g^{-1}$ DW	TC	1.0-13 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
	FT	L	14-34.3 days	230, 102, 56, 25, and 11 $\mu g\ l^{-1}$ (W)	TC	16.7, 45.5, 35.2, 31.2, and 10.9 $\mu g\ g^{-1}$ DW, respectively	Harrison *et al.*, 1984

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	AS	L	30 days	6-32 $\mu g\ g^{-1}$ (W)	SB	Significant reductions in percentage of clams observed siphoning with increasing exposure concentrations.	Sappington, 1987
	AS	L	30 days	6-32 $\mu g\ g^{-1}$ (W)	GLY, TWC, PRO	Significant reduction in GLY and significant increases in TWC and PRO for clams exposed to 32 $\mu g\ L^{-1}$.	Sappington, 1987
	AS	F	30 days (June)	16-27 $\mu g\ g^{-1}$ (W)	GLY, TWC, PRO	Significant reduction in GLY and protein and significant increase in TWC for clams exposed to $\geq$ 16 $\mu g\ L^{-1}$.	Sappington, 1987
	AS	F	30 days (August)	8-18 $\mu g\ L^{-1}$ (W)	GLY, TWC, PRO	Significant reduction in GLY, significant increase in PRO for clams exposed to $\geq$8 $\mu g\ L^{-1}$.	Sappington, 1987
	TP	F	30 days	<15-105 $\mu g\ L^{-1}$ (W)	GLY, TWC	No change in TWC. Significant reduction in GLY and significant increase in TWC after 10 days for clams at station closest to discharge. Same pattern of observations for clams at next farthest site from discharge after 30 days but not for clams closest to discharge.	Sappington, 1987
Iron	IPS	F	NA	40 ppm (W), 14,216 ppm (SD)	TC	430 ppm (WW)	Rodgers *et al.*, 1979
	TP	F	9 weeks	$\leq$0.1 mg l^{-1} (W)	TC	2- to 5-fold increases over baseline levels	Joy *et al.*, 1983
	IPS	F	NA	1.5-5% (DW-SD)	TC	BD	Elder and Mattraw, 1984

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Lead	IPS	F	NA	ND	SC	Negative correlation between weight and lead concentration for shells <0.2 g; consistent concentration of lead in shells weighing from 0.2-1.2 g.	Clarke *et al.*, 1979
	IPS	F	NA	<0.5 mg kg^{-1} (DW-SD)	TC	4.4-6.5 mg kg^{-1} WW (T) for samples collected from 5 different sites	USACE, 1981
	IPS	F	NA	10.2 mg kg^{-1} (DW-SD)	TC	10.3 mg kg^{-1} WW (T)	USACE, 1982
	IPS	F	9 weeks	ND	TC	Low variation among biweekly samples	Joy *et al.*, 1983
	IPS	F	NA	7-50 $\mu g\ g^{-1}$ (DW-SD)	TC	<0.1-1.0 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
	S	L	48 days	98.9 mg kg^{-1} (DW-SD)	TC	2.29 $\mu g\ g^{-1}$ DW; intermediate duration and sediment concentration results also reported	Tatem, 1986
Magnesium	IPS	F	9 weeks	4-6 mg l^{-1} (W)	TC	0- to 5-fold increases over baseline levels	Joy *et al.*, 1983
Manganese	IPS	F	NA	9.64 ppm (W) 6800 ppm (SD)	TC	300 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.06 ppm (W), 385 ppm (SD)	TC, SC	106 ppm (T), 28 ppm (SH)	Rodgers *et al.*, 1980
				0.07 ppm (W), 290 ppm (SD)	TC, SC	144 ppm (T), 33 ppm (SH)	
	TP	F	9 weeks	≤0.1 mg l^{-1} (W)	TC	2- to 10-fold increases over baseline levels	Joy *et al.*, 1983
	IPS	F	NA	700-2000 $\mu g\ g^{-1}$ (DW-SD)	TC	2.0-375 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
Mercury	IPS	F	NA	0.03 ppm (W), 0.46 ppm (SD)	TC	0.01 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.02 ppm (W), 0.53 ppm (SD)	TC, SC	1.17 ppm (T), 0.03 ppm (SH)	Rodgers *et al.*, 1980
				0.004 ppm (W), 0.40 ppm (SD)	TC, SC	0.30 ppm (T), 0.02 ppm (SH)	
	IPS	F	NA	0.06-0.12 mg kg^{-1} (DW-SD)	TC	0.08-0.13 mg kg^{-1} WW (T) for samples collected from 5 different sites	USACE, 1981
	IPS	F	NA	0.44 mg kg^{-1} (DW-SD)	TC	0.219 mg kg^{-1} WW (T)	USACE, 1982

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	IPS	F	NA	ND	TC	0.016 to 0.019 $\mu g\ g^{-1}$ WW in clams collected over a 5-month span in 1972	Caldwell and Buhler, 1983
	IPS	F	NA	BD	TC	<0.1-0.2 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
Molybdenum	IPS	F	NA	0.11 ppm (W), 0.60 ppm (SD)	TC, SC	16.3 ppm (T), 0.40 ppm (SH)	Rodgers *et al.*, 1980
				0.12 ppm (W), 0.36 ppm (SD)	TC, SC	2.1 ppm (T), 0.30 ppm (SH)	
Nickel	TP	F	9 weeks	ND	TC	Low variation among biweekly samples	Joy *et al.*, 1983
	IPS	F	NA	15-70 $\mu g\ g^{-1}$ (DW-SD)	TC	BD	Elder and Mattraw, 1984
Potassium	IPS	F	NA	6 ppm (W), 7789 ppm (SD)	TC	590 ppm (WW)	Rodgers *et al*, 1979
Selenium	IPS	F	NA	0.06 ppm (W), 1.35 ppm (SD)	TC	0.59 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	0.11 ppm (W), 0.88 ppm (SD)	TC, SC	3.9 ppm (T), 0.29 ppm (SH)	Rodgers *et al.*, 1980
				0.10 ppm (W), 0.60 ppm (SD)	TC, SC	16.5 ppm (T), 0.5 ppm (SH)	
	IPS	F	NA	ND	TC	0.10-0.23 mg kg^{-1} WW (T) for samples collected from 5 different sites	USACE, 1981
	IPS	F	NA	ND	TC	0.43 mg kg^{-1} WW (T)	USACE, 1982
Silver	TP	F	9 weeks	ND	TC	Low variation among biweekly samples	Joy *et al.*, 1983
Tin	IPS	F	NA	3-10 $\mu g\ g^{-1}$ (DW-SD)	TC	BD	Elder and Mattraw, 1984
Zinc	IPS	F	NA	3.47 ppm (W), 69.5 ppm (SD)	TC	39 ppm (WW)	Rodgers *et al.*, 1979
	IPS	F	NA	8.1 ppm (W), 63.3 ppm (SD)	TC, SC	500 ppm (T), 13.1 ppm (SH)	Rodgers *et al.*, 1980
				8.73 ppm (W), 46.7 ppm (SD)	TC, SC	564 ppm (T), 26 ppm (SH)	
	AS	F	28 days	0.218, 0.433, and 0.835 mg l^{-1} (W)	TC	Calculated BCFs = 631, 358, and 511, respectively (tested in presence of substrate)	Graney *et al.*, 1983

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	TP	F	9 weeks	0.1-0.5 mg l^{-1} (W)	TC	Tissue concentrations exhibited little variation from week to week and from baseline levels	Joy *et al.*, 1983
	IPS	F	NA	ND	TC	12 to 26 $\mu g\ g^{-1}$ WW in clams collected from the Columbia River over a 5-month span in 1972	Caldwel and Buhler, 1983
	IPS	F	ND	20-150 $\mu g\ g^{-1}$ (DW-SD)	TC	2.1-26 $\mu g\ g^{-1}$ DW	Elder and Mattraw, 1984
	AS	F	30 days	0.05, 1.0 mg L^{-1} (W)	EA	Significant reduction in cellulase activity in comparison with control clams for both treatments after 20 days and for 0.05 mg L^{-1} treatment after 10 days.	Farris, 1986
	AS	L	30 days	0.05, 1.0 mg L^{-1} (W)	EA	No significant differences among treatments after 30 days	Farris, 1986
	AS	F	30 days	0.05, 1.0 mg L^{-1} (W)	G	Significant reduction in growth of clams exposed to 1.0 mg L^{-1} by day 20 and in clams exposed to 0.05 mg L^{-1} by day 30.	Farris, 1986
	AS	F	30 days	0.05, 1.0 mg L^{-1} (W)	TC	Body burden levels of zinc increased from 180 to 433 $\mu g\ g^{-1}$ DW after 30 days of exposure to 0.05 mg L^{-1} and from 183-827 $\mu g\ g^{-1}$ DW after 30 days of exposure to 1.0 mg L^{-1}.	Farris, 1986

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	AS	L	30 days	0.10, 1.0 mg l^{-1} (W)	G	No significant differences in growth among control and Zn-exposed clams	Belanger *et al.*, 1986c
	AS	F	30 days	0.05, 0.50, 1.0 mg l^{-1} (W)	TWC	Significant increase in TWC for clams exposed to 1.0 mg l^{-1} in spring and summer study, significant increase in TWC for clams in all treatments in fall study	Belanger *et al.*, 1986c
	AS	F	30 days	0.025-1.0 mg l^{-1} (W)	TC	Total Zn uptake was seasonally influenced; total body burden of zinc in clams exposed to zinc treatments ranged from ≈500 to 1.900 μg Zn g^{-1} dry weight in a dose response fashion	Belanger *et al.*, 1986c
	AS	F	30 days	0.02-1.0 mg l^{-1} (W)	G	Spring study – weight gain impaired by day 10 and length increase by day 20 for adult and juvenile clams exposed to 0.05 mg l^{-1}. Summer study – weight gain reduced at 0.05 mg l^{-1}, weight gain and shell growth impaired at 1.0 mg l^{-1} by day 10. Fall study – both shell and weight growth inhibited at concentrations >0.09 mg l^{-1}.	Belanger *et al.*, 1986c
	IPS	F	ND	<10-20 μg g^{-1} (W) 5.7-26.6 mg kg^{-1} (DW-SD)	TC	10-14 mg kg^{-1} WW (T) for samples collected from 5 different sites	USACE, 1981

TABLE I *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	IPS	F	ND	110 $\mu g\ g^{-1}$ (W) 7.46 mg kg^{-1} (DW-SD)	TC	157 mg kg^{-1} WW (T)	USACE, 1982
	FT	L	10 days	162 $\mu g\ g^{-1}$ (DW-SD)	TC, WT, LIP	No significant difference between zinc contaminated sediment exposed clams and pre-exposure levels of zinc in clam tissues. No significant difference between treatments for weight or % lipid content.	Mac *et al.*, 1984
	S	L	24 hours	0.1, 0.3, 0.5, 0.9 mg L^{-1} (W)	VCB	Mean valve parting time was inversely related to zinc concentration.	Doherty *et al.*, 1987b

AEC = Adenylate energy charge, AS = artificial stream, BCF = bioconcentration factor, BD = below detection, BLD = bottom load detritus, DW = dry weight EA = enzyme activity, F = field, FT = flow-through, g = grams, G = growth, GLY = glycogen, HP = histopathology, IPS = indigenous population survey L = laboratory, LIP = lipid, NA = not applicable, ND = not determined, NR = not reported, ppm = parts per million, ppb = parts per billion, PRO = protein S = static, SB = siphoning behavior, SC = shell concentration, SD = sediment, SH = shell, T = tissue, TC = tissue concentration, TP = transplant TWC = tissue water content, VCB = valve closure behavior, W = water, WT = weight, WW = wet weight.

trays for 75 days to assess uptake of copper by Asiatic clams. They reported tissue concentrations of copper ranging from 10.9 to 45.5 $\mu g\ g^{-1}$ dry weight in clams exposed to concentrations of copper ranging from 11 to 230 $\mu g\ L^{-1}$ for 14 to 34.3 days (Table I).

Geisy *et al.* (1983) periodically sampled clams exposed to cadmium in polyethylene pans in static-renewal tests during 16- and 14-day exposure and recovery periods, respectively to assess the effects on adenylate concentrations. The adenylate energy charge (AEC) was calculated from the levels of adenosine mono-, di-, and triphosphate (AMP, ADP, and ATP, respectively) as determined by the luciferin-luciferase bioluminescence technique. They determined that exposure of Asiatic clams to 0.5 and 5.0 mg Cd L^{-1} resulted in significant reductions in the AEC in foot muscle as compared with control clams (Table I). They also reported a significant reduction in the AEC of clams starved for 30 days in comparison with control clams and clams starved for 10 and 20 days (Table III).

Graney *et al.* (1984) studied the influence of experimental variables on the uptake of cadmium by Asiatic clams in laboratory artificial streams. They found that the uptake of cadmium by Asiatic clams was elevated with increasing exposure duration, with a higher temperature, increased pH, and in the presence of food but reduced in the presence of substrate (Table I).

In support of these findings, Mac *et al.* (1984) reported that Asiatic clams did not accumulate zinc from contaminated sediment (Table I). Clams were exposed to sediment contaminated with zinc and polychlorinated biphenyls (PCBs) in a flow-through system at 20 °C for 10 days with a 2-day recovery period for gut clearance before analysis. Subsequently, Tatem (1986) reported that cadmium was not bioaccumulated from sediment by clams and there were no differences in the concentrations of lead in tissues of clams retrieved from contaminated sediments after 3 and 48 days (Table I). Clams were exposed to 10 L of clean sand or 10, 50, or 100% contaminated sediment in glass aquaria. Clams were also monitored for cadmium, lead, and PCBs during an 88-day depuration period.

Doherty *et al.* (1987b) discovered that mean valve parting time for Asiatic clams was inversely related to the exposure concentrations of cadmium and zinc in 24-h studies (Table I). They also reported that the rate of response was greater for exposure to cadmium. Durations of periods with valves parted or sealed were determined by monitoring the movements of the left hand valve of immobilized individuals.

In subsequent experiments, Doherty *et al.* (1988) reported that the levels of heavy metal binding protein (HMBP) in Asiatic clams was a function of the duration and mode of exposure to cadmium (Table I). They also reported that mode of exposure produced tissue specific differences in levels of HMBP. The maximum relative differences in HMBP levels between control and cadmium-exposed clams were ~2-fold. Clams were exposed to dissolved cadmium for 30 days in flow-through artificial streams or to cadmium-contaminated algae for 4 h day^{-1} for 15 days. Clams were exposed to cadmium-contaminated algae in static water trays for 4 h and kept in cadmium-free water in artificial streams for the remainder of each day for 15 days. HMBP levels in clam tissues were quantified by displacement of heavy metals from binding proteins by radioactive mercury.

TABLE II

Summary of studies primarily assessing the bioconcentration of pesticides from water and sediment by Asiatic clams

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Aldrin	TP	F	72 days	BD at day 60 (<62 ng l^{-1} (w)	TC	830 ng g^{-1} BCF = 13,390 (ng g fat/detection limit)	Hartley and Johnston, 1983
Arochlor 1254	IPS	F	NA	BD, ND, 0.001-0.003, BD, and 0.0016 μg g^{-1} (SD) from four stations sampled over a 1-year period	TC	0.138, 0.073, 0.004, 0.125, and 0.071 μg g^{-1} in clams collected over an overlapping 14-month period	Livingston *et al.*, 1978
	IPS	F	NA	77.9, <0.5 μg kg^{-1} (DW-SD)	TC	<25 μg kg^{-1} WW (T),	USACE, 1981
Arochlor 1260	IPS	F	NA	<1.0, 196 μg kg^{-1} (DW-SD)	TC	<25 μg kg^{-1} WW (T)	USACE, 1981
Bentazon	M	L	30 days	0.12 ppm (W)	TC	average concentration factor = 0.45	Booth *et al.*, 1973
Carbofuran (ring-^{14}C)	M	L	≃10 days	equivalent to 1 pound $acre^{-1}$ (W)	TC	parent compound not found in living clams; total metabolite concentration = 1.45 ppm (WW)	Yu *et al.*, 1974
Carbofuran (carbonyl-^{14}C)	M	L	≃10 days	equivalent to 1 pound $acre^{-1}$ (W)	TC	parent compound not found in living clams; total metabolite concentration = 0.164 and 0.077 in two separate tests	Yu *et al.*, 1974
Chlordane	IPS	F	NA	<1.0, <0.5 μg kg^{-1} (DW-SD)	TC	23.6, 40.0 μg kg^{-1} WW (T), respectively	USACE, 1981
α-Chlordane	TP	F	72 days	316 ng l^{-1} at day 60 (W)	TC	1520 ng g^{-1}; BCF = 4810	Hartley and Johnston, 1983
γ-Chlordane	TP	F	72 days	372 ng l^{-1} at day 60 (W)	TC	1610 ng g^{-1} at day 60; BCF = 4330	Hartley and Johnston, 1983

TABLE II *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Chlordane	IPS	F	NA	<1-3 $\mu g\ kg^{-1}$ (DW-SD) 1.7-10 $\mu g\ g^{-1}$ (DW-BLD)	TC	16-68 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
DDD	IPS	F	NA	ND	TC	93-490 $\mu g\ kg^{-1}$ in 18 samples from two sites collected over a 1-year period	Modin, 1969
	IPS	F	NA	<0.1-25 (SD) <0.1-12 (BLD)	TC	2.8-44 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
	IPS	F	NA	ND	TC	≤5-690 $\mu g\ kg^{-1}$ in 54 samples from two stations collected over a 5-year period	Butler, 1973
p,p'-DDE	IPS	F	NA	ND	TC	21-35 $\mu g\ kg^{-1}$ WW from three samples collected over an 8-month span	Claeys *et al.*, 1975
	IPS	F	NA	0.01 $\mu g\ l^{-1}$ (W)	TC	body burden levels of 0.25 and 0.10 $\mu g\ g^{-1}$ in consecutive years	Leard *et al.*, 1980
DDE	IPS	F	NA	<0.1-3.5 (SD) 1.5-4.0 (BLD)	TC	9.1-46 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
DDT	IPS	F	NA	ND	TC	≤5-1100 $\mu g\ kg^{-1}$ in 54 samples from two stations collected over a 5-year period	Butler, 1973
DDT-R	IPS	F	NA	BD, ND, 0.002-0.028, BD and 0.005 $\mu g\ g^{-1}$ (SD) from four stations sampled over a 1-year period	TC	0.105, 0.201, 0.056, 0.081, and 0.110 $\mu g\ g^{-1}$ in clams collected over an overlapping 14-month period	Livingston *et al.*, 1978
DDT	IPS	F	NA	<0.1-0.9 (SD) <0.1-7.8 (BLD)	TC	<0.1-16 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984

TABLE II *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
o,p'-DDT	IPS	F	NA	ND	TC	measurable levels ranged from 0.06-0.67 $\mu g\ g^{-1}$	Leard *et al.*, 1980
p,p'-DDT	IPS	F	NA	ND	TC	measurable levels ranged from trace quantities ($<0.01\ \mu g\ g^{-1}$) to 0.54 $\mu g\ g^{-1}$	Leard *et al.*, 1980
	IPS	F	NA	ND	TC	10-15 $\mu g\ kg^{-1}$ WW from three samples collected over an 8-month span	Claeys *et al.*, 1975
Dieldrin	IPS	F	NA	ND	TC	$\leq$5-24 $\mu g\ kg^{-1}$ in 44 samples from two stations collected over a 5-year period	Butler, 1973
	M	L	34 days	2 ppb (W)	TC	average concentration factor = 1015	Sanborn and Yu, 1973
	TP	F	72 days	891 ng l^{-1} at day 60 (W)	TC	3150 ng g^{-1}; BCF = 3450	Hartley and Johnston, 1983
	IPS	F	NA	<0.1-0.1 (SD) 0.3-1.7 (BLD)	TC	0.5-2.9 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
Endosulfan	IPS	F	NA	BD-(SD) BD-(BLD)	TC	<0.1-0.2 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
Endosulfan sulfate	IPS	F	NA	NR	TC	10 $\mu g\ kg^{-1}$ WW (T)	USACE, 1981
Endrin	IPS	F	NA	ND	TC	$\leq$5-18 $\mu g\ kg^{-1}$ in 22 samples from two stations collected over a 5-year period	Butler, 1973
	IPS	F	NA	BD-(SD) BD-(BLD)	TC	<0.1-0.4 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
Heptachlor	TP	F	72 days	BD at day 60 (<49 ng l^{-1}) (W)	TC	510 ng g^{-1}; BCF = 10,630 (ng g fat/detection limit)	Hartley and Johnston, 1983

TABLE II *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Heptachlor epoxide	TP	F	72 days	880 ng l^{-1} at day 60 (W)	TC	2050 ng g^{-1}; BCF = 2330	Hartley and Johnston, 1983
	IPS	F	NA	BD (SD), <0.1-0.66 (BLD)	TC	<0.1-0.6 μg kg^{-1} DW	Elder and Mattraw, 1984
Hexachlorocyclo-hexane (BHC)	TP	F	72 days	590 ng l^{-1} at day 18 (W)	TC	1320 ng g^{-1}; BCF = 2240	Hartley and Johnston, 1983
Lindane	TP	F	72 days	296 ng l^{-1} at day 60 (W)	TC	773 ng g^{-1}; BCF = 2610	Hartley and Johnston, 1983
Methyl parathion	IPS	F	NA	0.08-0.46 μg l^{-1} (W)	TC	NR	Leard *et al.*, 1980
PCB	IPS	F	NA	ND	TC	≤5 μg kg^{-1} in a single sample	Butler, 1973
	IPS	F	NA	ND	TC	390-1170 μg kg^{-1} WW from four samples collected over an 8-month span	Claeys *et al.*, 1975
	IPS	F	NA	<1-36 (SD) 1.0-7.0 (BLD)	TC	10-44 μg kg^{-1} DW	Elder and Mattraw, 1984
	IPS	F	NA	<0.5, 64 mg kg^{-1} (DW-SD)	TC	<25 mg kg^{-1} WW (T)	USACE, 1982
	FT	L	10 days	19.6 μg g^{-1} (DW-SD)	TC, WT, LIP	significant increase in tissue level of PCBs in clams exposed to PCB-contaminated sediment BAF = 0.2; no significant difference between treatments in weight or % lipid content of clams.	Mac *et al.*, 1984
PCB (Aroclor 1242, 1254)	S	L	48 days	1.7 mg kg^{-1} (DW-SD) 0.6 mg kg^{-1} (DW-SD)	TC	2.2 μg g^{-1} tissue WW (controls = 0.38 μg g^{-1} PCB), intermediate duration and sediment concentration results also reported, BAF = 0.54-12.52	Tatem, 1986

TABLE II *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Strobane	IPS	F	NA	BD-(SD) BD-(BLD)	TC	<1-58 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984
TDE	IPS	F	NA	ND	TC	≤5-590 $\mu g\ kg^{-1}$ in 54 samples from two stations collected over a 5-year period	Butler, 1973
p,p'-TDE	IPS	F	NA	NR	TC	17-28 $\mu g\ kg^{-1}$ WW from three samples collected over an 8-month span	Claeys *et al.*, 1975
	IPS	F	NA	NR	TC	measurable levels ranged from trace quantities (<0.01 $\mu g\ g^{-1}$ to 0.49 $\mu g\ g^{-1}$	Leard *et al.*, 1980
Toxaphene	IPS	F	NA	0.46 $\mu g\ l^{-1}$ (W)	TC	body burden levels of 2.87 and 1.39 $\mu g\ g^{-1}$ in consecutive years	Leard *et al.*, 1980
				2.11 $\mu g\ l^{-1}$ (W)	TC	body burden levels of 7.68 and 3.62 $\mu g\ g^{-1}$ in consecutive years	
				1.42 $\mu g\ l^{-1}$ (W)	TC	body burden levels of 4.43 and 3.78 $\mu g\ g^{-1}$ in consecutive years	
	IPS	F	NA	BD-(SD) BD-(BLD)	TC	<10-300 $\mu g\ kg^{-1}$ DW	Elder and Mattraw, 1984

BCF = Bioconcentration factor, BD = below detection, BLD = bottom load detritus, DW = dry weight, DDT-R = DDT residues, F = field, G = grams, IPS = indigenous population survey, L = laboratory, LIP = lipid, M = microcosm, NA = not applicable, ND = not determined, S = static, SD = sediment, T = tissue, TC = tissue concentration, TP = transplant, W = water, WT = weight, WW = wet weight.

Mac *et al.* (1984) reported a minor (BAF=0.2) but significant increase in the tissue level of PCBs in clams held in contaminated sediment for 10 days (Table II). Tatem (1986) reported measurable differences in tissue levels of PCBs between control and treated clams after 3 days of exposure to PCB-contaminated sediment. The tissue concentrations of PCBs continued to increase through day 48 of the study (Table II). Depuration of PCBs from clam tissue was very slow. Measurable levels of PCBs in treated clams remained above that found in control clams 88 days after transfer of treated clams to clean water.

Anderson *et al.* (1978) examined the effects of a series of natural environmental parameters and ammonia on the gill ciliary beating rate of Asiatic clams. Ciliary beating rate was measured by synchronizing the rate of flashing of a stroboscopic light with the rate of beating of the lateral cilia from an excised gill. The stroboscopic light served as the substage light source for coupled dissecting microscopes under which the gills were observed. They reported that the ciliary beating rate of lateral cilia of excised clam gills was completely inhibited at 0.09 to 0.10 mg L^{-1} of ammonia (Table III).

Belanger *et al.* (1986a,b) exposed juvenile and adult clams to chrysotile asbestos for 30 days in circular 15-L polycarbonate containers in static tests. They reported reductions in siphoning behavior, growth, and release of larvae during exposure to chrysotile asbestos (Table III). They also reported significant increases in tissue water content and fluid-filled spaces of gill tissue while demonstrating that asbestos fibers were deposited in soft tissues of clams exposed to asbestos.

Graney and Geisey (1988) exposed Asiatic clams to sodium dodecyl sulfate in static systems for 96 h and in flow-through tests for 60 days at 20 °C. Amino acid concentrations in adductor muscle and mantle tissue of clams were periodically measured as was respiration, tissue water content, and condition. They reported significant increases in the free amino acid concentration of mantle and adductor muscle tissue of clams exposed to sodium dodecyl sulfate for 60 days (Table III). Oxygen consumption and condition were reduced among certain groups of clams while tissue water content increased in some clams.

In a series of studies, Asiatic clams were included as a biological component of model laboratory ecosystems in which the fate of three potential aquatic contaminants was assessed. In each of the studies, investigators monitored radiolabeled samples of each of the compounds in the water and organisms of the model ecosystems for 30 to 34 days. Investigators calculated concentration factors (CFs), the ratio of chemical in organism to that in water, to assess the relative biomonitoring potential among organisms within the systems.

Sanborn and Yu (1973) reported a CF of 1015 for dieldrin in Asiatic clams (Table II) but higher CF values (~115 000, 7500, 6150, 2150, and 1300) for other components of a model ecosystem (snails, algae, fish, *Daphnia,* and *Elodea,* respectively). Booth *et al.* (1973) reported that Bentazon (3-isopropyl-1H-2,1,3-benzothiadiazin-4 (3H)-1-2,2-dioxide) was not taken up by Asiatic clams (Table II) but was accumulated to a certain extent by other organisms in a model ecosystem (CF=2, 5, 8, 9, 10, and 52 in *Elodea,* snail, mosquito, *Daphnia,* algae, and crab, respectively). Yu *et al.* (1974) reported that Carbofuran (2,2-dimethyl-2,3-dihydrobenzofuranyl 7-N-methylcarbamate) was highly toxic to all of the

TABLE III

Summary of studies primarily assessing the sublethal physiological and biochemical responses of Asiatic clams subjected to treatments with a broad group of environmental contaminants and stressors

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Ammonia	FT	L	10 hours	0.01-0.10 mg L^{-1} (W)	GCBR	50, 90, and 100% reduction in GCBR at 0.05, 0.06, and 0.09 mg L^{-1}, respectively.	Anderson *et al.*, 1978
	AS	L	30 days	0.29 mg L^{-1} (W)	GLY, TWC	Significant reduction in GLY and TWC	Sappington, 1987
	AS	L	4 days	0.54 mg L^{-1} (W)	GLY, TWC	Significant reduction in GLY, no change in TWC	Sappington, 1987
	AS	L	30 days	0.29, 0.54 mg L^{-1} (W)	SB	Significant reduction in SB	Sappington, 1987
Asbestos	S, C	L	96 hours	0-10^8 f l^{-1} (W)	SB	reduction in siphoning in absence of food, no reduction in siphoning due to asbestos in presence of food (studies conducted with clams 12.5-17.0 mm in length)	Belanger *et al.*, 1986a, 1987
	S, C	L	96 hours	0-10^8 f l^{-1} (W)	TC	no uptake observed in absence of food, uptake observed in fed clams exposed to 10^8 f l^{-1} (studies conducted with clams 12.5-17.0 mm in length)	Belanger *et al.*, 1986a, 1987
	S, C	L	30 days	0-10^8 f l^{-1} (W)	SB	reduction in siphoning among all asbestos-treated groups (studies conducted with clams 12.5-17.0 mm in length)	Belanger *et al.*, 1986a, 1987

TABLE III *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	S, C	L	30 days	0-10^8 f l^{-1} (W)	TC	asbestos fibers observed in tissues of clams exposed to 10^8 f l^{-1}, BCF for whole clams = 1.9 (studies conducted with clames 12.5-17.0 mm in length)	Belanger *et al.*, 1986a, 1987
	S, C	L	30 days	0-10^8 f/l (W)	G	growth reduced in clams exposed to asbestos at concentrations of $\geq 10^4$ f l^{-1} (studies conducted with clams 12.5-17.0 mm in length)	Belanger *et al.*, 1986a, 1987
	S, C	L	30 days	0-10^8 f l^{-1} (W)	TWC	percent tissue water was significantly elevated in asbestos-exposed clams	Belanger *et al.*, 1986a, 1987
	S, C	L	30 days	0-10^8 f l^{-1} (W)	HP	fluid-filled spaces significantly increased and size for clams exposed to 10^8 f l^{-1}	Belanger *et al.*, 1986a, 1987
	S, C	L	30 days	0-10^8 f l^{-1} (W)	LRP	dose-response relationship observed in numbers of larvae released and asbestos exposure concentration	Belanger *et al.*, 1986a, 1987
	S, C	L	30 days	0-10^8 f l^{-1} (W)	SB	winter clams – significant reductions in siphoning in all asbestos treatments; summer clams – significant reductions in siphoning among clams exposed to $\geq 10^4$ f l^{-1} (study conducted with clams 5.2-8.6 mm in shell length)	Belanger *et al.*, 1986b

TABLE III *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	S, C	L	30 days	0-10^8 f l^{-1} (W)	G	winter clams – weight gain significantly altered at 10^5 f l^{-1}; no differences among groups for wet or dry weights at 30 days; summer clams – shell and tissue growth significantly reduced at 10^4 f l^{-1}; no differences among groups for wet or dry weight at 30 days	Belanger *et al.*, 1986b
	S, C	L	30 days	0-10^8 f l^{-1} (W)	TWC	winter clams – significantly higher levels of water present in clams exposed to 10^5-10^8 f l^{-1}; summer clams – significantly higher levels of water present in clams exposed to 10^5-10^8 f l^{-1}	Belanger *et al.*, 1986b
	S, C	L	30 days	0-10^8 f l^{-1} (W)	TC	detectable quantities of asbestos fibers found in summer and winter clams exposed to 10^8 f l^{-1} only	Belanger *et al.*, 1986b
	IPS	F	NA	10^8 f l^{-1} (W)	TC	BCF values for whole clams ranged from 1442-5222	Belanger *et al.*, 1987
Chlorine	AS	L	30 days	0.30 mg l^{-1} TRC (W)	GLY	Significant reductions in GLY among clams exposed to chlorine at ambient temperature (23 °C) after 25 days.	Sappington, 1987
	TP	F	32 days	0.3-0.4 mg L^{-1} TRC (W)	GLY	No significant differences in GLY among sites.	Sappington, 1987

TABLE III *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
	AS	L	30 days	0.30 mg L^{-1} TRC (W)	SB	Significant reduction in the % if clams observed siphoning following 30 days of exposure to 0.3 mg L^{-1} TRC at temperatures ranging from 7-33 °C.	Sappington, 1987
Industrial-sewage effluent	TP	F	6 months	ND	AEC, GLY, LIP	AEC and glycogen significantly reduced while lipid levels were increased in clams exposed to effluent	Cantelmo-Cristini *et al.*, 1985
	TP	F	5 months	ND	SG	reduction in absolute shell growth and growth rate in comparison with clams at control sites	Fritz and Lutz, 1986
Monochcloramine	AS	L	7 days	0.73 mg L^{-1} (W)	GLY, TWC	Significant reduction in GLY at ≥4 days. No change in TWC after 7 days.	Sappington, 1987
	AS	L	30 days	0.73 mg L^{-1} (W)	SB	There were no clams siphoning in monochloramine after 30 days, whereas there were 64±11% siphoning in control treatment.	Sappington, 1987
Power plant effluent	TP	F	14 days	ND	EA	Clams at site upstream from plant discharge had highest cellulase activity. Clams at farthest site downstream had next highest level of cellulase activity, lowest levels of EA were found in clams immediately downstream from plant discharge.	Farris, 1986

TABLE III *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Sodium dococyl sulfate	S	L	4 days	12.5, 25.0 mg L^{-1} W)	FAAC	Significant increase in FAAC in mantle tissue as nmol mg $protein^{-1}$ and in adductor muscle as nmol mg WW^{-1} for clams exposed to 25.0 mg L^{-1}.	Graney and Geisy, 1988
	FT	L	60 days	0.65-23.6 mg L^{-1} (W)	FAAC	Significant increase in total FAAC in adductor muscle after 30 days for clams exposed to ≥3.0 mg L^{-1}. Significant increase in total FAAC in mantle tissue after 15 days for clams exposed to 3.0 mg L^{-1} and after 30 days for clams exposed to ≥0.65 mg L^{-1}.	Graney and Geisy, 1988
	FT	L	60 days	0.65-7.8 mg L^{-1} (W)	OC	Significant reduction in OC between days 5 and 30 for clams exposed to ≥3.0 mg L^{-1}. No significant differences in OC after 60 days of treatment for clams exposed to ≤3.0 mg L^{-1}.	Graney and Geisy, 1988
	FT	L	60 days	0.65-7.8 mg L^{-1} (W)	CN	Significant reduction in CN for clams exposed to 3.0 mg L^{-1} for 60 days.	Graney and Geisy, 1988
	FT	L	60 days	0.65-7.8 mg L^{-1} (W)	TWC	Significant increase in TWC after 30 days of treatment for clams exposed to 3.0 mg L^{-1}.	Graney and Geisy, 1988

TABLE III *(cont.)*

Stressor/ Contaminant	Test type	Study location	Duration	Exposure concentration	Parameter monitored	Observation	Citation
Starvation	S	L	30 days	NA	AEC	significantly reduced in clams starved 30 days in comparison with control clams and clams starved 10 and 20 days.	Geisy *et al.*, 1983
Thermal effluent	TP	F	6 months	ΔTemp = 7-13 °C between worst case sites and reference sites; absolute maximum temp = 35 °C	G	100% mortality among clams at station closest to discharge point, growth reduced at next closest and furthest sample site (stations 2 and 5) as compared with intermediate sites (stations 3 and 4)	Foe and Knight, 1987
	TP	F	6 months	ΔTemp = 7-13 °C; maximum Temp = 35 °C	CN	lowest index values obtained for clams nearest discharge point (station 2) followed by clams furthest from discharge point (station 5)	Foe and Knight, 1987
Thermal effluent	TP	F	6 months	ΔTemp = 7-13 °C maximum Temp = 35 °C	SFG	Greatest at station 4 and decreased with proximity to effluent discharge point; site furthest from discharge point generated values intermediate to 2 stations closes to discharge point	Foe and Knight, 1987

AEC = Adenylate energy charge, BCF = bioconcentration factor, C = circulated, CN = condition, DW = dry weight, EA = enzyme activity, F = field, $f l^{-1}$ = fibers liter^{-1}, FAAC = free amino acid concentration, FT = flow through, g = grams, GCBR = gill ciliary beating rate, G = growth, GLY = glycogen, HP = histopathology, IPS = indigenous population survey, L = laboratory, LIP = lipid, LRP = larval release pattern, NA = not applicable, ND = not determined, OC = oxygen consumption, S = static, SB = siphoning behavior, SC = shell concentration, SD = sediment, SFG = scope for growth, SH = shell, T = tissue, Temp = temperature, TC = tissue concentration, TP = transplant, TRC = total residual chlorine, TWC = tissue water content, WW = wet weight.

organisms in the experimental microcosm immediately after application to the system. Carbofuran was not found in any of the organisms used to restock the system 20 days after the initial application of the pesticide. The data suggested that Carbofuran was hydrolyzed to CO_2 and other metabolites prior to restocking efforts (Table II).

FIELD STUDIES

An extensive series of studies were reported in which indigenous populations of Asiatic clams were sampled to assess body burden levels of metals and pesticides. Rodgers *et al.* (1979) reported that the concentrations of ten elements (arsenic, cadmium, chromium, copper, iron, manganese, mercury, potassium, selenium, and zinc) in sediment were orders of magnitude greater than in water of the New River, Virginia. Concentrations of these same elements were always greater in clam tissues than in water except for mercury but less than that found in sediment (Table I). Clarke *et al.* (1979) analyzed empty shells of Asiatic clams collected from rivers and reservoirs in the vicinity of Nashville, Tennessee for the presence of lead. They discovered that shell size of Asiatic clams was related to the concentration of lead in empty shells (Table I).

The U.S. Army Corps of Engineers (US ACE, 1981, 1982) conducted an in-depth survey of meteorological, hydrological, chemical, and biological conditions at 19 sites in and around Lake Seminole and the Chatahoochee, Flint, and Apalachicola Rivers in Florida. The surveys revealed the presence of seven metals (arsenic, cadmium, chromium, lead, mercury, selenium, and zinc) in tissues of Asiatic clams from five stations (Table I). The concentrations of arsenic, cadmium, and lead were greater in clam tissue than sediment. The concentrations of mercury, chromium, and zinc in clam tissue and sediment differed by less than 2-fold for most samples.

Caldwell and Buhler (1983) analyzed tissues from Asiatic clams residing in the Columbia River, Oregon for mercury, cadmium, and zinc on three separate occassions. They noted that Asiatic clams tended to have higher zinc levels in their tissues than some of the marine bivalves collected from bays and estuaries along the Oregon coast. Tissue levels of cadmium and mercury were low (Table I).

Elder and Mattraw (1984) monitored sediments and Asiatic clams from the Apalachicola River, Florida for metals, trace elements, pesticides, and PCBs at five different sites on three separate occasions. They did not monitor water and suspended materials for contaminants because they assumed that significant accumulations of trace substances would occur in the bottom materials alone. They reported on eight metals in tissues of Asiatic clams (Table I). Concentrations of lead and copper in sediment were ~100- and 3-to 4-fold higher than in clam tissue, respectively. The concentrations of five other metals (arsenic, cadmium, chromium, manganese, and zinc) were also higher in sediment than in clam tissue.

Modin (1969) summarized the results of a study assessing the levels of pesticides in molluscs collected from the Sacramento-San Joaquin River Delta entering San Francisco Bay in 1967 that included the Asiatic clam. He reported that DDD, DDE, DDT, Dieldrin, and Endrin were routinely found in tissues of Asiatic clams from rivers feeding San Francisco Bay (Table II). He also reported that one sample of clam tissue contained ten

times more DDT (1.1. ppm) than any sample collected in San Francisco Bay. Butler (1973) included the data presented by Modin (1969) in a summary of the levels of pesticides in Asiatic clams for the same sites from 1968 to 1972. He reported that quantifiable concentrations of DDE, TDE, DDT, Dieldrin, and Endrin residues were routinely found in Asiatic clams during surveys of two sites over a period of six years (Table II).

Claeys *et al.* (1975) reported that Asiatic clams contained much higher levels of DDT and PCBs (Table II) than individuals of *Anodonta* collected together from the Columbia River, Oregon. Levels of DDT residues in *Anodonta* were seasonally influenced and ranged from 2 to 14.9 ppb. Levels in *Corbicula* ranged from 53 to 78 ppb with no marked seasonal variation either in DDT or the proportion of metabolites. *Corbicula* samples also had consistently higher levels of PCBs than *Anodonta* samples.

Livingston *et al.* (1978) conducted a four-year study of trends in organochlorine residues in sediments and organisms from bodies of water in and around Apalachicola Bay, Florida from 1972 to 1976. They reported lower concentrations of Aroclor 1254 and DDT and its metabolites in sediments than in clams inhabiting those sediments (Table II).

Leard *et al.* (1980) collected seven species of freshwater bivalve including Asiatic clams in 1972 and 1973 from major stream systems in Mississippi. Clam tissues were analyzed for chlordane, endrin, toxaphene, p,p'-DDE', p,p'-TDE, p,p'-DDT, and o,p'-DDT and found to contain detectable levels of toxaphene and DDT and its metabolites (Table II). Surveys by the U.S. Army Corps of Engineers (US ACE, 1981, 1982) revealed the presence of chlordane and endosulfan sulfate in tissues of Asiatic clams from two sites in and around Lake Seminole, Florida (Table II). Concentrations of 16 other chlorinated hydrocarbons in clam tissue ranged from <0.1 to <25 $\mu g\ kg^{-1}$ tissue wet weight. The concentration of chlordane and most other pesticides in sediment were <1.0 $\mu g\ kg^{-1}$ except for Aroclor 1254 and 1260 at two sites (77.9 and 196 $\mu g\ kg^{-1}$ dry weight, respectively).

Elder and Mattraw (1984) reported the presence of ten pesticides and PCBs in tissues of Asiatic clams from the Apalachicola River, Florida. They noted an increasing trend in the accumulation of DDT, dieldrin, chlordane, and PCB residues from sediments to bottom-load detritus to biological tissue (Table II). The most consistent trend was for chlordane. The concentration of chlordane was below detection in most sediment samples, at concentrations ten times greater in bottom-load detritus, and 50 times greater in clam tissue.

Several investigators conducted uptake studies in which Asiatic clams were transplanted from uncontaminated to contaminated sites to document the presence of contaminants in the environment. Joy *et al.* (1983) transplanted clams to four sites on the Kanawha River, West Virginia and analyzed clam tissues for ten heavy metals over a nine week period. Clams were held in cages floated by a styrofoam collar at a water depth of ~15 cm. Each cage contained a sand and gravel substrate ~3 cm deep. They reported little variation in the levels of zinc, copper, cadmium, lead, nickel, and silver from week to week for nine weeks and little change from baseline levels (Table I). The greatest increases above baseline levels were observed for iron, manganese, and magnesium but iron and manganese levels fluctuated widely from one period to another. They also found

chromium levels to be highly variable.

Graney *et al.* (1983) constructed flow-through artificial streams adjacent to the New River, Virginia and exposed clams in those streams to cadmium, copper, and zinc for 28 to 30 days. Water and tissue concentrations of metals were used to calculate bioconcentration factors (BCFs). They reported BCFs of ~22 600 and 17 700 for copper, 3770 and 1750 for cadmium, and 631, 358, and 511 for exposure to zinc (Table I). Lower BCFs for zinc may have been influenced by lower test temperatures (6 °C lower than cadmium and 10 °C lower than copper).

Hartley and Johnston (1983) transplanted Asiatic clams from Lake Sangchris, Illinois to the Kaskaskia River near Tuscola, Illinois. Clams were placed in sand in aluminum baking pans held in cages of rolled aluminum anchored to the bottom of the river for 72 days. They reported BCFs for eight pesticides in Asiatic clams ranging from 2330 for heptachlor epoxide to 13 390 for aldrin (Table II). They also reported increases in shell dimensions after the first 18 days of the study with no changes observed for the remainder of the 72-day study. Mean weight increased through the first 36 days of the study and did not change thereafter. Fat per gram of soft tissue increased from 25.3 to 28.4 mg during the first 9 days, reached 29.5 mg after 36 days of the study then fell to 26.4 mg after day 36. The loss of fat did not correspond to a loss in weight but did occur during a period of severe low water temperatures (0 to 2 °C). The investigators did not comment on whether the changes in fat per gram of soft tissue between observations were statistically significant. The investigators did note an association between the reductions in fat and the concentrations of all the pesticides except lindane and heptachlor epoxide.

Cantelmo-Cristini *et al.* (1985) transplanted Asiatic clams to sites above and below the combined effluents of a chemical plant and a municipal sewage treatment plant on the Raritan River, New Jersey for six months. Clams were held in wood framed cages with the top, bottom, and portions of two opposing sides composed of 0.16-mm mesh cloth. Foot tissues from clams were analyzed for adenylates with a semi-micro spectrophotometric assay while whole clams were assayed for lipid and glycogen. They reported that the AEC for Asiatic clams downstream from the chemical and sewage treatment plants was significantly lower than that of clams from control sites upstream (Table III). Similar results were observed by these investigators in a preliminary study conducted in 1981 and 1982. Cantelmo-Cristini *et al.* (1985) also reported reductions in the glycogen content of clams at the downstream site nearest the outfalls. Lipid concentrations in these clams were elevated in comparison with clams from all other sites. There were no differences in glycogen or lipid among clams from all other sites.

Fritz and Lutz (1986) assessed the effects of the same combined effluents from plants on the Raritan River on growth of Asiatic clams over five months. Growth was analyzed by counting shell microgrowth increments in specific regions of the shell valve under a compound microscope at a magnification of 100x. They demonstrated that changes in growth of Asiatic clams were indicative of exposure to aquatic contaminants. They reported that microgrowth increments in the outer fine crossed-lamellar shell layer of Asiatic clams were formed at the rate of ~1 per day and that clams transplanted to sites receiving a combined chemical and sewage effluent grew slower and deposited fewer

microgrowth increments than clams in cages at control stations (Table III).

Foe and Knight (1986) assessed the effects of exposure of Asiatic clams to a zinc- and copper-contaminated effluent by monitoring reproduction, shell growth, condition, and tissue concentrations of the metals among wild clams and clams transplanted in two different fashions. One method for transplanting clams consisted of placing clams in cage units constructed of plastic fluorescent light egg-crate-type panelling covered with polyethylene screen. Each cage unit was held ~0.5 m above the bottom in an upright position by metal rebar stakes driven into the substrate. The alternate transplant method consisted of tethering clams by short lenghts of 5 lb monofilament fishing line to a heavier nylon line that was staked taut along the bottom. Clams were transplanted at several locations around the outfall of a proposed waste treatment plant in the Sacramento-San Joaquin Delta, California and monitored for ten months.

Subsequently, Foe and Knight (1987) used the caging transplant method in a six-month study assessing the impact of a thermal discharge from a power plant on Asiatic clams. They reported that clams transplanted to a series of sites at increasing distances from the source of a variably contaminated effluent revealed a trend of increasing stress for clams closest to the source. Growth, condition, and scope for growth were all lower for clams at the station nearest the discharge for which survival of clams was high (Table III). High levels of mortality plagued clams transplanted to the site nearest the point of discharge.

LABORATORY/FIELD COMPARATIVE STUDIES

Rodgers *et al.* (1980) exposed Asiatic clams to solutions of copper and zinc in flow-through artificial streams in the laboratory and assessed the sublethal effects of exposure on gaping, filtering activity, and concentrations of copper and zinc in soft tissues of clams. The influence of the presence of substrate was also assessed. They also collected Asiatic clams from sites in the New River, Virginia above and below the heated discharge of a power generating plant at quarterly sampling intervals for one year and determined tissue concentrations for a series of elements.

Rodgers *et al.* (1980) reported that the concentrations of ten elements (arsenic, cadmium, cesium, chromium, copper, manganese, mercury, molybdenum, selenium, and zinc) were usually lowest in river water, intermediate in clam valves, and highest in either clam viscera or sediment in samples and clams (Table I). The concentrations of three elements (molybdenum, selenium, and zinc) were consistently greater in clam tissue than in sediment. Concentrations of two elements (copper and manganese) were consistently greater in sediment than in clam tissue. Concentrations of zinc and copper in tissues of clams exposed to these metals in laboratory bioassays greatly exceeded the concentrations in field clams.

Farris (1986) assessed the effects of exposure of Asiatic clams to zinc and cadmium in field-located and laboratory artificial streams on body burden levels of these two metals, growth, and enzyme activity (Table I). Tests were conducted for 30-day periods in different seasons over several years. He reported that cellulase activity was significantly reduced in clams exposed to $\geq$0.012 mg Cd L^{-1} after 20 days but significantly elevated in clams exposed to $\leq$0.025 mg Cd L^{-1} after 30 days compared to control clams (Table I).

Cellulase activity in clams exposed to zinc varied in relation to exposure concentration and duration (Table I).

Subsequently, Farris *et al.* (1988) assessed the sensitivity of cellulase activity in Asiatic clams exposed to copper in field-located artifical streams and in clams transplanted to sites up- and downstream from the point of discharge of an effluent from a power plant on the Clinch River, Virgina. Transplanted clams were placed in cobble containing nylon mesh cages and analyzed for enzyme activity after 14 days in one study and after 10 and 30 days in a second study conducted one year later. They reported that cellulolytic activity was significantly reduced in clams exposed to ≥ 16 μg Cu L^{-1} after 10 days of treatment (Table I). Transplanted clams retrieved from downstream sites nearest the point of discharge for a power plant effluent demonstrated significantly reduced levels of cellulase activity after 14 days in one study and after 30 days in another (Table III).

Belanger *et al.* (1986c) assessed the effects of exposure of Asiatic clams to zinc in field-located and laboratory artificial streams during five 30-day studies. Variations in the experimental design included assessing the effects of zinc on clams in the absence of substrate and monitoring selected groups of clams during a 34-day recovery period. They reported significant reductions in growth, significant increases in tissue water content, and significant accumulations of zinc in clams exposed to zinc at concentrations as low as 0.05 mg L^{-1} in field-located artificial streams (Table I). The extent of the significance of these changes was dependent on both zinc concentration and the exposure duration. Cessation of exposure to zinc in one study resulted in the depuration of accumulated zinc and resumption of growth at rates similar to controls for clams previously exposed to lower concentrations of zinc. Unfortunately, clams exposed to zinc in laboratory artificial streams did not exhibit any measurable growth within any treatment thereby preventing comparisons with data generated in field-located artificial streams.

Belanger *et al.* (1987) compared the results obtained from exposure of clams to asbestos in the laboratory (Belanger *et al.*, 1986a,b) with clams exposed to a natural source of chrysotile asbestos in the field. Comparisons were made on the basis of fiber size distributions and BCFs. They reported that Asiatic clams were more efficient at accumulating chrysotile asbestos in the field than was predicted from studies conducted in the laboratory (Table III). They reported low BCF values for the uptake of chrysotile asbestos in clams exposed to asbestos in the laboratory (slightly greater than 1) in comparison with BCF valves as high as 100 for clams residing in a body of water (Lake Silverwood, California) contaminated with asbestos from the natural erosion of serpentine parent rock.

Sappington (1987) assessed the effects of exposure of Asiatic clams to chlorine, ammonia, monochloramine, and copper. Studies were conducted in the laboratory and field and within an industrial cooling water system. Study durations were mostly 30 days in length but at various times of the year and a wide range of temperatures.

Sappington (1987) reported that exposure of Asiatic clams to copper in laboratory artificial streams for 30 days produced significant reductions in glycogen and significant increases in protein and tissue water content and a significant reduction in the number of clams actively siphoning (determined by recording the proportion of clams with extended

siphons at predetermined observation times) (Table I). Exposure of Asiatic clams to sublethal concentrations of copper (Table I), ammonia, chlorine, or monochloramine (Table III) in laboratory artificial streams resulted in significant reductions in the levels of glycogen and the percentage of clams observed siphoning after 30 days (Tables I and III). Exposure of Asiatic clams to copper in field-located artificial streams did not produce consistent trends in the levels of protein and percent tissue water content (Table I). Clams transplanted to sites in and around a copper contaminated effluent did not produce consistent trends in changes in glycogen and tissue water content over time or among stations (Table I).

Discussion

BIOACCUMULATION

Asiatic clams accumulate and concentrate trace elements and metals to levels that are orders of magnitude greater than levels that could be found in the waters from which the clams were collected. Rodgers *et al.* (1980) concluded that *C. fluminea* appeared to be an efficient bioaccumulator of heavy metals and other potentially toxic elements without suffering adverse effects. Graney *et al.* (1983) concluded that the Asiatic clam may be a reliable indicator of heavy metal contamination by demonstrating high BCFs for three different metals (cadmium, copper, and zinc). Harrison *et al.* (1984) and Doherty *et al.* (1988) demonstrated that dissolved copper and cadmium, respectively were readily accumulated by Asiatic clams in laboratory studies. In contrast to the conclusions of these investigators, Joy *et al.* (1983) concluded that the reliability of *C. fluminea* as a biological indicator species for some heavy metals was questionable. Unfortunately, Joy *et al.* (1983) did not report concentrations for all the elements in water preventing an assessment of the bioaccumulation potential for each of the elements by *Corbicula.*

There was only one study that simultaneously assessed the ability of clam shell or soft tissue to accumulate trace elements and metals. Rodgers *et al.* (1980) demonstrated that concentrations of trace elements were higher in soft tissues than shells. Clarke *et al.* (1979) reported that Asiatic clam shells were good indicators of contamination of aqueous systems by lead but collected empty shells rather than living organisms. Shells of Asiatic clams may be of value for monitoring contamination of freshwater systems by heavy metals but there is little experimental evidence to document conditions under which the approach would be appropriate. Previously, Imlay (1982) reviewed the use of freshwater mussel shells for monitoring heavy metals. He noted that barium, cadmium, copper, iron, lead, magnesium, manganese, mercury, nickel, silver, sodium, strontium, and zinc had all been monitored in freshwater mussel shells. He recommended the use of shells from 20 different species of freshwater mussel for monitoring heavy metals. *Corbicula* was not included in the review by Imlay (1982) but this may have been a result of the lack of published data for representatives of this genus.

Asiatic clams do not accumulate and concentrate trace elements and metals from sediments to an appreciable extent. Rodgers *et al.* (1979) reported that trace elements

concentrations were always higher in sediment than in clam tissue. Rodgers *et al.* (1980) reported that the concentrations of several elements were higher in clam tissue than in sediment but the BCFs for these elements were less than 10. In contrast, the BCFs for elements in clam tissue from water was nearly always greater than 10 and for certain elements greater than 100. Elder and Mattraw (1984) found no correlation between trace elements in sediment and clam tissue and remarked that the bioavailability of heavy metals to *Corbicula* was not necessarily related to their concentrations in sediments. Tatem (1986) reported that cadmium was not bioaccumulated from sediment by clams and there were no differences in the concentrations of lead in tissues of clams collected on days 3 and 48 of their study. Graney *et al.* (1984) noted that the uptake of cadmium by Asiatic clams was reduced in the presence of substrate.

There are fewer studies addressing the ability of Asiatic clams to concentrate pesticides but the available data indicate that Asiatic clams are efficient accumulators of lipid soluble and environmentally persistent organic compounds. Sanborn and Yu (1973) reported a concentration factor for dieldrin in Asiatic clams in excess of 1000. Claeys *et al.* (1975) concluded that the high levels of DDT found in *Corbicula* during a survey of indigenous bivalves from the Columbia River, Oregon might be a result of an extraordinary ability of this species to accumulate organochlorine compounds. Livingston *et al.* (1978) reported that concentrations or Aroclor 1254 and DDT were lower in sediment than in tissues of Asiatic clams. Leard *et al.* (1980) found that analysis of water samples did not reflect concentrations of pesticides in clams but that a ban on the widespread use of DDT and extensive flooding of rivers in Mississippi in the second year of the program was reflected in a reduction in levels of pesticides in clam tissues in the second year.

Unlike heavy metals and trace elements, experimental results suggest that pesticides in sediments are bioavailable to Asiatic clams. Elder and Mattraw (1984) noted that there was an increasing trend in the concentrations of 4 pesticides from sediments to bottom-load detritus to Asiatic clam tissue. Tatem (1986) also reported that PCBs were bioavailable to clams from sediment with measurable differences between control and treated clams after 3 days of exposure with increasing concentrations of PCBs in clam tissues through day 48. Depuration of PCBs from clam tissue was very slow. Calculated BAF values for PCBs in clams ranged from 0.54 to 12.5. Mac *et al.* (1984) reported that PCBs were accumulated from sediment by clams.

Hartley and Johnston (1983) suggested that differences in BCF values among compounds could be due to polarity of the molecule. They reported that the more polar epoxides were not bioconcentrated to the same degree as the parent compounds. Absolute levels of pesticide accumulation in clams is also related to season and physiological condition of clams (Johnston and Hartley, 1981). Hartley and Johnston (1983) noted a reduction in the concentrations of all but two pesticides in clam tissue corresponding with a reduction in fat of clams. Since other investigators have reported monthly and seasonal changes in lipid content for a variety of bivalves (Gabbot and Bayne, 1973; Dare and Edwards, 1975; Davis and Wilson, 1983; Cantelmo-Cristini *et al.*, 1985; Hagar and Dietz, 1986), it may be more appropriate to express pesticide concentrations in clams on the basis of lipid or fat content as opposed to total tissue weight.

INTERSPECIES COMPARISONS

Studies reporting comparisons between Asiatic clams and other bivalves for accumulation of heavy metals were limited. Caldwell and Buhler (1983) noted that Asiatic clams from the Columbia River, Oregon tended to have higher zinc levels in their tissues (12 to 26 μg g^{-1} wet weight) than some of the marine bivalves collected from bays and estuaries along the Oregon coast. In contrast, Elder and Mattraw (1984) noted that *Corbicula* from the Apalachicola River in Florida had lower concentrations of metals and organic compounds than other species of bivalve molluscs (primarily marine) reported in other studies but since they were different species in different systems, the factors affecting the differences could not be specified.

Studies reporting comparisons between Asiatic clams and other bivales for accumulation of pesticides were more extensive than for metals and trace elements. Modin (1969) reported that one sample of clam tissue contained 10 times more DDT (1.1 ppm) than any sample collected in San Francisco Bay. Butler (1973) noted that Asiatic clams demonstrated an ability to store organochlorine residues similar to that demonstrated by five marine molluscs. In those instances in which a material was not accumulated by Asiatic clams, it was not accumulated to any extent by any other organisms (Booth *et al.*, 1973). Claeys *et al.* (1975) found that PCBs accumulated far more heavily in Asiatic clams than in any of the other mollusks examined. They reported that Asiatic clams contained much higher levels of DDT and PCBs than individuals of *Anodonta* collected from the Columbia River, Oregon and while DDT levels in *Anodonta* were seasonally influenced, those in *Corbicula* were not. Leard *et al.* (1980) reported that *Corbicula manilensis* (now identified as *C. fluminea*) was the best indicator of organochlorine pesticides in major Mississippi river systems in comparison with six other freshwater Pelecypoda.

In contrast to these studies, Sanborn and Yu (1973) reported that dieldrin was concentrated to a greater degree in other organisms of the laboratory microcosms even though concentration factors for Asiatic clams were in excess of 1000. Mac *et al.* (1984) concluded that the oligochaete worm *Octolasion tyrtaeum* was more appropriate for accumulation studies of PCBs than Asiatic clams. They felt that use of Asiatic clams for measuring bioaccumulation was questionable because it may cease feeding, it had the lowest BCF in comparison with *O. tyrtaeum,* fathead minnows, and yellow perch, and they were uncertain of the appropriate material to analyze (tissue or shell).

GEOGRAPHIC DISTRIBUTION AND AVAILABILITY

Additional factors influencing the choice of a biomonitoring organism is the availability of that organism either in extensively distributed indigenous populations throughout the contaminated areas or as an abundant source of organisms in uncontaminated areas that can be transplanted to sites of interest. Claeys *et al.* (1975) found that the Asiatic clam and a species of *Anodonta* were the most abundant molluscs at sites in the Columbia River from which samples were collected. Foster and Bates (1978) employed *Quadrula quadrula* to monitor copper levels in a receiving stream below an electroplating plant outfall but recommended the use of *Corbicula* for in-stream monitoring studies with caged organisms because of their abundance and distribution. Elder and Mattraw (1984) selected Asiatic

clams for trace metal analysis because it was the only mollusc that could be collected in large quantities from various points on the Apalachicola River in Florida. Although Mac *et al.* (1984) recommended the use of an oligochaete as opposed to Asiatic clams for accumulation studies of PCBs, they conceded that *O. tyrtaeum* was not readily available.

Despite these arguments, Asiatic clams should not be used as a biological monitor in systems in which it is not already present. Some investigators have reported that *Corbicula* may have displaced endemic bivalves from systems in which it has become established (Gardner *et al.*, 1976; Fuller and Richardson, 1977; Kraemer, 1979). It has also invaded industrial cooling water systems of production and power plants causing severe economic consequences (Morton, 1979; Cherry *et al.*, 1980; Hamm, 1982). While the need to monitor levels of environmental contamination through biological agents is a worthwhile endeavor, it does not warrant the man-assisted spread of an invasive biofouling organism.

ALTERNATE ASSESSMENT APPROAHCES

Several investigators demonstrated that there are sub-lethal parameters other than tissue concentrations of contaminants by which detrimental effects on aquatic organisms can be assessed. Alternate approaches to assess the impacts of aquatic contaminants on an environment as reflected in the responses of Asiatic clams include both traditional and non-traditional methodologies. These include tissue water concentration (Belanger *et al.*, 1986b,c), histopathology (Martin and Sparks, 1971; Belanger *et al.*, 1986a,b; 1987), adenylate energy charge (Geisy *et al.*, 1983; Cantelmo-Cristini *et al.*, 1985), reproductive output (Belanger *et al.*, 1986b), condition and scope for growth (Foe and Knight, 1987), enzyme inhibition (Farris, 1986; Farris *et al.*, 1988), and growth (Fritz and Lutz, 1986; Belanger *et al.*, 1986a,b,c; Foe and Knight, 1987). The advantages of these approaches are that they do not rely on sophisticated instrumentation to determine tissue concentrations of contaminants, they do not distinguish between classess of contaminants but respond in a negative fashion to any detrimental agent, and they enable the investigator to monitor the impact of variable effluents on organisms that may not have bioconcentratable components (e.g. temperature and pH). Investigators may monitor trends in mortality at sites of increasing distances from a point of entry and changes in sublethal physiological parameters among clams at sites not experiencing mortalities. All of these approaches provide information regarding the existence of detrimental conditions for clams at a particular site. Preference for an approach by an investigator will be a site-specific question relative to the expected class of contaminant, the extent of the resources available to the investigator, and the degree of sensitivity viewed necessary to demonstrate the absence or existence of detrimental conditions.

Despite the broad array of techniques available for demonstrating deleterious conditions in the environment, it appears that field studies provide greater consistency in results than laboratory studies attempting to predict the results of exposure of clams in the field. Belanger *et al.* (1987) noted nearly a 100-fold difference in the concentration of asbestos by clams in the field as opposed to those exposed to asbestos in the laboratory. Graney *et al.* (1984) identified several factors that influenced the uptake of heavy metals by Asiatic clams to the point that field conditions may not be realistically replicated in the lab.

Farris (1986) reported no differences in cellulolytic activity among treatments for clams exposed to zinc in laboratory artificial streams but found significant differences among treatments when clams were exposed to zinc in field-located artificial streams. Belanger *et al.* (1986c) were unable to promote growth of clams exposed to zinc in laboratory artificial streams for any treatment including control clams (perhaps due to an inadequate diet) preventing comparisons with data generated in field-located artificial streams.

Factors contributing to this dichotomy may center on the inability of investigators to replicate complex field conditions in laboratory simulations. Non-replicable field conditions would include the influence of seasonal changes in abiotic parameters during long-term studies (>30 days) and the availability of an adequate diet for clams in long-term studies. There is little information available regarding dietary requirements of clams, metabolic processes in relation to diet and temperature, the capacity for anaerobic metabolism in clams and its relationship to temperature, and the physiological consequences of suboptimal diets in conjunction with exposure to potentially toxic substances.

Conclusion

Asiatic clams are widespread, abundant, fast growing, generally tolerant of exposure to toxic materials, and possess multiyear lifespans. They concentrate organic pollutants from both water and sediment and heavy metals from water. A variety of sublethal parameters are currently being investigated to assess the ability of *Corbicula* to monitor the presence of non-bioconcentratable pollutants in freshwater systems. Unfortunately, the results of laboratory studies may be limited in predicting the responses of clams exposed to stressors in the field. While these conclusions would normally lead to a recommendation for field studies, Asiatic clams should not be introduced to *Corbicula*-free drainage systems for biomonitoring activities because of its invasive biofouling attributes.

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