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New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	Phthalates
Description of stressor	Phthalic acid diesters, also called phthalates, have been used as plasticisers in many plastics since the 1930's, with a quarter of the total plasticiser ever produced being diethylhexylphthalate. Phthalates are used industrially in the production of a variety of household and consumer goods including plastic polymers, lubricating oils, and carriers for perfumes in cosmetics. Because of their widespread use, phthalates have become one of the most abundant industrial pollutants in the environment. Many phthalates are also classified as toxic chemicals by the EPA's Toxic Release Inventory.
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	Phthalates are fat soluble, so tend to concentrate in body fat. Several phthalates are testicular toxicants. Part of this toxicity is believed to involve depletion of testicular Zinc, and may include the death and disintegration of the testicular germ cells. Therefore, phthalates are likely to have a negative effect on animal reproduction, which is critical to Biological integrity, Biodiversity, Habitat/ecosystem health, and Ecosystem function.
Key impacts selected (critical ecological effects)	Due to their persistence in the environment, phthalates are commonly found in groundwater, rivers and drinking water. Inland waters in or near contaminated sites remediated by NJDEP or EPA are always tested for contaminants including phthalates. Therefore, a sampling of this ready source of data will be used for this report. There are several types of phthalates used in industry and subsequently found in the environment. For simplicity purposes, this report focuses on Bis-2-ethylhexylphthalate, one of the more common phthalates. While little research has been done on the mechanism by which phthalates are transported, how far, etc., for the sake of this report I will assume that phthalates are distributed evenly across all ecosystems and the closer an ecosystem is to an industrial area, the greater the likelihood that those ecosystems will have a higher concentration of phthalates.
Exposure Assessment	
Exposure routes and pathways considered	The following benchmarks are used in this study: Toxicity of chemicals to earthworms Phytotoxicity in soil Freshwater life (surface waters) chronic screening values Sediment

Population(s)/ecosystem(s) exposed statewide

Lower Raritan River:
 A variety of pollutant sources have likely impacted the lower Raritan River. These include landfills, industrial sources, and permitted discharges from municipal wastewater treatment plants and non-point source pollution has contributed to the overall contaminant loadings in surface water and sediment.

Inland waters:
 Discharge Pond/ Unnamed tributary to Whippany River, Lucent Technologies, 67 Whippany Rd., Morris County, NJ
 The former discharge pond receives storm water runoff from the site and eventually flows into the Whippany River. The pond more closely resembles an intermittent stream and riparian habitat along the stream. The width ranges from 2 to 3 feet and begins as an 8 x 10ft ponded area. At dry periods soils resemble flood plain soil. Numerous herbaceous and woody plant species have been identified in the area.

Nellie's Pond, Conwed Boded Fiber, Delanco, Riverside Township, Burlington Co., NJ

Las Brisas Lake, Voorhees, NJ

Bordens Run Creek, Plumstead, Ocean Cty., NJ

Fenwick Creek, National Freight, Salem Twp., Salem Cty., NJ

Quantification of exposure levels statewide

Bis-(2-ethylhexyl)phthalate
 1996-1999
 Threshold Effect Level (TEL) = 182 ug/kg (Marine Sediment)
 Upper Effects Threshold (UEL) = 750 ug/kg (Freshwater)

Lower Raritan:
 Edgeboro landfill
 Actual value for sediment: 3800 ug/kg

Inland Waters:
 Lucent Discharge Pond
 Actual value for sediment: 3200 ug/kg

Nellie's Pond
 Actual value for surface water: 2.9 µg/L

Las Brisas Lake, Voorhees, NJ

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	<p>Actual value for sediment: 238 ug/kg</p> <p>Bordens Run Creek, Plumstead, Ocean Cty., NJ Actual value for sediment: 420 ug/kg</p> <p>Fenwick Creek, National Freight, Salem Twp., Salem Cty., NJ Actual value for sediment: 240 ug/kg</p>
Specific population(s) at increased risk	Very little research has been conducted on phthalates effect on the natural environment. The bulk of the research on this toxin has been related to human exposure.
Quantification of exposure levels to population(s) at increased risk	There is no tissue data for animals exposed to Phthalates.
Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	Random sampling of existing data sets for site remediations involving inland waters.
Risk Characterization	

<p>Risk estimate(s) by population at risk</p> <p>Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)</p> <p>{tc \l2 "Score}Assessment of severity/irreversibility</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage: - many species threatened/endangered - major community change - extensive loss of habitats/species Long time for recovery</p>	<p>Much is yet to be learned about the severity of phthalates on ecosystems. Many phthalates are classified as toxic chemicals by the EPA's Toxic Release Inventory.</p>	<p>Score</p> <p>3</p>
3 - Adverse affect on structure and function of system: all habitats intact and functioning		

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<p>population abundance and distributions reduced Short time for recovery 2 - Ecosystem exposed but structure and function hardly affected 1 - No detectable exposure</p>		
<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade) 5 - Often and increasing 4 - Often and continuing 3 <input type="checkbox"/> Occasional 2 <input type="checkbox"/> Rare 1 - Possible in the future 0 <input type="checkbox"/> Unlikely (or 0.1)</p>	<p>Because of their widespread use, phthalates have become one of the most abundant industrial pollutants in the environment.</p>	3
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude) 5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>See above.</p>	2
	<p>Total</p>	18

<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>Much is yet to be learned about the severity, behavior, transport mechanisms, etc. of phthalates on ecosystems. (H)</p>
<p>Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief description. (Data Gaps; highlight significant data needs)</p>	<p>See above (H).</p>
<p>Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, -, =, ≡, where + is improvement), and brief description.</p>	<p>(0). The more we learn about the threat from this pollutant, the better capable we will be in the future for controlling it.</p>
<p>Potential for catastrophic impacts* (H,M,L) and brief description (*Short-term drastic negative impacts having widespread geographic scope)</p>	<p>L: Although there is limited knowledge on the effects of phthalates, it is likely the potential is low for widespread drastic impacts.</p>

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Link to other Work Groups (e.g., socioeconomic impacts)	The bulk of the research conducted on phthalates relates to human health.
Extent to which threat is currently regulated or otherwise managed	Reporting requirements for EPCRA.
Barriers to restoration	Phthalates in the environment show a moderate resistance to bacterial degradation, but are easily degraded in the laboratory.
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	H: Phthalates are used industrially in the production of a variety of household and consumer goods including plastic polymers, lubricating oils, and carriers for perfumes in cosmetics.
Small business industry	H
Transportation	L
Residential	L
Agriculture	L
Recreation	L
Resource extraction	L
Government	L
Natural sources/processes	L
Orphan contaminated sites	M
Diffuse Sources	
Sediment sinks	H
Soil sinks	M
Non-local air sources incl. deposition	H

Biota sinks	H
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Summary Statement:

Phthalic acid diesters, also called phthalates, have been used as plasticisers in many plastics since the 1930's, with a quarter of the total plasticiser ever produced being diethylhexylphthalate. Phthalates are used industrially in the production of a variety of household and consumer goods including plastic polymers, lubricating oils, and carriers for perfumes in cosmetics. Because of their widespread use, phthalates have become one of the most abundant industrial pollutants in the environment. Phthalates are fat soluble, so they tend to concentrate in body fat. Several phthalates are testicular toxicants. Therefore, phthalates are likely to have a negative effect on animal reproduction. Due to their persistence in the environment, phthalates are commonly found in groundwater, rivers and drinking water. For simplicity purposes, this report focuses on Bis-2-ethylhexylphthalate, one of the more common phthalates. While little research has been done on the mechanism by which phthalates are transported, how far, etc., for the sake of this report it was assumed that phthalates are distributed evenly across all ecosystems and the closer an ecosystem is to an industrial area, the greater the likelihood that those ecosystems will have a higher concentration of phthalates. Due to phthalates prevalence in the environment and toxicity, the scores are moderate: Total Score - 110; Average Score – 22.

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Statewide Analysis of Threat – Phthalates

Threat = PHTHALATES

{"Threat = PHTHALATES" }

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	4	4	3	48
Marine Waters	2	3	2	12
Wetlands	4	4	3	48
Forests	1	1	1	1
Grasslands	1	1	1	1
			Total Score	110
			Average Score	22

Risk by Watershed Management Region

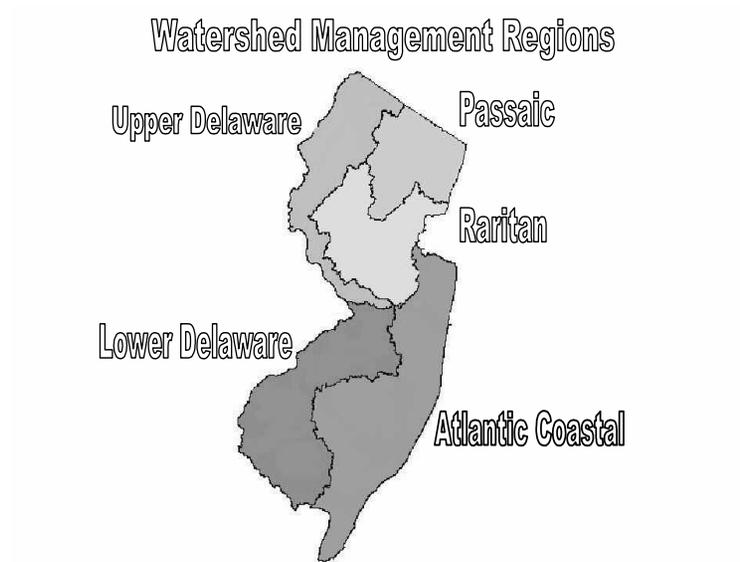
{"Risk by Watershed Management Region" }

THREAT =	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	M	NA	H	L	L
Passaic	M	L	H	L	L
Raritan	M	L	H	L	L
Atlantic	M	M	H	L	L
Lower Delaware	M	M	H	L	L
Region/Watershed (secondary)					

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Urban	M	M	H	L	L
Suburban	M	M	M	L	L
Rural	L	L	L	L	L

H=high, M=medium, L=low, NA = not applicable



New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	Polychlorinated Biphenyls (PCBs)
Description of stressor	<p>PCBs are industrial products used from the 1930s into the 1970s. EPA banned its production and use in 1977 although presence in the environment continues because of the chemical stability of the compounds and releases from historical uses. PCBs are lipid soluble; for that reasons and their biological stability, they tend to bioaccumulate in animal species.</p> <p>PCBs are very stable compounds to thermal and chemical stresses making them ideal for certain industrial applications such as for heat transfer, hydraulic systems and in electrical components. An unfortunate consequence of their stability is their persistence in biological systems. While the last of PCBs were manufactured almost 25 years ago, their presence in the environment will be noted for decades to come.</p> <p>There are many forms of PCBs depending upon the number and location of chlorine atoms. This fact makes describing the risks from PCBs problematic because the different chemical forms have different toxicological properties. For the purpose of this report, we refer to the total levels of PCBs and apply risk factors based on the greatest toxicity. This is not an unreasonable set of assumptions, because the most persistent of the PCBs are heavily chlorinated, and exhibit the greatest level of toxicity. However, as with other assumptions in risk assessments, assuming the most toxic forms of PCB to carry out the analysis results in a possible overstatement of specific effects.</p> <p>The path for PCBs to enter the environment is linked to historical and ongoing soil contamination and runoff into lakes and rivers. Most PCBs associate with soil material and are transported via erosion. Sediments at the bottom of lakes and rivers are the sinks of the greatest concentrations.</p> <p>Reproductive effects may be the most sensitive endpoint for PCB toxicity. This may be related to the biochemical function of PCB to induce mixed function oxidases thereby increasing the rate of formation of reactive epoxides, known to be active in conjugation with DNA (EPA, 1999). This also explains the carcinogenic potential of PCB's.</p> <p>In addition to reproductive and carcinogenic effects, PCBs have shown neurotoxicity, immunological effects, hepatotoxicity, effects on the gastrointestinal and respiratory systems, and dermal effects (EPA 1999). Most of these effects occur at concentrations of PCBs higher than found in the environment, therefore ecological observations focus on reproductive effects which can result in observable effects under current levels of contamination. The correlation of PCB structure to toxicity is based on the ability of the molecule to assume a planar configuration. Chlorination of the bi-cyclic structure is a factor, which promotes planar geometry. The more planar the PCB, the closer is its reactivity properties to 2,3,7,8-tetrachloro dibenzo-p-dioxin. Dioxin is a planar molecule that is very efficient at MFO induction and resistant to further oxidation leading to its persistence.</p>

	<p>PCBs are persistent, bioaccumulative and toxic (PBTs) as defined by EPA. The biomagnify up the food chain (i.e., concentrations of PCBs increase in biota as you move to higher trophic (feeding) levels in the food chain).</p>
<p>Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function</p>	<p>There are many studies of PCB effects in humans that show developmental problems, reproductive problems and immune system problems that show up as increased susceptibility to infection (NJCRP). Studies on other species focus on reproductive efficiencies, particularly with eggshell thinning in birds and the viability of fish fry to lower concentrations; common effects from many chlorine containing pollutants such as chlorinated pesticides in addition to PCBs. Other species at the top of the food chain feeding off fish show developmental and reproductive problems as well.</p> <p>Fish tissue contamination is a good indicator of PCB contamination and a key link in the pathway to toxicity for fish-eating (piscivorous) species. Mammals and higher trophic level fish and mammals and birds that use fish as a significant portion of their diet may be at risk when PCB concentrations are elevated although monitoring the tissue level of PCBs in terrestrial species is less often carried out.</p> <p>There are also effects to species living in the sediments (benthic infauna) exposed to direct contamination. The dermal contact and ingestion of sediment material provides two paths for PCBs to accumulate in the tissue. Mortality from direct exposure is the most easily studied and reported PCB toxicity evaluation. Pollutant tolerant species can dominate an ecological niche when contamination levels are high and pollution intolerant species die off.</p>
<p>Key impacts selected (critical ecological effects)</p>	<p>For the reasons cited above, this report focuses on three key impacts: Reproductive effects in raptors. Fish tissue contamination Biodiversity in sediment-dwelling (benthic) invertebrates.</p>
<p>Exposure Assessment</p>	
<p>Exposure routes and pathways considered</p>	<p>Transport of PCBs into aquatic environments is largely through the mobilization of PCB containing soils. There is air deposition of PCBs due to the continual remobilization of the contaminant from soils and water (often through flooding) to the air.</p> <p>The primary focus for PCB ecological effects is in aquatic systems. Uptake of PCBs from the sediment is undertaken by benthic insects and other invertebrates. PCBs are largely associated with sediment, but equilibrium conditions allow for low concentrations in the water column as well. Direct exposures can take place with dermal contact of sediment, ingestion of sediment particles, and ingestion of water. Dermal contact is an important pathway for benthic infauna such as aquatic worms (EPA, 1999). After direct exposures, PCBs accumulate in lipid tissue. Bioaccumulation results with the ingestion of directly exposed species by higher order species. PCBs are lipophilic (they partition to lipid and are not very soluble in water). Species of small vertebrates including fish feed on the invertebrates. Because of bioaccumulation, higher order predators show higher concentrations of the PCBs in their tissue. In general, the concentrations in higher order consumers are higher than those exposed through contact and water intake, and the effects are more pronounced.</p>
<p>Population(s)/ecosystem(s) exposed statewide</p>	<p>Both freshwater and marine ecosystems suffer from PCBs effects. Freshwater systems may be of greater importance due to the localized high concentrations in freshwater sediments higher than on the ocean floor (EPA 1999)</p>

	<p>due to the localized high concentrations in freshwater sediments, higher than on the ocean floor (EPA, 1999). Estuaries are another important site of PCB contamination. Estuaries are an important habitat for species migrating between marine and freshwater environments, and estuaries are a natural area for sedimentation, meaning a natural site for PCB deposition.</p> <p>There are three measures of exposure that help describe PCB levels in the New Jersey environment: Loading from known sources of contamination, Sediment concentrations Tissue prevalence</p> <p>There are two studies informative regarding sources of contamination. In the Delaware River watershed, transport of PCBs into the River are quantified during low flow and high flow events. (Delaware Basin Commission, 1998). In the New York/New Jersey Harbor system, measured and modeled transport from the Hudson river has been reported (EPA 1998). The main value of these studies is to identify potential locations where elevated PCBs may be of interest. However, translating transport calculations to biological effect is problematic.</p> <p>Sediment concentrations are available from a variety of sources covering different parts of the state. As with the loading studies, the greatest information is available for the Delaware River basin and the New York/New Jersey bays. Additional information is available from particular inland sites of known contamination (NJDEP, unpublished).</p> <p>Sediment contamination is useful for understanding biological effects because of a strong correlation (and significant causation) between sediment PCBs and biological uptake, first through benthic organisms but ultimately in fish tissue and piscivorous species.</p> <p>The final pieces of exposure data are tissue, organ and egg samples. These most direct measures of PCB contamination cover only parts of the state, but provide a picture of the range of possible impacts.</p> <p>The PCB studies in the Hudson River provide the most comprehensive analysis of connections between sediment concentrations and contamination of aquatic and terrestrial species. From this study, some generalizations are possible linking sediment concentrations to possible contamination of aquatic species. (Terrestrial contamination is more difficult to ascertain because of the variable rate of aquatic species as food for terrestrial species.)</p>
<p>Quantification of exposure levels statewide</p>	<p><u>Measurements of Source loadings</u></p> <p>In the Delaware River system, a study by the Delaware River Commission shows that continuing loading of PCBs is taking place. In this study, it was reported that about 36 g/day are added to the river during low flow conditions. During high flow conditions (spring melts and heavy rain), greater than 900 g/day are added to the river system from tributaries.</p> <p>A small percentage of these loadings (0-2%) occur up stream from Metropolitan Philadelphia. More than half occur in the Philadelphia area with the remainder (2-30%) resulting from downstream sources (Delaware Basin Commission, 1998).</p>

	<p>In the lower Hudson River, contamination from upstream continues to deliver PCBs in significant concentrations to the New York/New Jersey Harbor. Tens of thousands of kilograms of PCBs exist in upstream sediments and these continue to move downstream mostly via sediment movement. (EPA, 1998). In addition, local sources from Metropolitan New York are probably adding further PCB loads although no studies on those loads are available for this report.</p> <p>Measures of sediment concentrations</p> <p>In New Jersey, the greatest historical contamination of sediment noted for this report is a contaminated site known as the Burnt Fly Bog, which ultimately empties into South River (Monmouth County). Sediment concentrations of PCBs were noted as 77.4 ppm. In other contaminated sites, such as Green Pond Brook of a discharge pond emptying into Whippany River or a wetland emptying into Lake Lefferty, concentrations range closer to 1 ppm. (NJDEP, unpublished) These isolated sites compare to the general contamination of the Delaware River (.06-.2 ppm) or sediments in the New York/New Jersey Harbor (typically in the .2 – 1 ppm range). In all cases, these concentrations exceed Threshold effect values, set at .07 ppm by the state of New Jersey (NJDEP 1998) .04 ppm in New York for the Hudson River and as low as .01 ppm set by the Ontario Ministry of the Environment (as compiled in EPA, 1998, Table 5-6.)</p> <p><u>Measures of tissue concentrations</u></p> <p>A key indicator for PCB concentrations is fish tissue. Fish tissue levels of PCBs are a good indicator of PCB levels in sediments for areas serving as fish habitat. In addition, other species that may be a food source for higher trophic levels may show elevated concentrations when local fish show elevated concentrations. In areas with greater PCB contamination (such as the New York/New Jersey harbor), fish show greater concentrations of PCBs than in less contaminated river systems (such as Southeast New Jersey.) Variations in fish tissue concentration are both species and watershed dependent. Higher values are in bluefish and American eels, which often show levels greater than 1 ppm while other species such as Bullhead and perch are typically lower than 1 ppm. Threshold values for fish are as set as low as .29 ppm protective of sensitive fish fry. Other threshold values for other species are in the range of 1.5 to 3.1 ppm. Therefore, there are some exceedances of some of the threshold values.</p>
Specific population(s) at increased risk	Primarily, piscivores including osprey, eagles, mink, otter with some potential on omnivores such as raccoons and potentially insectivores such as swallows and bats.
Quantification of exposure levels to population(s) at increased risk	Fish are the primary source of PCB in the diet of effected species. Therefore, measures of fish tissue concentration provide a useful indicator of potential PCB exposure to piscivores such as the specific populations identified above.
Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	<p><u>Reproductive effects in raptors</u></p> <p>Elevated concentrations of PCBs measured either in avian tissue or in eggs are indicative of reproductive success. A benchmark level of PCBs in avian eggs for the Hudson is .33 ppm. (as reported in EPA (b) In the New Jersey studies looking at raptor population carried out between 1989 and 1998 concentrations in osprey eggs have decreased (from 7.7 to 3.3 ppm in the Delaware Bay, and from 4.1 to 1.8 on the Atlantic Coast) during the time period from but remain significantly above the benchmark levels. (Clark, Stansley, and Niles, 2000)</p>

When observing reproductive effectiveness of the osprey, there is a marked increase in eggshell thickness and nesting success when comparing the 1989 and 1998 results. Because the raptors are increasing in population, it is possible that an effective threshold for reproductive success has been re-crossed, this time to the advantage of the osprey.

Noting the link between egg contamination and fish consumption,. In 1989, the concentrations in sampled fish ranged from .18 – 1.2 ppm. In 1998, that declined to .06 - .43 ppm.

In the language of benchmarks, One EPA study resulted in a recommended criteria for chemical residue concentrations of 0.11 ppm for fish eating wildlife. Again, the osprey egg evidence supports that impacts to reproductive success are noticeably enhanced due to reductions in the contamination of consumed fish.

Tissue contamination of fish

Tissue concentrations vary significantly based on two factors; species and location. Only a few species have differential PCB benchmark values (salmonids being the primary example having a lower benchmark), and most of the species with contamination data have similar benchmark values. Therefore, the major conclusion to draw from contamination data is the geographic location of contaminants. The Upper New York/New Jersey Harbor system, including Newark Bay are hosts to fish with the greatest contamination in the state. In 1982, several species exhibited tissue concentrations greater than 2 ppm (leading to actions closing the area as a fishery.) (Belton and Ruppel, 1983). Later in the 1980s and early 1990s, concentrations had decreased somewhat but striped bass were still showing average concentrations approaching 2 ppm (Kennish and Ruppel, 1996) The Delaware River Basin also shows decreases in fish tissue concentration although at generally lower levels than in seen in the New York/New Jersey harbors. For White Perch, tissue concentrations decreased from 1.2 ppm to .37 ppm. The change in Channel Catfish has been less dramatic, decreases from 0.67 to 0.44 ppm.

In the Hudson study, the Upper Hudson near the General Electric plants (an historic source of PCB pollution) show PCB contamination larger than any recorded in New Jersey (sediment concentrations greater than 100 ppm). While the sediment and fish tissue concentrations are well above threshold values, the impacts on fish populations or biodiversity are not observed. This is an indicator that other stresses on fish populations may be stronger factors in their survival than PCB contamination. This effect can be extrapolated to New Jersey, as well. Habitat conditions for fish nesting, zebra mussels, and river channel modifications probably have greater impacts on fish species than do PCB contamination (New Jersey 305(b) reports).

Studies leading to the development of threshold values show a range of observed effects. Most effects on adult fish only take place when tissue concentrations are greater than 20 ppm. Much greater sensitivity is observed on fish fry survivability and egg viability. In these cases, concentrations greater than 1 ppm PCB shows effects.

As a result of these studies, several benchmark values exist. For fish reproduction and fry viability, EPA cites benchmarks ranging from 0.1-40 ppm (EPA, 1999, Table 4-7) While concentrations in fish often exceed the lower range of benchmarks for fry viability, one factor influencing fish tissue concentration is the age of the fish. Young fish do not have the exposure levels of older fish. Unfortunately, there is little effort being undertaken to measure PCB concentrations in developing fish. To fill in this knowledge gap, we can rely on overall fish species population

	<p>numbers to determine if there are gross effects on fry viability. Results from the Hudson River study suggest that PCB effects on populations are not significant on populations. (EPA, 1999)</p> <p><u>Biodiversity in sediment dwelling (benthic) invertebrates</u> There is no data available for this report that directly measures benthic invertebrate concentrations of PCBs. However, the intimate connection between sediment and these species allows for correlations between known sediment concentrations and benthic invertebrate concentrations of PCBs. In the Hudson River study, benthic invertebrates exceed threshold values whenever concentrations in sediment result in whole body concentrations greater than 5 ppm using the most sensitive species identified in the Hudson River study (EPA, 1999, Table 4-4).</p>	
Risk Characterization		
<p>Risk estimate(s) by population at risk</p> <p>Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)</p>	<p>Reproductive effects in raptors – 3.5 X 3 X 3 Benthic invertebrates – 1.5 X 3 X 1 Fish Tissue concentrations – 2.5 X 2 X 2.5</p>	<p>Score 33.5 4.5 12.5</p>
<p>Assessment of severity/irreversibility</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage:</p> <ul style="list-style-type: none"> • many species threatened/endangered • major community change • extensive loss of habitats/species <p>Long time for recovery</p> <p>3 - Adverse affect on structure and function of system:</p> <ul style="list-style-type: none"> • all habitats intact and functioning • population abundance and distributions reduced <p>Short time for recovery</p> <p>2 - Ecosystem exposed but structure and function hardly affected</p> <p>1 - No detectable exposure</p>	<p>Reproductive effects in raptors – 3-4 Benthic invertebrates – 1-2 Fish tissue concentrations – 2-3</p>	<p>33.5 3 12.5</p>

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<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade)</p> <p>5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)</p>	<p>Reproductive effects in raptors 3 Benthic invertebrates - 3 Fish tissue concentrations - 2</p>	
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude)</p> <p>5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>Reproductive effects in raptors -3 Benthic invertebrates - 1 Fish tissue concentrations - 2-3</p>	
	Total	17
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>M - We base most of our conclusions on the cause and effect linkages developed in the Hudson River studies to support PCB removal actions sponsored by USEPA. This extensive study is one of the few that relates sediment concentrations, concentrations in benthic organisms, fish tissues and terrestrial piscivores. Unfortunately, as with most chemical assessments, the impacts noted in the Hudson are confounded by the existence of many other pollutants, particularly chlorine containing contaminants that exhibit similar effects to PCBs.</p>	
<p>Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief description. (Data Gaps; highlight significant data needs)</p>	<p>L/M - PCBs have been in the research spotlight for more than 20 years. The remaining challenge is to isolate PCB specific impacts from impacts noted for general chemical pollution. There are few prospects that the results from this research are forthcoming.</p>	
<p>Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, !, =, ≡; where + is improvement), and brief description.</p>	<p>++ Estimates from the Great Lakes States suggest that reductions in fish tissue contamination are decreasing 10% annually. (Michigan DEQ, 1997) This presumes no additional point source loading (and assumes no air deposition.) but does allow for the continuing cycling of previously deposited contamination. The studies on osprey suggest similar decreases in New Jersey. Flooding events will continue to provide PCB loadings to watersheds and sediments as noted in the Delaware River study showing significant loading during wet weather events.</p>	
<p>Potential for catastrophic impacts* (H,M,L) and brief description (*Short-term drastic negative impacts having widespread geographic scope)</p>	L	
<p>Link to other Work Groups (e.g., socioeconomic impacts)</p>	?	
<p>Extent to which threat is currently regulated or otherwise managed</p>	<p>There are two keys to PCB management. The first is the elimination of continued manufacture and use. Because this step took place in 1977, concentrations are decreasing. The other step is the removal and isolation of PCB</p>	

	<p>contaminated sediments. In some cases, additional sedimentation will cover past contaminants, but disruption of river and lake sediments enhances the possibility of future mobilization. Recent evidence of resuspension is available in data measuring PCB concentrations in suspended sediments in the Lower Hudson and Raritan Bay. In these studies, measures taken later in the day show greater concentrations of PCB contaminated sediment, presumably as the result of greater boat traffic causing disruption of sediments.</p> <p>Control of PCB discharges and the remediation of PCB-contaminated hazardous waste sites are regulated under the Industrial Site Recovery Act (ISRA), Spill Compensation and Control Act, Toxics Substances Control Act (TSCA), Water Pollution Control Act (WPCA), Resource Conservation and Recovery Act (RCRA), Comprehensive Environmental Response, Compensation and Liability Act of 1980, as amended by Superfund Amendments and Reauthorization Act of 1986 (CERCLA) and the Hazardous Site Discharge Remediation Act.</p>
Barriers to restoration	Dredging is a two-edged sword. While leading to a long term reduction in PCB contamination of watersheds, the dredging event can result in short-term mobilization of significant PCBs and cause movement of PCB contamination downstream.
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	Historical industry releases
Small business industry	L
Transportation	
Residential	L
Agriculture	L
Recreation	L
Resource extraction	L
Government	L
Natural sources/processes	L
Orphan contaminated sites	M
Diffuse Sources	
Sediment sinks	H
Soil sinks	M (soil analysis in New Jersey did not identify any general level of contamination. However, contaminated sites may continue to be discovered based on the use of PCBs for many applications prior to their prohibition (Fields et al, 1993).
Non-local air sources incl. deposition	PCBs continue to cycle through the atmosphere
Biota sinks	M Food chain transfers are important sources to upper trophic level biota.

Issue: Polychlorinated Biphenyls - PCBs
Author: Ken Jones (Green Mountain Institute)
Version: 01/04/01

Summary Statement:

PCBs continue to exist in aquatic environments at levels that are known to cause biological effects. Some species may have been harmed to a significant extent effecting their population levels and viability as a dominant species. In most cases, these species are making a recovery, but the levels of PCBs are still sufficient to pose challenges to their complete recovery. In many ways, PCBs represent a useful description of reversible damages caused by persistent bioaccumulating toxics. While the use and manufacture of PCBs was eliminated in 1977, sediment contamination persists and the alteration of species populations will take decades to recover. The good news is that there is evidence of recovery even with the continuing presence of contamination.

There remains uncertainty as to whether there are particular hot spots that have significantly altered ecological function due to PCBs. In most cases of localized PCB contamination, there are other chemical and physical threats that confound conclusions regarding the isolated effects of PCBs.

In summary, the “M” ranking for irreversibility is based on the evidence that over 10 years, some possible changes have been noted, but damage is still evident and additional decades are necessary to see full recovery of affected species (if recovery is possible)

The “M-H” ranking of frequency is based on PCB contamination occurring generally throughout New Jersey watersheds.

The “M” ranking of magnitude is based on some visible evidence of impact, but not the wholesale destruction of ecological systems that may exist with other environmental stressors.

The threats to marine waters are considered less significant than inland waters. Wetlands are presumed to be similar to other inland waters.

Issue: Polychlorinated Biphenyls - PCBs
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Statewide Analysis of Threat

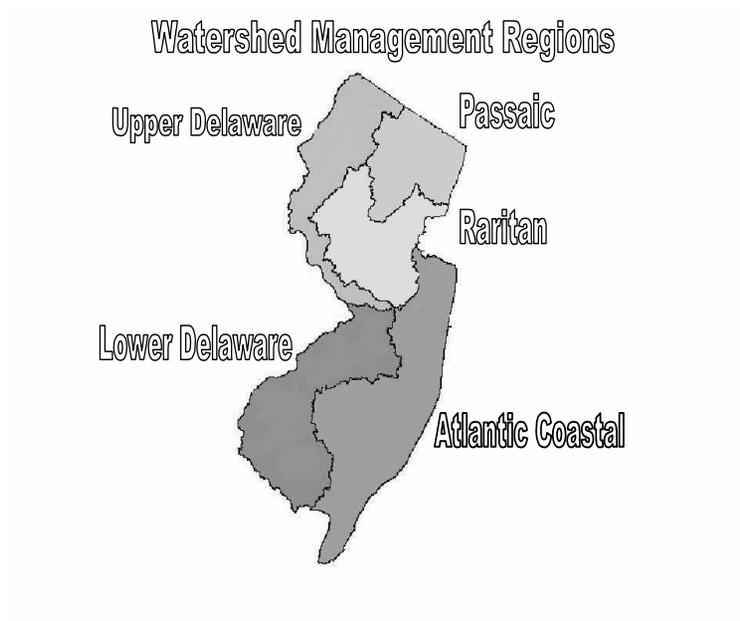
Threat = Medium

Ecosystem	Severity	Irreversibility	Frequency	Magnitude	Score
Inland Waters	3		5	3	45
Marine Waters	3		3	2	18
Wetlands	3		5	3	45
Forests	1		1	1	1
Grasslands	1		1	1	1
Total Score					110
Average Score (Total ÷ 5)					22

Risk by Watershed Management Region

THREAT =	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L	NA	NA	NA	NA
Passaic	M/H	M	NA	NA	NA
Raritan	H	M	NA	NA	NA
Atlantic	L/M	L	NA	NA	NA
Lower Delaware	M	L/M	NA	NA	NA
Region/Watershed (secondary)					
Urban	M	M	M	L	L
Suburban	L/M	L/M	M	L	L
Rural	L	L/M	M	L	L

H=high, M=medium, L=low, NA = not applicable



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New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
<p>Hazard Identification</p> <p>Stressor</p> <p>Description of stressor</p>	<p>Polycyclic Aromatic Hydrocarbons (PAH)</p> <p>Polycyclic Aromatic Hydrocarbons (PAHs) are compounds composed of two or more fused benzene rings. While there are thousands of PAH compounds, environmental concern has focused on those that range from molecular weight 128 (Naphthalene, 2-ring structure) to 300 (Coronene, 7- ring structure). PAHs are products of incomplete combustion and ubiquitous in the environment as a result of natural processes (e.g., forest fires, microbial synthesis, volcanic activities) and anthropogenic activities (e.g., industrial activities using high temperature combustion/pyrolysis of organic matter, petroleum releases). PAHs comprise 15-40% of heavy fuel oils (e.g., Fuel Oil Nos. 4,5, and 6- Bunker C). These higher boiling distillates are less volatile and less water-soluble than lighter petroleum fractions, thus more resistant to degradation and abiotic losses, less mobile, and more persistent in the environment.</p> <p>PAHs vary substantially in their toxicity to plants and wildlife. Generally, unsubstituted, more soluble, lower molecular weight PAHs (2 or 3 rings) are acutely toxic but not carcinogenic; higher molecular weight, less soluble PAHs (4 to 7 rings) are carcinogenic, mutagenic, or teratogenic to a wide variety of organisms.</p> <p>In aquatic media, most aquatic organisms accumulate PAHs from low concentrations in the ambient media, with varying rates depending on the specific organism, specific PAH compound(s), and the amount of organic matter present. PAHs in surface water and sediment have been linked to liver tumors in fish, as well as inhibited reproduction, delayed emergence of larvae, decreased respiration and heart rate, and various effects on other organs of aquatic invertebrates (Eisler, 1987).</p> <p>PAH toxicity to soil invertebrates has been demonstrated by two recent studies. In a bioremediation study using an initial soil concentration of 4500 mg/kg total PAH, tests using 100% contaminated soil exhibited 100% mortality to earthworms (<i>Eisenia foetida</i>) at test initiation (Hund and Traunspurger, 1994). In another study, earthworm toxicity tests were conducted on PAH-contaminated soils from four manufactured gas plants. LC₅₀ values ranged from 129 to 3700 mg/kg total PAH (Loehr, et al., 1995).</p> <p>Limited data are available on the toxicological effects of PAHs on amphibians and reptiles. Those cited in Eisler (1987) were laboratory studies based on PAH crystal implantation and intraperitoneal injection of pure compound</p>

	<p>The major effect to avian wildlife cited by Eisler (1987) were embryotoxic effects of PAHs applied to the surface of mallard eggs in a synthetic petroleum mixture. As an example of reported effects, 0.002 ug/egg of 7,12-dimethylbenz(a)anthracene caused 26% mortality, and in survivors, produced significant reduction in embryonic growth and numerous other anomalies.</p> <p>PAHs are known for their ability to produce malignant tumors in skin and most epithelial tissues of many mammalian species. Eisler (1987) extensively reports on latency periods, target organs and tissues, and cellular transformation for specific PAHs. Examples of dietary doses of various PAHs causing carcinogenic effects range from 0.00004 – 3300 mg/kg body weight.</p> <p>Phytotoxic effects of petroleum hydrocarbons have been recognized since the early 1900s and include effects on plant photosynthesis, respiration, transpiration, seed germination, root elongation, shoot weight, plant height, and cellular changes. Recent plant bioassays indicated total PAH soil concentrations at 5880 mg/kg to be highly toxic with no seed germination occurring (Baud-Grasset, et al., 1993); 4500 mg/kg total PAH in soil showed repressed plant growth in two species tested (Hund and Traunspurger, 1994).</p> <p>PAHs show little tendency to biomagnify in food chains despite their high lipid solubility, probably because most PAHs are rapidly metabolized (Eisler, 1987).</p>
<p>Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function</p>	<p>The impacts considered are biological integrity and biodiversity. PAH levels elevated beyond natural background may cause acute or chronic toxicity, leading to changes in the composition, diversity, and function of normal plant and animal populations and communities.</p>
<p>Key impacts selected (critical ecological effects)</p>	<p>Key impacts evaluated are biological integrity and biodiversity. These impacts are evaluated via the comparison of PAH analytical data from various media with ecotoxicologically-based “Estimated Lowest Chronic Value For All Organisms” or USEPA Secondary Chronic Values (Suter and Tsao, 1996; NJDEP SWQS or NAWQC not available), sediment quality guidelines, and soil benchmarks.</p>
<p>Exposure Assessment</p>	
<p>Exposure routes and pathways considered</p>	<p>Exposure pathways result from natural background and diffuse anthropogenic sources as well as localized contamination of environmental media from industrial sites, petroleum releases, etc. Soil contamination is caused by direct placement of hazardous waste, discharges or spills, and the presence of historic fill containing PAH residues. Aquatic contamination is via point or non-point source discharges, including wastewater treatment effluents, surface water runoff or direct discharge from contaminated property, discharge of contaminated groundwater to surface water and/or wetlands, leaching from contaminated fill, and petroleum spills. Atmospheric deposition can affect all media.</p> <p>The primary exposure route for PAHs in terrestrial and aquatic ecosystems is ingestion (dietary, incidental soil ingestion, and ingestion via preening following petroleum spills). Exposure media include soil, surface water, sediment, and food.</p>

Population(s)/ecosystem(s) exposed statewide	<p>All ecosystems in the state are exposed to PAHs due to their natural occurrence and diffuse anthropogenic sources. Anthropogenic inputs have resulted in elevated PAH levels in many surface water bodies and terrestrial settings, especially in urban/industrial areas.</p>
Quantification of exposure levels statewide	<p>Statewide exposure to ambient/natural background and enriched levels of PAHs are discussed as follows. Concentrations for solid matrices are reported as mg/kg, dry weight.</p> <p>Ambient PAH Exposure Levels</p> <p>Ambient concentrations of PAH compounds in environmental media result from natural sources such as forest fires, microbial synthesis, and volcanic activity, as well as from a wide variety of diffuse anthropogenic sources, such as atmospheric deposition from by-products of combustion and petroleum releases.</p> <p>Ambient levels of 15 PAH compounds were measured in 11 river basins (22 stations) throughout the state and presented in Water Resources Data, New Jersey (USGS 1998). Surface water concentrations were not measured. With two exceptions, all data were less than 1.0 mg/kg. In the Raritan River Basin at Peter's Brook, PAH concentrations ranged from 0.03 mg/kg for Naphthalene to 5.1 mg/kg for Fluoranthene (total PAH = the total 24.87 mg/kg). In the Hudson River Basin, PAH concentrations ranged from 0.024 mg/kg for Acenaphthylene to 2.6 mg/kg for Fluoranthene (total PAH = 12.7 mg/kg) (USGS 1998).</p> <p>New Jersey ambient soil levels (mg/kg) measured in the Coastal Plain Province for selected carcinogenic PAHs are summarized as follows (BEM Systems, 1997):</p> <p>Benzo(a) anthracene: Mean – 0.14 Range - 0.01 – 1.82</p> <p>Benzo(a)pyrene: Mean - 0.14 Range - 0.01 – 1.80</p> <p>Benzo(b)fluoranthene: Mean – 0.35 Range - 0.05 - 4.09</p> <p>Benzo (b+j+k)fluoranthene): Mean - 0.35 Range- 0.05 – 4.10</p> <p>Dibenzo(a,h)anthracene: Mean – 0.03 Range – 0.01 - .29</p>

Indeno(1,2,3-c,d)pyrene:
Mean - 0.07
Range- 0.01-0.57

Chrysene: Mean - 0.15
Range -0.01 - 1.78

New Jersey ambient soil levels (mg/kg) measured in the Urban Piedmont Province for selected carcinogenic PAHs are summarized as follows (Wong and Saunders, 1998):

Benzo(a)Anthracene: Mean- 0.60
Range-0.05-5.20

Benzo(a)Pyrene: Mean - 0.61
Range-0.04-6.5

Benzo(b)fluoranthene: Mean-0.72
Range-0.05-9.4

Benzo(k)fluoranthene: Mean-0.58
Range-0.06-7.3

Dibenz(a,h)anthracene: Mean- 0.25
Range-0.04-1.5

Indeno(1,2,3-cd)pyrene: Mean-0.27
Range-0.05-2.9

Benchmark Values

Sediment – Fresh water, LEL, mg/kg dry weight (Persaud et al., 1993)

Acenaphthene: see “marine/estuarine”
value

Acenaphthylene: see “marine/estuarine”
Value

Anthracene: 0.220

Benzo(a)anthracene: 0.320

Benzo(k)fluoranthene: 0.240

Benzo(g,h,i)perylene: 0.170

Benzo(a)pyrene: 0.370

Chrysene: 0.340

Dibenzo(a,h)anthracene: 0.060

Fluoranthene: 0.750

Fluorene: 0.190

Indeno(1,2,3-cd)pyrene: 0.200

2-methylnaphthalene: see “marine/estuarine” value

Naphthalene: see “marine/estuarine”
Value

Phenanthrene: 0.560

Pyrene: 0.490

Total PAH; 4.0

Sediment – Marine/Estuarine, ER-L, mg/kg dry weight (Long et al., 1995)

Acenaphthene; 0.016

Acenaphthylene; 0.044

Anthracene; 0.085

B enzo(a)anthracene; 0.261

Benzo(k)fluoranthene; see “fresh water” value

Benzo(g,h,i)perylene: see fresh water value

Benzo(a)pyrene: 0.43

Chrysene: 0.384

Dibenzo(a,h)anthracene: 0.063

Fluoranthene: 0.600

Fluorene: 0.019

Indeno (1,2,3-cd) pyrene: see “fresh water” value

2-methylnaphthalene: 0.070

Naphthalene: 0.16

Phenanthrene: 0.240

Pyrene: 0.665

Surface Water

	<p>“Estimated Lowest Chronic Value for All Organisms” except where noted, µg/l (Suter and Tsao, 1996)</p> <p>Acenaphthene – 74 Anthracene – 0.09 Benzo(a)anthracene – 0.65 Benzo(a)pyrene – 0.3 Fluoranthene – 15 Fluorene – 3.9 (Secondary Chronic Value) Naphthalene – 620 Phenanthrene – 200</p> <p>Soil:</p> <p>Acenaphthene – PRG: 20 mg/kg, plant toxicity (Efroymson, et al., 1997) Fluorene – LOEC range: 170 – 206 mg/kg OECD soil (Efroymson, et al., 1997)</p> <p>Specific population(s) at increased risk</p> <p>The benthic macroinvertebrate community in surface water bodies in urban/industrial settings or adjacent to contaminated sites or other historic point or non point-sources are at increased risk. These surface water systems may show sediment PAH levels greater than the state-wide ambient exposure levels. Plant and animal communities in the vicinity of historic fill may be at increased risk.</p>
<p>Quantification of exposure levels to population(s) at increased risk</p>	<p>PAH Exposure Levels Associated with Contaminated Sites and Urban/Industrial Settings</p> <p>For inland fresh water and fresh water wetlands, data associated with contaminated sites are used to represent exposure of elevated PAH levels to aquatic receptors. Statewide enriched PAH exposure levels were estimated from the average or 95% upper confidence level (UCL) sediment concentrations associated with a limited number of sites representing these ecosystems. Surface water is not typically analyzed for all sites, likely due to the low solubility of PAH compounds. The sites chosen for this evaluation have undergone a Baseline Ecological Evaluation (BEE) or full Ecological Risk Assessment, pursuant to N.J.A.C.7:26 E-3.11 and 4.7, with data generated by certified laboratories and validated in accordance with USEPA and State protocols. While remedial actions may have been performed at some of these sites, the data are judged to be good estimates of likely exposure levels caused by contaminated sites statewide. Average or 95% UCL PAH sediment concentrations, mg/kg dry weight, associated with contaminated sites are as follows;</p> <p>Inland Fresh Water Sediments: four sites, data summaries attached. Of these, two sites evaluated surface water for PAHs and concentrations for all 15 PAHs were “ND” in both cases. Results expressed as mg/kg, dry weight.</p> <p>Acenaphthene – 6.49 Acenaphthylene – 1.40</p>

	<p>Anthracene – 4.26 Benzo(a)anthracene – 8.33 Benzo(k)fluoranthene - 5.00 Benzo(g,h,i)perylene- 3.4 Benzo(a)pyrene – 7.22 Chrysene – 8.1 Dibenzo(a,h)anthracene – 1.28 Fluoranthene – 16.88 Fluorene – 4.39 Indeno(1,2,3-cd)pyrene– 3.75 2-methylnaphthalene – ND Naphthalene – 14.12 Phenanthrene – 14.88 Pyrene – 12.10</p> <p>Fresh Water Wetlands: one site out of four examined had PAH analyses; data summary attached. Results expressed as mg/kg, dry weight.</p> <p>Pyrene – 3.23 (0.07 – 16.00)</p>
	<p>Acenaphthene – ND Acenaphthylene – ND Anthracene – ND Benzo(a)anthracene – 0.30 Benzo(k)fluoranthene - 0.37 Benzo(g,h,i)perylene- ND Benzo(a)pyrene – 0.34 Chrysene – 0.34 Dibenzo(a,h)anthracene – ND Fluoranthene – 0.65 Fluorene – ND Indeno(1,2,3-cd)pyrene– ND 2-methylnaphthalene – ND Naphthalene – ND Phenanthrene – 0.35 Pyrene – 0.68</p> <p>Estuarine ecosystems were evaluated on the basis of individual large rivers, due to the availability of comprehensive sediment databases. Data associated with specific contaminated sites are included where available for additional information. Concentrations presented are mean values, mg/kg dry weight, for individual PAHs except where indicated.</p>

	<p>Estuarine Ecosystems:</p> <p>1. Newark Bay - 28 stations, includes samples in the Arthur Kill, Passaic River, and Hackensack River, mg/kg dry weight (Adams, et al., 1998):</p> <p>Acenaphthene – 0.092 Acenaphthylene – 0.20 Anthracene – 0.51 Benzo(a)anthracene – 0.91 Benzo(k)fluoranthene - 2.11 Benzo(g,h,i)perylene- 0.58 Benzo(a)pyrene – 0.52 Chrysene – 1.08 Dibenzo(a,h)anthracene – 0.15 Fluoranthene – 1.28 Fluorene – 0.11 Indeno(1,2,3-cd)pyrene– 0.58 2-methylnaphthalene – 0.11 Naphthalene – 0.22 Phenanthrene – 0.42 Pyrene – 1.14 TOTAL PAH: 11.47</p>
	<p>2. Lower Harbor -28 sample stations; includes Raritan and Sandy Hook Bays, mg/kg dry weight (Adams et al., 1998)</p> <p>Acenaphthene – 0.017 Acenaphthylene – 0.04 Anthracene – 0.06 Benzo(a)anthracene – 0.14 Benzo(k)fluoranthene - 0.29 Benzo(g,h,i)perylene- 0.11 Benzo(a)pyrene – 0.11 Chrysene – 0.16 Dibenzo(a,h)anthracene – 0.02 Fluoranthene – 0.20 Fluorene – 0.03 Indeno(1,2,3-cd) pyrene– 0.12 2-methylnaphthalene – 0.03 Naphthalene – 0.05 Phenanthrene – 0.12</p>

	<p>Pyrene – 0.20 TOTAL PAH: 2.18</p> <p>3. Passaic River: (NOAA, 2000) Mean value, mg/kg dry weight, from 9 databases representing Passaic River studies)</p> <p>Acenaphthene – 3.92 Acenaphthylene – 1.28 Anthracene – 2.92 Benzo(a)anthracene – 3.07 Benzo(k)fluoranthene - 2.32 Benzo(g,h,i)perylene- 1.71 Benzo(a)pyrene – 2.93 Chrysene – 3.2 Dibenzo(a,h)anthracene – 1.38 Fluoranthene – 6.16 Fluorene – 2.14 Indeno(1,2,3-cd)pyrene– 1.75 2-methylnaphthalene – 1.54 Naphthalene – 1.92 Phenanthrene – 6.07 Pyrene – 6.06 TOTAL PAH: 32.30</p>
	<p>4. Hackensack River (Passaic River Study, 1996)</p> <p>Acenaphthene – ND Acenaphthylene – ND Anthracene – 0.49 Benzo(a)anthracene – 0.95 Benzo(k)fluoranthene - 0.54 Benzo(g,h,i)perylene- 0.69 Benzo(a)pyrene – 0.76 Chrysene – 0.61 Dibenzo(a,h)anthracene – ND Fluoranthene – 0.90 Fluorene – ND Indeno(1,2,3-cd)pyrene– 0.63 2-methylnaphthalene – ND Naphthalene – 0.46</p>

Phenanthrene – 0.48

Pyrene – 1.30

5. Elizabeth River (Passaic River Study, 1996) mg/kg, dry weight

Acenaphthene – ND

Acenaphthylene – ND

Anthracene – 1.4

Benzo(a)anthracene – 4.6

Benzo(k)fluoranthene - 3.4

Benzo(g,h,i)perylene- 4.5

Benzo(a)pyrene – 5.9

Chrysene – 5.4

Dibenzo(a,h)anthracene – ND

Fluoranthene – 5.9

Fluorene – ND

Indeno(1,2,3-cd)pyrene– 3.9

2-methylnaphthalene – ND

Naphthalene – ND

Phenanthrene – 4.6

Pyrene – 8.7

6. Lower Delaware River:

(Eisle, 1995) means based on between 9-34 samples, mg/kg dry weight)

Acenaphthene – 0.015

Acenaphthylene – 0.009

Anthracene – 0.03

Benzo(a)anthracene – 0.09

Benzo(k)fluoranthene - Not available

Benzo(g,h,i)perylene- 0.13

Benzo(a)pyrene – 0.13

Chrysene – 0.12

Dibenzo(a,h)anthracene – 0.04

Fluoranthene – 0.13

Fluorene – 0.05

Indeno(1,2,3-cd)pyrene– 0.13

2-methylnaphthalene – 0.05

Naphthalene – 0.04

Phenanthrene – 0.09

	<p>Pyrene – 0.13</p> <p>7. Lower Raritan River (representative data from the Horseshoe Road Superfund Site, based on mean of 7 Raritan River sediment samples, mg/kg dry weight (CDM Federal Programs Corp., 1999).</p> <p>Acenaphthene – 1.01 Acenaphthylene – 1.01 Anthracene – 0.95 Benzo(a)anthracene – 0.61 Benzo(k)fluoranthene - 0.96 Benzo(g,h,i)perylene- 0.94 Benzo(a)pyrene – 0.53 Chrysene – 0.85 Dibenzo(a,h)anthracene – 1.01 Fluoranthene – 0.50 Fluorene – 1.01 Indeno(1,2,3-cd)pyrene– 0.938 2-methylnaphthalene – 1.01 Naphthalene – 1.01 Phenanthrene – 0.92 Pyrene – 0.55</p>
	<p>8. Atlantic Coastal Water (Eisle, 1995) mean based on 2 samples, mg/kg dry weight</p>
Dose/Impact-Response Assessment	

Quantitative impact-assessment employed

Statewide-enriched surface sediment concentrations of PAHs were compared to sediment screening criteria to assess the risk to the aquatic benthic community. The selection of specific PAHs for comparison was based on the availability of screening criteria. Surface water data were not evaluated, due to the extremely limited data available (only limited inland freshwater data, largely reported as “non-detects”). Soil levels of Acenaphthene and Fluorene were compared to those two available benchmarks. The hazard quotient (HQ) method was used to conduct a screening level risk assessment where:

$$\text{HQ} = \text{Estimated Environmental Concentration} / \text{Benchmark Concentration}$$

This approach assumes that concentrations of the contaminant were representative of the exposure to biota in the three ecosystems evaluated. HQ values <1 indicate PAH concentrations are at a level where adverse effects are not expected; HQ values >1 indicate the potential for adverse ecological effects.

Risk Characterization

Risk estimate(s) by population at risk	PAHs ASSOCIATED WITH CONTAMINATED SITES AND/OR ENRICHED URBAN/INDUSTRIAL SETTINGS	Score
	INLAND FRESH WATER SEDIMENTS: HAZARD QUOTIENTS	
	Acenaphthene: $6.49 / 0.5 = 12.98$	4
	Acenaphthylene: $1.40 / 0.64 = 2.19$	2
	Anthracene: $4.26 / 0.22 = 19.36$	4
	Benzo(a)anthracene: $8.33 / 0.32 = 26.08$	5
	Benzo(k)fluoranthene: $5.00 / 0.24 = 20.83$	4
	Benzo(g,h,i)perylene: $3.4 / 0.17 = 20$	4
	Benzo(a)pyrene – $7.22 / 0.37 = 19.51$	4
	Chrysene: $8.1 / 0.34 = 23.82$	4
	Dibenzo(a,h)anthracene: $1.28 / 0.06 = 21.33$	4
	Fluoranthene: $16.88 / 0.75 = 22.51$	4
	Fluorene: $4.39 / 0.19 = 23.11$	4
	Indeno(1,2,3-cd)pyrene: $3.75 / 0.20 = 18.75$ 2-methylnaphthalene – ND	4 1
	Naphthalene: $14.12 / 2.1 = 6.72$	3
	Phenanthrene: $14.88 / 0.56 = 26.57$	5
	Pyrene: $12.10 / 0.49 = 24.69$	4
	FRESH WATER WETLANDS: HAZARD QUOTIENTS	
	Acenaphthene – ND	1
	Acenaphthylene – ND	1
	Anthracene – ND	1
	Benzo(a)anthracene: $0.30 / 0.32 = 0.94$	1
	Benzo(k)fluoranthene: $0.37 / 0.24 = 1.54$	2
	Benzo(g,h,i)perylene- ND	1
	Benzo(a)pyrene: $0.34 / 0.37 = 0.92$	1
	Chrysene: $0.34 / 0.34 = 1.0$	2
	Dibenzo(a,h)anthracene – ND	1
	Fluoranthene: $0.65 / 0.75 = 0.87$	1
	Fluorene – ND	1
	Indeno(1,2,3-cd)pyrene – ND	1
	2-methylnaphthalene – ND	1
	Naphthalene – ND	1
	Phenanthrene: $0.35 / 0.56 = 0.63$	1
	Pyrene: $0.68 / 0.49 = 1.39$	2

ESTUARINE ECOSYSTEMS: HAZARD QUOTIENTS		
Newark Bay:		
Acenaphthene: $0.092/0.016 = 5.75$		3
Acenaphthylene: $0.20/.044 = 4.55$		2
Anthracene: $0.51/0.085 = 6.00$		3
Benzo (a) anthracene: $0.91/.261 = 3.49$		2
Benzo (k) fluoranthene: $2.11/0.24 = 8.79$		3
Benzo (g,h,i) perylene: $0.58/0.17 = 3.41$		2
Benzo (a) pyrene: $0.52/0.43 = 1.16$		2
Chrysene: $1.08/0.384 = 2.81$		2
Dibenzo(a,h)anthracene: $0.15/.063 = 2.38$		2
Fluoranthene: $1.28/0.60 = 2.13$		2
Fluorene: $0.11/0.019 = 5.79$		3
Indeno(1,2,3-cd)pyrene: $0.58/0.2 = 2.9$		2
2-methylnaphthalene: $0.11/0.07 = 1.57$		2
Naphthalene: $0.22/0.16 = 1.38$		2
Phenanthrene: $0.42/0.24 = 1.75$		2
Pyrene: $1.14/0.665 = 1.71$		2
TOTAL PAH: $11.47/4 = 2.87$		2
Passaic River		
Acenaphthene – $3.92/.016 = 245$		2
Acenaphthylene – $1.28/.044 = 29$		1
Anthracene – $2.92/.085 = 34$		1
Benzo(a)anthracene – $3.07/.261 = 11.76$		1
Benzo(k)fluoranthene - $2.32/0.24 = 10.05$		2
Benzo(a)pyrene – $2.93/0.43 = 6.8$		1
Chrysene – $3.2/0.384 = 8.33$		1
Dibenzo(a,h)anthracene – $1.38/0.063 = 21.90$		1
Fluoranthene – $6.16/0.60 = 10.27$		1
Fluorene – $2.14/0.019 = 112$		1
Indeno(1,2,3-cd)pyrene– $1.75/0.2 = 8.75$		2
2-methylnaphthalene – $1.54/0.07 = 22.0$		1
Naphthalene – $1.92/0.16 = 12$		1
Phenanthrene – $6.07/0.24 = 25.29$		1
Pyrene – $6.06/0.665 = 9.11$		1
		1
		1

4. Hackensack River	
Acenaphthene – ND	5
Acenaphthylene – ND	5
Anthracene – $0.49/0.085 = 5.76$	5
Benzo(a)anthracene – $0.95/0.26 = 3.65$	4
Benzo(k)fluoranthene - $0.54/0.24 = 2.25$	3
Benzo(g,h,I)perylene- $0.69/0.17 = 4.06$	3
Benzo(a)pyrene – $0.76/0.43 = 1.77$	3
Chrysene – $0.61/0.384 = 1.59$	3
Dibenzo(a,h)anthracene – ND	4
Fluoranthene – $0.90/0.60 = 1.5$	3
Fluorene – ND	5
Indeno(1,2,3-cd)pyrene– $0.63/0.20 = 3.15$	3
2-methylnaphthalene – ND 9.67	4
Benzo(g,h,I)perylene- $1.71/0.17 =$	4
Naphthalene – $0.46/0.16 = 2.88$	4
Phenanthrene – $0.48/0.24 = 2.0$	3
Pyrene – $1.30/0.665 = 1.95$	
5. Elizabeth River	
Acenaphthene – ND	1
Acenaphthylene – ND	1
Anthracene – $1.4/0.085 = 16.5$	3
Benzo(a)anthracene – $4.6/0.261 = 17.6$	2
Benzo(k)fluoranthene - $3.4/0.24 = 9.67$	2
Benzo(g,h,I)perylene- $4.5/0.17 = 26.5$	2
Benzo(a)pyrene – $5.9/0.43 = 13.7$	2
Chrysene – $5.4/0.384 = 14.1$	2
Dibenzo(a,h)anthracene – ND	1
Fluoranthene – $5.9/0.60 = 9.8$	2
Fluorene – ND	1
Indeno(1,2,3-cd)pyrene– $3.9/0.2 = 19.5$	2
2-methylnaphthalene – ND	1
Naphthalene – ND	2
Phenanthrene – $4.6/0.24 = 19.1$	2
Pyrene – $8.7/0.665 = 13.1$	2

Lower Delaware River:		
Acenaphthene – 0.015/0.016 = 0.93		1
Acenaphthylene – 0.009/0.044 = 0.20		1
Anthracene – 0.03/.085 = 0.35		4
Benzo (a) anthracene – 0.09/0.261 = 0.35		4
Benzo (k) fluoranthene - Not available		3
Benzo (g,h,i) perylene- 0.13/0.17= 0.76		5
Benzo (a) pyrene – 0.13/0.43 = 0.30		4
Chrysene – 0.12/0.384= 0.31		4
Dibenzo (a,h) anthracene –0.04/0.063=0.63		1
Fluoranthene – 0.13/0.60 = 0.21		3
Fluorene – 0.05/0.019 = 2.23		1
Indeno (1,2,3-cd) pyrene– 0.13/0.2 = 0.65		4
2-methylnaphthalene – 0.05/0.07 = 0.71		1
Naphthalene – 0.04/0.16 = 0.25		1
Phenanthrene – 0.09/0.24 = 0.38		4
Pyrene – 0.13/0.665 = 0.20		4
Lower Raritan River		
Acenaphthene – 1.01/0.016 = 63		1
Acenaphthylene – 1.01/0.044 = 22.95		1
Anthracene – 0.95/0.085 = 11.18		1
Benzo (a) anthracene – 0.61/0.261 = 2.33		1
Benzo (k) fluoranthene – 0.96/0.24 = 4.0		1
Benzo (g,h,i) perylene- 0.94/0.17 = 5.53		1
Benzo (a) pyrene – 0.53/0.43 = 1.23		1
Chrysene – 0.85/0.384 = 2.21		1
Dibenzo (a,h) anthracene – 1.01/0.063 = 16.03		1
Fluoranthene – 0.50/0.60 = 0.83		1
Fluorene – 1.01/0.019 = 53.2		2
Indeno (1,2,3-cd) pyrene–0.938/0.2 = 4.7		1
2-methylnaphthalene – 1.01/ 0.07 = 14.4		1
Naphthalene – 1.01/0.16 = 6.3		1
Phenanthrene – 0.92/0.24 = 3.8		1
Pyrene – 0.55/0.665 = 0.83		1

Assessment of severity/irreversibility	8. Atlantic Coastal Water		
	Acenaphthene – $0.01/0.016 = 0.6$	5	
	Acenaphthylene – $0.01/0.44 = 0.23$	4	
	Anthracene – $0.05/0.085 = 0.59$	4	
	Benzo(a)anthracene – $0.09/0.261 = 0.34$	2	
	Benzo(k)fluoranthene - Not available	2	
	Benzo(g,h,I)perylene- $0.16/0.17= 0.94$	3	
	Benzo(a)pyrene – $0.13/0.43 = 0.30$	2	
	Chrysene – $0.09/0.384 = 0.23$	2	
	Dibenzo(a,h)anthracene – $0.02/0.063= 0.32$	4	
	Fluoranthene – $0.27/0.60 = 0.45$	1	
	Fluorene – $0.02/0.019 = 1.05$	5	
	Indeno(1,2,3-cd)pyrene– $0.18/0.2 = 0.9$	2	
	2-methylnaphthalene – $0.03/0.07= 0.42$	4	
	Naphthalene – $0.03/0.16 = 0.19$	3	
	Phenanthrene – $0.14/0.24 = 0.58$	2	
	Pyrene – $0.43/0.665 = 0.65$	1	
	URBAN/TERRESTRIAL ECOSYSTEM – LIBERTY STATE PARK AREAS OF HISTORIC FILL: HAZARD QUOTIENTS (Evaluated maximum of range against two available soil screening values)		
	Acenaphthene – $0.39/20 = 0.02$	1	
	Fluorene – $1.7/170 = 0.01$	1	
	Lower Harbor Sediments: Acenaphthene: $0.017/.016 = 1.06$ Acenaphthylene: $0.04/.044 = 0.9$		
	The following scale was used when assigning a score to the generated HQs:		

Issue: Polycyclic Aromatic Hydrocarbons (PAH)

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	HQ	Score
	<1	1
	1-5	2
	3	
	11-25	4
	25+	5
Assessment of frequency of effect(s)	See attached table <i>Statewide analysis of Threat</i>	
Size of population(s) and/or extent of the State/habitat affected (magnitude)	See attached table <i>Statewide analysis of Threat</i>	
	Total	
Assessment of uncertainties in this assessment (H,M,L) and brief description	There is moderate-high uncertainty with the freshwater and terrestrial portions of this assessment due to limited sediment, surface water, and soil data availability, lack of comprehensive database for management of contaminated site data, and potential for elevated analytical detection limits due to matrix interference. There is lower uncertainty associated with the estuarine portion of this assessment due to availability of comprehensive data sets from regulatory agencies (e.g., USEPA, NOAA, DRBC). There is particularly low uncertainty in the Passaic River due to multiple phases of the Passaic River Study, which has comprehensively characterized sediments downstream of the Dundee Dam.	
Potential for additional data to result in a significant future change in this risk estimate (H,M,L) and brief description. (Data Gaps; highlight significant data needs)	Additional data and data management strategies for environmentally sensitive areas in the State which are affected by diffuse anthropogenic PAH sources and historic fill containing PAH residues would better characterize risk from this threat. Ecotoxicologically-based soil screening benchmarks for additional PAHs of concern would facilitate risk estimations for this contaminant. Data from the Upper Delaware watershed are limited.	
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, -, =, =, where + is improvement), and brief description.	The potential for significant future change in the underlying risk from PAH is rated (0). While long term improvement is expected due to source controls and remedial actions at contaminated sites, risk is expected to remain stable for a 5-10- year timeframe, due to technical and administrative complexities associated with remedial investigations and remedial actions, especially in areas enriched by diffuse anthropogenic sources and historic fill.	
Potential for catastrophic impacts (H,M,L) and brief description	L: The potential for catastrophic impacts to NJ's ecosystems is considered to be low, based on the regulations in place to control contaminant discharges and require clean-ups, and current knowledge of the ecotoxicology of PAH compounds.	
Link to other Work Groups (e.g., socioeconomic impacts)	Potential socioeconomic impacts of PAH contamination include costs associated with source controls, contaminated site remediations, and natural resources damage assessments.	
Extent to which threat is currently regulated	Control of PAH discharges and the remediation of PAH-contaminated hazardous waste sites are regulated under the Industrial Site Recovery Act (ISRA), Spill Compensation and Control Act, Solid Waste Management Act (SWMA), Water Pollution Control Act (WPCA), Resource Conservation and Recovery Act (RCRA), Comprehensive Environmental Response, Compensation and Liability Act of 1980, as amended by Superfund Amendments and Reauthorization Act of 2986 (CERCLA) and the Hazardous Site Discharge Remediation Act.	
Barriers to restoration	Presence of historic fill containing PAH residues; technical and economic complexities associated with remedial activities	

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Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	H: PAHs are products of incomplete combustion, and are present in industrial chemical wastes such as coal tar, petroleum refinery sludges, waste oils and fuels, and wood treating residues; heat and power generation; controlled refuse incineration.
Small business industry	M
Transportation	H: internal combustion engines
Residential	L: open burning
Agriculture	L
Recreation	L
Resource extraction	H: Petroleum refining
Government	L
Natural sources/processes	H Microbial synthesis, volcanic activity, forest and prairie fires.
Orphan contaminated sites	
Diffuse Sources	
Sediment sinks	H: in areas of known contaminated waste sites and/or urban/industrial areas
Soil sinks	H: in areas of petroleum spills; contaminated fill
Non-local air sources incl. Deposition	H: 43,000metric tons of PAHs discharged into atmosphere each year (Eisler, 1987)
Biota sinks	L: Little evidence of food chain biomagnification

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Statewide Analysis of Threat

Threat = Polycyclic Aromatic Hydrocarbons

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	3	3	2	18
Marine Waters	3	3	2	18
Wetlands	3	3	2	18
Forests	2	2	1	4
Grasslands	1	1	1	1
Total Score				59
Average Score				11.8

Risk by Watershed Management Region

THREAT = PAHs	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L	NA	L	L	L
Passaic	M-H	H	H	L	L
Raritan	M-H	M-H	M-H	L	L
Atlantic	L	L	L	L	L
Lower Delaware	L	L	L	L	L
Region/Watershed (secondary)					
Urban	M-H	M-H	M-H	NA	NA
Suburban	L	L	L	L	L
Rural	L	L	L	L	L

H=high, M=medium, L=low;



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New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	QPX Parasite in Shellfish (Quahog Parasite X)
Description of stressor	Primitive member of the Phylum Labyrinthomorpha (Kingdom Protista)(Pokorny 1985); one of the “slime-net” protists. QPX is a common saprophytic organisms in marine/estuarine environments (Ford, in press; Porter 1990); usually found in high salinities (c. 30 ppt)(Ford, in press; Kraeuter, HSRL, pers. comm.). Lower temperature may be important in the seasonal pattern (Ford, in press).
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	Biological integrity: Associated with mortalities in cultured and natural populations of hard clams (Ford, in press).
Key impacts selected (critical ecological effects)	Mortalities in cultured and natural populations of hard clams (Ford, in press).
Exposure Assessment	
Exposure routes and pathways considered	In Massachusetts, studies seem to show that clams become infected in planting areas after 2 yrs and followed by mortalities; seed clams were not likely source of QPX (Ford, in press).
Population(s)/ecosystem(s) exposed statewide	New Jersey: QPX was first discovered in the 1950s in hard clams in a Prince Edward Island, Canadian hatchery (Whyte et al., 1994). In 1976, a similar organism to the QPX, was found in dead and dying hard clams collected in Barnegat Bay (Atlantic coast region, WMA #13) in early Dec. after a severe early freeze (Ford, in press; unpublished report by T. Keller, HSRL, 1977). Subsequent surveys in 1996 and 1997 found the QPX in New Jersey (Kraeuter et al.) 1998) and Virginia where high prevalence was associated with mortality in some locations (Ford, in press).
Quantification of exposure levels statewide	Not known
Specific population(s) at increased risk	Cultured hard clams in the field, <i>Mercenaria mercenaria</i>
Quantification of exposure levels to population(s) at increased risk	Evidence suggests that infections of QPX and subsequent mortalities occurred in molluscs growing under conditions fostering proliferation of such pathogens in stressed hosts (e.g., high density cultures or natural beds, low temperature, maladapted stocks, suboptimal growing or holding conditions)(Ford, in press).
Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	Clams with moderate to severe infections of QPX had lower growth and meat condition compared to those with light or no detectable infections (Ford, in press; Smolowitz et al.) 1998).

	<p>Heavily infected and dead clams were found at the surface, growing in coarsely grained sandy sediments, and had marked chipping of the shell margin (Ford, in press) – but this symptom was not found in infected clams growing in a finer grained sediment.</p> <p>Inflammatory response: Retracted, swollen mantle edge (with nodules)(Ford, in press; Smolowitz et al. 1998). Clams in U.S. sites have infected mantle and gills (Ford, in press; Ragone Calvo et al.) 1998; Smolowitz et al. 1998). Clams in New Jersey and Virginia had parasites appearing moribund while this was not true of clams in CA and MA studies (Ford, in press).</p>	
Risk Characterization		
<p>Risk estimate(s) by population at risk</p> <p>Assessment of severity/irreversibility Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage: • many species threatened/endangered • major community change • extensive loss of habitats/species Long time for recovery</p> <p>3 - Adverse affect on structure and function of system: • all habitats intact and functioning • population abundance and distributions reduced Short time for recovery</p> <p>2 - Ecosystem exposed but structure and function hardly affected</p> <p>1 - No detectable exposure</p>	<p>Infections may become increasingly severe with time (Ford, in press); Clams associated with fine-grained sediments were not showing shell chipping or moribundity as opposed to clams in coarser grained sandy sediments; QPX parasites frequently appear moribund in NJ (Ford, in press); Irreversibility not known except environment may influence severity.</p>	<p>Score</p> <p>2</p>

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<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade)</p> <p>5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)</p>	<p>Varies in different states.</p>	<p>3</p>
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude)</p> <p>5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>Estuarine waters: Hard clam populations in Barnegat Bay Estuary but potentially, maybe other estuaries.</p>	<p>1</p>
	<p>State: Average Score Total</p>	<p>0.75</p>
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>High at this time; many uncertainties remain regarding the aspects of QPX disease (Ford, in press).</p>	
<p>Potential for additional data to result in a significant future change in this risk estimate (H,M,L) and a brief description</p>	<p>(High) 1. More clarification of epizootiological aspects of the disease is needed (Ford, in press), esp. relating to environmental factors (e.g., temperature, seasonal differences). Natural populations need to be studied in addition to cultured populations of hard clams. A determination of the QPX ecological role (e.g., facultative or opportunistic pathogen) is needed.</p>	
<p>Potential for future change in the underlying risk from this stressor (+++, ++, +, 0, -, =, ≡) and brief description</p>	<p>(0) Potential impacts from change in underlying risk do not seem high based on history. QPX was first found more than 25 yrs. ago, but has not caused a major problem. Good animal husbandry should greatly minimize problems.</p>	
<p>Potential for catastrophic impacts* (H,M,L) and brief description</p> <p>(*Short-term drastic negative impacts having widespread geographic scope)</p>	<p>Medium to low: even though mortalities have been observed, there needs to be further quantification of effects.</p>	
<p>Link to other Work Groups (e.g., socioeconomic impacts)</p>	<p>Socioeconomic impacts of mortality to hard clam populations in Barnegat Bay and perhaps, other estuaries.</p>	
<p>Extent to which threat is currently regulated or otherwise managed</p>	<p>Specifically, QPX is not regulated. NJDEP's National Shellfish Sanitation Program surveys shellfish growing in waters in the state and classifies them according to the presence and abundance of coliform bacteria and significant sources of potential contamination. Water data are combined with land use, water hydrography and pollution source information to classify the NJ shellfish growing waters for harvesting.</p>	

Barriers to restoration	Not known
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	Not known
Large business/industry	
Small business industry	
Transportation	
Residential	
Agriculture	
Recreation	
Resource extraction	
Government	
Natural sources/processes	
Orphan contaminated sites	
Diffuse Sources	
Sediment sinks	
Soil sinks	
Non-local air sources incl. deposition	
Biota sinks	

Issue: QPX Parasite in Shellfish
Author: M. Gastrich
Version: 05/02/00

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Issue: QPX Parasite in Shellfish
 Author: M. Gastrich
 Version: 05/02/00

Statewide Analysis of Threat

Threat = QPX (Quahog Parasite Unknown)

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	N/A	N/A	N/A	
Marine Waters	2	3	1	6
Wetlands	N/A	N/A	N/A	
Forests	N/A	N/A	N/A	
Grasslands	N/A	N/A	N/A	
Total Score				6
Average Score				1.2

Risk by Watershed Management Region

THREAT = QPX Watershed Management Region	ECOSYSTEM				
	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	N/A	N/A	N/A	N/A	N/A
Passaic	N/A	N/A	N/A	N/A	N/A
Raritan	N/A	N/A	N/A	N/A	N/A
Atlantic	N/A	Medium		N/A	N/A
Lower Delaware	N/A	N/A	N/A	N/A	N/A
Region/Watershed (secondary)					
Urban	N/A	N/A	N/A	N/A	N/A
Suburban	N/A	Low	N/A	N/A	N/A
Rural	N/A	Low	N/A	N/A	N/A

New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings																																
Hazard Identification																																	
Stressor	Road Salt																																
Description of stressor	Road salts (sodium chloride, calcium chloride, potassium chloride and calcium magnesium acetate) applied as deicers have the potential to cause adverse effects to freshwater ecosystems. Road salts often also contain ferrocyanide salts that are added as an anti-caking agent (Environment Canada 2000). Primary sources of road salts entering the environment are runoff from storage facilities and application of these compounds to roadways. After the snow melts, these compounds enter soil, surface water and groundwater. In water these compounds dissociate into the chloride ion and the associated anion. Once concentrations of these ions are raised in freshwater systems, they may reach levels that can cause acute or chronic effects to aquatic biota, such as benthic macroinvertebrates, fish, reptiles and amphibians. Ferrocyanide salts added to road salt may also pose potential risks to birds ingesting it directly as a source of grit (Environment Canada 2000). Sparrows, finches and other granivores are known to do this during winter months.																																
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	<p><u>Biological Integrity</u> – <i>Freshwater Systems</i> - Road salts can potentially affect the biological integrity of freshwater systems if application levels and subsequent transport into water bodies results in concentrations exceeding tolerance levels of aquatic biota. The following concentrations of salts have been documented as causing deleterious effects to aquatic biota:</p> <table border="0"> <thead> <tr> <th style="text-align: left;"><i>Chloride</i> (mg/L)</th> <th style="text-align: left;"><i>Response</i></th> <th style="text-align: left;"><i>Organism</i></th> <th style="text-align: left;"><i>Reference</i></th> </tr> </thead> <tbody> <tr> <td>2,267</td> <td>LC100</td> <td>Oligochaetes (48 hr exposure)</td> <td>Hamilton et al. (1975)</td> </tr> <tr> <td>1,380</td> <td>LC100</td> <td>Chironomids (48 hr exposure)</td> <td>Hamilton et al. (1975)</td> </tr> <tr> <td>4,000</td> <td>LC50</td> <td>Trichoptera (72 hr exposure)</td> <td>Sutcliffe (1961)</td> </tr> <tr> <td>6151</td> <td>LC50</td> <td>Caddisfly</td> <td>Hamilton et al. (1975)</td> </tr> <tr> <td>7650</td> <td>LC50</td> <td>Fathead Minnow adult (96 hr exp.)</td> <td>Adelman et al. (1976).</td> </tr> <tr> <td>10,000</td> <td>NOEC</td> <td>Bluegill Sunfish adult (96 hr exp.)</td> <td>Kszos et al. (1990)</td> </tr> <tr> <td>12,000</td> <td>LC20</td> <td>Rainbow Trout</td> <td>SUNY ESF (1971)</td> </tr> </tbody> </table>	<i>Chloride</i> (mg/L)	<i>Response</i>	<i>Organism</i>	<i>Reference</i>	2,267	LC100	Oligochaetes (48 hr exposure)	Hamilton et al. (1975)	1,380	LC100	Chironomids (48 hr exposure)	Hamilton et al. (1975)	4,000	LC50	Trichoptera (72 hr exposure)	Sutcliffe (1961)	6151	LC50	Caddisfly	Hamilton et al. (1975)	7650	LC50	Fathead Minnow adult (96 hr exp.)	Adelman et al. (1976).	10,000	NOEC	Bluegill Sunfish adult (96 hr exp.)	Kszos et al. (1990)	12,000	LC20	Rainbow Trout	SUNY ESF (1971)
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	<p>In addition, the U.S. EPA chronic continuous criterion for protection of freshwater aquatic life from chloride is 230 mg/L, while the acute criterion for protection of aquatic life is 860 mg/L (U.S. EPA 1998).</p> <p><i>Roadside Vegetation-</i> Plants have shown effects in soil with chloride concentrations as low as 215 mg/kg chloride (Environment Canada 2000). However, effects of chloride concentrations in the field may be confounded with other factors influencing roadside vegetation communities, such as disturbance and soil types.</p> <p><i>Effects of Direct Ingestion by Birds-</i> A study of house sparrows fed granular sodium chloride reported an LD50 of 3181 mg/kg, with birds showing histopathological lesions such as gizzard edema at concentrations greater than 500 mg/kg (Hollinger et al. 1999).</p> <p><u>Biodiversity</u> No studies were located addressing potential impacts on biodiversity. Biodiversity is unlikely to be significantly affected unless application rates are very high, resulting in receiving water bodies having chloride concentrations higher than the toxicological effects levels of freshwater organisms (see “Exposure Estimate” below). The primary impacts on biodiversity are likely to be to local systems, particularly to lower organisms such as the benthic macroinvertebrate community in freshwater ecosystems.</p> <p>The vegetative diversity of roadside systems may also be affected by road salt concentrations in soil (Environment Canada 2000), although this has not been well studied, and is confounded by other factors, such as disturbance.</p> <p><u>Habitat/Ecosystem Health</u></p> <p>Primary impacts on habitat and ecosystem health are likely to occur when impacts from road salts are combined with other non-point source pollutants in stormwater runoff entering receiving water bodies, particularly in urban watersheds. A Canadian study (Mayer et al. 1999) found that impacts are likely to be greatest to ponds located adjacent to roadways, and creeks located within urban areas. Reduced diversity in the food chain, and dominance by more saline tolerant species may result in corresponding food chain imbalance due to effects on the forage base of higher receptors.</p> <p>Concentrations of road salts in soils adjacent to roadways or uncovered storage piles may also affect vegetation. Impacts are likely to be greatest when combined with other factors, such as disturbance and/or erosion favoring weedy invasive species.</p> <p><u>Ecosystem Function</u> High and persistent concentrations of chlorides (e.g. NaCl>2000 mg/L) may harm aquatic life via direct saline stress, as well as anoxic conditions caused by prolonged water stratification in lakes and ponds.</p>
Key impacts selected (critical ecological effects)	Toxicity to Freshwater Biota; Reduced Aquatic Diversity
Exposure Assessment	
Exposure routes and pathways considered	<p>Surface Water Runoff into Ponds, Lakes, Streams</p> <p>Leaching into Shallow Groundwater (from uncovered Storage Areas) and Subsequent Release into Surface Water</p> <p>Uptake of chlorides by roadside plants</p> <p>Direct ingestion of rock salt by birds</p>

Population(s)/ecosystem(s) exposed statewide	Areas along roadways, particularly adjacent water bodies (lakes, ponds, streams and rivers) statewide, but particularly in portions of the state receiving greater road salt application. These include areas next to highways and major traveled roadways, urban areas, and areas in the northern portion of the state receiving greater snowfall.	
Quantification of exposure levels statewide	Quantification of exposure levels statewide was conducted by screening USGS ambient water quality data against USEPA ambient water quality criteria for chloride to determine what water bodies/watersheds are potentially at risk from road salt effects. Potential impacts to roadside vegetation and birds ingesting salt were not quantifiable with available information.	
Specific population(s) at increased risk	Aquatic life; particularly younger life stages of fish.	
Quantification of exposure levels to population(s) at increased risk	Same as above.	
Dose/Impact-Response Assessment		
Quantitative impact-assessment employed	Comparison of state water quality data with effect levels on aquatic biota documented in the literature (i.e., USEPA Ambient Water Quality Criteria for Protection of Aquatic Life).	
Risk Characterization		
<p>Risk estimate(s) by population at risk</p> <p>Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude) Assessment of severity/irreversibility</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage:</p> <ul style="list-style-type: none"> • many species threatened/endangered • major community change • extensive loss of habitats/species <p>Long time for recovery</p> <p>3 - Adverse affect on structure and function of system:</p> <ul style="list-style-type: none"> • all habitats intact and functioning • population abundance and distributions reduced <p>Short time for recovery</p>	<p>The assessment conducted was a brief screening assessment based upon the literature on effects. No data on actual effects were measured in any water body. The analysis indicated that 4 of the 136 water quality monitoring stations statewide for which data were available had concentrations of chloride in surface water exceeding the USEPA chronic continuous concentration (CCC) value of 230 mg/L for protection of aquatic life from chloride. These stations exceeded the CCC at least once during the three-year analysis period from 1997-2000, with the recorded chloride concentration in parentheses:</p>	2

<p>2 - Ecosystem exposed but structure and function hardly affected 1 - No detectable exposure</p>	<p>RAHWAY R NR SPRINGFIELD NJ (1300 mg/L) COOPER RIVER AT HADDONFIELD NJ (470 mg/L) GREEN BK AT NORTH PLAINFIELD NJ (310 mg/L) RAMSEY BROOK AT ALLENDALE NJ (290 mg/L)</p>	
<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade)</p> <p>5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)</p>	<p>Exceedances of water quality criteria can be expected to vary between years due to weather conditions. Seasonal variation was also noted; chloride concentrations are generally higher in winter months.</p>	<p>3</p>
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude)</p> <p>5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>Four of 136 stations showed exceedances.</p>	<p>1</p>
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>H - Uncertainties associated with this analysis are as follows. First, effect data were compared to ambient monitoring data for chlorides in the dissolved form. The analysis thus assumes that all of the chloride present in the surface water body that was analyzed is present in the dissolved form. Using ambient stream monitoring data on chlorides to identify</p>	

	potential ecological risks also assumes that all chlorides detected in surface water can be attributed to road salts. In addition, sample results represent a snapshot in time; effects are likely to vary temporally, as chloride and ion concentrations are likely to vary seasonally. While the data set included data from different seasonal events, it cannot be considered representative of all conditions. The data set used (USGS water quality data) was based on the past three years of data (1997-2000), and hence represents years that did not have as high a snowfall total as this year. Thus, road salt applications were likely to have been less. Due to the lack of available data, this analysis was also biased toward rivers and streams, and did not include ponds and lakes near roadways that would be expected to be most affected by road salt. There is also uncertainty about the exposure point concentration as well; water quality data from monitoring locations may not represent the actual concentration of chlorides at their point of entry into a given water body. In short, the analysis is probably not sufficiently conservative to determine the full extent of potential risks.
Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief description. (Data Gaps; highlight significant data needs)	H- the analysis conducted is only a screening level analysis subject to the assumption that chloride concentrations are reflective of road salt, and that exceedances of water quality criteria constitute risk. Only a subset of the state's water bodies were included in the analysis, based upon available data. Also, actual site-specific risks would have to be evaluated in order to assess risks with greater certainty.
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, !, =, ≡; where + is improvement), and brief description.	(+): Current trend in NJ is for some improvement. There is significant opportunity for improvement in road salt storage and application techniques. Storage areas should be covered to avoid leaching of salts, and road salt piles should be placed on impervious cover in order to avoid groundwater contamination. In the State of Vermont, a program has been recently initiated wherein field crews determine road salt application rates based upon road temperature rather than ambient air temperature (Lawson, personal communication, 1999). Since road temperatures are typically higher than air temperatures, they have significantly reduced road salt application rates. Other states such as Massachusetts limit road salt application near reservoirs. Limiting applications adjacent to freshwater lakes and ponds could reduce aquatic risks.
Potential for catastrophic impacts (H,M,L) and brief description	L – Unless a truck overturns or similar spill event occurs. In that case impacts are likely to be temporary and localized, as the material is easily cleaned up.
Link to other Work Groups (e.g., socioeconomic impacts)	Socioeconomic Impacts; Storm Water Runoff Quality/Watershed Management
Extent to which threat is currently regulated or otherwise managed	Unknown.
Barriers to restoration	Need for vehicular transportation; Urban areas tend to be more greatly affected, so developed areas pose a barrier to restoration of the pristine, natural condition.
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	L
Small business industry	L
Transportation	H – road salt application and storage by highway and road departments, airports, parking lots, etc.
Residential	M – residents apply salt to driveways, sidewalks and other areas that eventually enters into storm drains and ultimately

	waterways
Agriculture	L
Recreation	L
Resource extraction	L
Government	H – see transportation above
Natural sources/processes	L – naturally occurring chlorides occurring at higher levels are primarily limited to marine and estuarine habitats
Orphan contaminated sites	L
Diffuse Sources	
Sediment sinks	L – material dissolves readily
Soil sinks	L- material dissolves into soil solution where it can migrate into surface water; soil effects are primarily limited to roadways or areas adjacent to storage facilities with poor handling practices
Non-local air sources incl. deposition	L
Biota sinks	L

Summary Statement:

Studies have indicated potential ecological risks to aquatic and roadside biota from application of road salts. Many of the studies to date have originated in Canada, where road salt applications and ambient water quality concentrations of salts are higher than have been reported for New Jersey. While road salt may affect roadside vegetation, and birds feeding on it directly, no data on these effects was located specific to New Jersey. A review of available water quality data from the U.S. Geological Survey database for stations located in New Jersey indicated that over the past three years (1997-2000) chloride concentrations at only four of 136 stations had chloride concentrations exceeding the U.S. EPA chronic continuous concentration (CCC), an indicator of deleterious ecological effects. These stations all exceeded the CCC during winter months, when chloride concentrations from road salt would be highest. In general, urban areas would be expected to be of greatest risk, due to heavier salt applications. However, there is significant uncertainty associated with this analysis. The analysis is a brief screening assessment; no data on actual effects were actually measured in any water body. The data set used was based on the past three years of data, and cannot be considered representative of all years, including those with higher snowfall and higher road salt applications. Thus, road salt applications were likely to have been less. Due to the lack of available data, this analysis was also biased toward rivers and streams, and did not include ponds and lakes near roadways that would be expected to be most affected by road salt. Finally, the analysis focuses exclusively on road salt and does not address habitat degradation and other potential synergistic impacts on water quality resulting from other non-point source pollutants, particularly in urban areas.

Issue: Road Salt
 Author: P. Bovitz
 Version: 03/19/01

Statewide Analysis of Threat

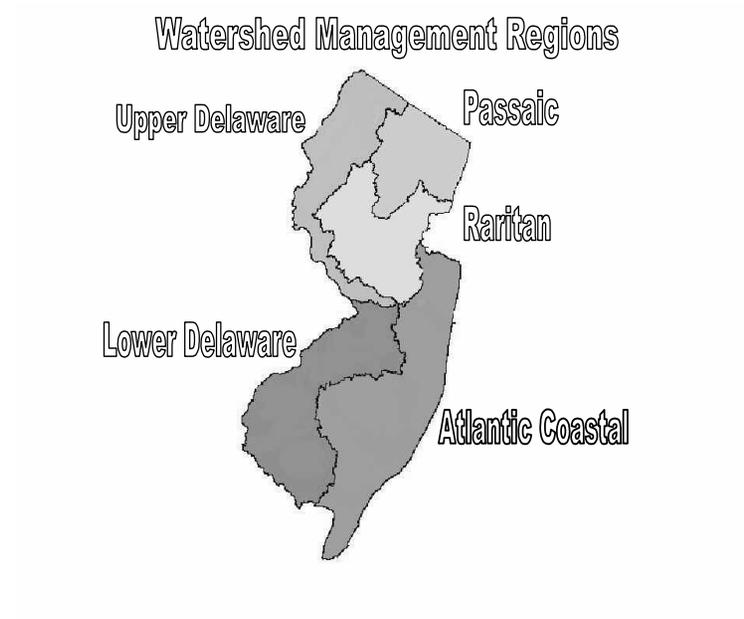
Threat = Road Salt

Ecosystem	Severity	Irreversibility	Frequency	Magnitude	Score
Inland Waters	2		3	1	6
Marine Waters	1		0.1	1	0.1
Wetlands	2		3	1	6
Forests	1		0.1	1	0.1
Grasslands	1		0.1	1	0.1
Total Score					12.3
Average Score (Total ÷ 5)					2.46

Risk by Watershed Management Region

THREAT = Road Salt	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L	NA	L	L	L
Passaic	M	L	M	L	L
Raritan	M	L	M	L	L
Atlantic	L	L	L	L	L
Lower Delaware	M	L	M	L	L
Region/Watershed (secondary)					
Urban	H	L	H	L	L
Suburban	M	L	M	L	L
Rural	L	L	L	L	L

H=high, M=medium, L=low, NA = not applicable



Issue: Road Salt
Author: P. Bovitz
Version: 03/19/01

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Issue: Starlings
Author: Bovitz
Version: 09/18/00

New Jersey Comparative Risk Project
Ecological Quality Technical Work Group
Stressor-Specific Risk Assessment

Stressor:

Starlings

Issue Summary:

The European starling (*Sturnus vulgaris*) is an exotic species intentionally introduced into the United States from Europe in 1890-1891 (Ehrlich et al 1988). Within 60 years populations had increased to the point where the bird had expanded its range as far as the West Coast. While starlings as a group are considered open country birds, the European starling is a habitat generalist that has adapted well to urban and suburban landscapes. Starlings are highly adaptable, exhibiting a broad range of food habits, enabling them to exploit a variety of habitats that have been altered by man. Starlings are hole-nesting birds that aggressively defend their nest sites, often outcompeting native species such as bluebirds, great-crested flycatchers, common flickers and others. They may raise up to 3 broods per year, and are a resident species year round throughout the state. Starlings are considered colonial breeders. While they may often be seen foraging individually or in small groups, the birds become more gregarious during fall and winter months, when they may form large aggregations. Roosting flocks of starlings may number in the thousands to millions of birds.

These ecological characteristics have led to the success of the starling in New Jersey, where it is now one of the most common bird species in the state. These characteristics have also confounded control efforts in other parts of the country, where detergents have been applied at winter roosts in an effort to control local populations. But these efforts have not made a dent in the overall population (Ehrlich et al, 1988). The North American population has been estimated at over 200 million birds, and it is likely that future efforts at control of this species will continue to be unsuccessful (Ehrlich et al 1988). Moreover, since a good deal of the success of this species is related to habitat conversion to suburban and urban landscapes that this species can tolerate, significant barriers exist to the potential restoration of natural conditions occurring prior to the introduction of this species. As a result, ecological risks from this species were not further evaluated in the NJ Comparative Risk program.

Summary: Impacts from starlings primarily involve outcompeting other avian native species. Their success is also related to habitat conversion and fragmentation (e.g., forest fragmentation).

Risk Rating: Medium. Low risk in already developed areas; moderate in areas such as the Pine Barrens; and moderate to high in less developed or undeveloped areas such as the Highlands.

Uncertainty: L

Data Gaps: Detailed studies are lacking on the impacts of starlings on native avian communities in fragmented forest systems.

Trend: (0) the underlying risk is not expected to change substantially over the next five to 10 years.

Ehrlich, P. R., D.S. Dobkin, and D. Wheye. 1988. *The Birder's Handbook*. Simon and Schuster, N.Y. 785 p.

Issue: Thermal Pollution
Author: Bergstein
Version: 05/01/00

New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Stressor:

Thermal Pollution

Issue Description:

Thermal pollution refers to elevated ambient surface water temperatures resulting from industrial discharges. Elevated water temperature may increase metabolic and respiration rates, altering behavior patterns (e.g., feeding and migration) of aquatic organisms. Although rising temperatures may enhance the growth rate of some organisms, eventually higher temperatures can adversely affect reproduction and survival. The extent of damage depends on the rate of temperature change, duration of the exposure, and where the ambient temperature lies in relation to the tolerance range of a given species.

Status in New Jersey

Thermal modification is among the parameters included in the water quality standards that are developed by States, Territories, and Tribes. Specifically, each State is responsible for identifying those waters within its boundaries for which controls on thermal discharges are not stringent enough to assure protection and propagation of a balanced indigenous population of shellfish, fish, and wildlife. Based on the most recent information submitted by New Jersey to EPA's TMDL¹ Program (1998), approximately .7 percent of the impairment to water quality in the State is attributable to thermal pollution identified in the Middle Delaware-Musconetcong, Raritan, Hackensack-Passaic, Middle Delaware, Lower Delaware, and Cohansey Maurice watersheds. In contrast to stressors such as fecal coliform, or total phosphorus, thermal modification currently, does not represent a significant category of water quality impairments reported by New Jersey. We do not expect thermal pollution to pose greater ecological risk in the future given the stringency of permit requirements. Nevertheless, the State updates its water quality fact sheets on a biennial basis, allowing an ongoing means of monitoring the status of this stressor over time.

Risk: LOW

Uncertainty: LOW

Trend: No change in future risk expected (0)

Catastrophic Potential: LOW; Thermal shock, such as when power plants shut down in the winter, can result in fish kills. However, this is usually localized to the area of the plant discharge.

¹ A Total Maximum Daily Load (TMDL) is the sum of the allowable loads of a single pollutant from all contributing point and nonpoint sources.

New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment
 Framework

Findings

Hazard Identification	
Stressor	Tin
Description of stressor	<p>Tin is an element that naturally occurs in the environment; metallic tin is derived from the minerals cassiterite and stannite (Blunden et al., 1985). At least 260 organotin compounds are known and most are manufactured (except for some methyltin compounds; Laughlin et al., 1985).</p> <p>Eisler (1989) summarized the hazards of inorganic tins and organotins to fish and wildlife: Inorganic tin compounds are not highly toxic due to low solubility, poor absorption, low accumulation and rapid excretion. Organotins can cause impaired behavior, and reduced growth, reproduction, and survival. Tributyltin compounds are markedly toxic to aquatic organisms. Bioconcentration of organotins is high in marine waters but biomagnification is not observed due to degradation mechanisms. Environmental concentrations of organotins are not likely to be directly toxic to birds and mammals.</p> <p>Tin is not an essential element to plants, and is readily taken up by plants, remaining primarily in the root system (Wallace and Romney 1977). Tin may be biomethylated to a more toxic form. Plant growth reductions following exposure to inorganic tin have been measured but no specifics on mode of toxicity are available (Efroymsen et al., 1997a).</p> <p>Tin is used in a number of products including food cans, alloys (brass, pewter, bronze, and solder), and toothpaste (stannous fluoride) (Eisler, 1989). Organotins are used in the manufacture of antioxidants (diorganotins; PVC stabilizers), and as biocides (triorganotins; agricultural fungicides, insecticides, slimicide in cooling towers, marine paints). Antifouling paints using organotin were developed in the 1960's (Eisler, 1989). Use of these paints expanded until the 1980's on both recreational and commercial vessels. Realization of the effects of triorganotins on aquatic organisms and the accumulation of these compounds in marinas led to legislative action. In New Jersey, tributyltin (TBT) is regulated as a restricted use pesticide under state and federal laws (Pesticide Control Code, N.J.A.C. 7:30-2.3; Federal Insecticide, Fungicide, and Rodenticide Act, 7 U.S.C. 121). In NJ, TBT paint can only be applied by certified applicators to vessels 25 meters (82 feet) or larger, or to aluminum hulls. The acceptable release rate for TBT paints is not to exceed 4 µg/cm²/day. Seventy percent of the world's fleet of ships uses TBT paint (ENS, 1999), including commercial vessels and many of the world's navies (U.S. EPA, 2000a).</p> <p>Marine gastropods (mollusks) are a sensitive indicator of TBT contamination. Mud snails (<i>Nassarius obsoletus</i>) near estuarine marinas have exhibited imposex, the development of male characteristics over a functionally normal female reproductive anatomy; imposex has also been induced in the laboratory with three TBT compounds at concentrations of 4.5 to 5.5 µg/L (Smith, 1981 in Eisler, 1989).</p>
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	<p>Due to the relatively low toxicity of Sn, as compared to organotins, this assessment will primarily focus on impacts due to organotins, and in particular TBT. Where available, data on Sn will be presented in the assessment.</p> <p>Impacts considered included biological integrity and biodiversity. Impacts under these categories could include acute or chronic toxicity leading to changes in populations or community structure. Tissue concentrations of TBT were also considered in relation to the impacts on the organism itself (e.g., growth or reproductive effects). Ecosystem effects were not considered due to the paucity of data related to</p>

	ecosystem health or ecosystem function.		
Key impacts selected (critical ecological effects)	Key impacts used for this evaluation include biological integrity (e.g., sediment concentrations that impact benthic communities), biodiversity, and tissue concentrations.		
Exposure Assessment			
Exposure routes and pathways considered	Exposure to TBT in aquatic ecosystems is primarily through the water (e.g., gill) and ingestion (i.e., diet) pathways. Exposure media include surface water, sediment, and food (e.g., benthic invertebrates). Octanol-water partition coefficients (K_{ow}) for tributyltins in seawater vary from 5,500 –7,000, but can be modified by salinity and speciation products (Laughlin et al., 1986 in Eisler, 1989). These K_{ow} values indicate TBTs will accumulate in lipids. Bioconcentration of organotins has been observed to be high, but degradation precludes biomagnification (Eisler, 1989).		
Population(s)/ecosystem(s) exposed statewide	Aquatic systems are the primary ecosystems exposed, and in particular marine waters with large vessel/ship traffic (marinas, shipyards, etc). The primary sources of TBT to the aquatic environment include: Paint leaching from boat hull surfaces Runoff from sites where boats are painted Spills Other sources include use of biocides in cooling towers and industrial discharges (Ruppel et al., 1988). Organotin compounds, especially TBT, were highest near marinas and harbors with peak concentrations found in spring and early summer (Eisler, 1989). Organotins were concentrated in surface microlayers of surface waters relative to subsurface layers (Eisler, 1989).		
Quantification of exposure levels statewide	Average concentrations of TBT in NJ media were used to estimate statewide exposure levels. <table style="width: 100%; border: none;"> <tr> <td style="text-align: left;">Matrix</td> <td style="text-align: center;">Benchmark Value</td> </tr> </table> <p>reported a lowest observed effect concentration (LOEC) of 0.101 mg/kg (dry wt.) for a 42 day exposure of the marine polychaete, <i>Armandia brevis</i> to TBT in sediment. They indicated that the juveniles were approximately 3 times more sensitive than adults to TBT exposure. Meador (2000) calculated a sediment quality guideline for TBT using data from <i>Armandia brevis</i> (the less sensitive species), organic carbon-normalized sediment concentration [sed_{oc}] and the following formula:</p> $[\text{sed}_{oc}] = \frac{[\text{tissue}]}{\text{BSAF} \cdot f_{lip}}$ <p>Where: [tissue] is the tissue residue used for protection (LOER or NOER), BSAF is the biota sediment accumulation factor, and f_{lip} is the average lipid content for the species.</p>	Matrix	Benchmark Value
Matrix	Benchmark Value		

Sediment: *Marine benchmarks:* Apparent Effects Threshold (AET) = >3.4 mg/kg as TBT (Buckman, 1999). Meador and Rice (1999) The resulting value of 14,262 ng/g organic carbon equates to a TBT sediment quality guideline of 143 ng/g dry wt. (for a sediment with 1% TOC; Meador, 2000). In NY-NJ Harbor the average TOC concentration ranged from 1.7 to 2.5% in the various subbasins (Adams et al., 1998). Adjusting for TOC would result in a TBT sediment quality guideline range of 243 to 358 ng/g for NY-NJ Harbor. **This is only an estimate for comparison purposes, using a reported less sensitive species. These values should not be considered suitable sediment quality guidelines at this time.** Using a more sensitive species than *Armandia* could result in a lower sediment quality guideline.

Freshwater benchmarks: Background concentrations of Sn in freshwater sediment were listed as 5.0 mg/kg (Buckman, 1999). This value will be used as the benchmark.

Little data is available for freshwater sediments. USGS (DeLuca et al., 1999) collected 14 samples from NJ streams as part of the Long Island-New Jersey National Water Quality Assessment Program in 1997. Fine grain-sized sediment (<63 μ fraction) concentrations of Sn were all below the detection limit (5 mg/kg) except at one site (7 mg/kg). Sn was detected at an average concentration of 44.6 mg/kg (38 samples) in East Lake, a site downstream of a former foundry in Cumberland County.

The average concentration of Sn in the Newark Bay area (28 sites including Arthur Kill, Kill Van Kull, Passaic River, and Hackensack River) was 15.3 mg/kg, and for the New York/New Jersey Harbor (168 sites) the average was 4.96 (Adams et al., 1998). The average based on multiple studies listed in NOAA's database (NOAA, 2000) for the New York/New Jersey Harbor was 10.7 mg/kg (225 samples) with a range of 0.1 to 100 mg/kg.

Organotins were also measured with the following average concentrations (Adams et al., 1998; NOAA database, in mg/kg):

	Monobutyltin	Dibutyltin	Tributyltin	Tetrabutyltin
Newark Bay	0.011	0.039	0.069	0.005
NY/NJ Harbor (Adams et al., 1998)	0.005	0.016	0.030	0.004
NY-NJ Harbor (NOAA Database) (number of samples)	0.038 (216)	0.048 (216)	0.059 (216)	0.005 (168)

TBT was detected in sediments in the Delaware Estuary in 50 of 80 samples (NOAA, 1998). TBT ranged in concentration from approximately 0.002-0.048 mg/kg, with an average detected concentration of approximately 0.005 mg/kg

TBT concentrations in Chesapeake Bay sediments show a decrease in concentration in surface (more recent) sediments, whereas TBT in deeper sediments has not significantly degraded over the last decade (Unger, 1999). Concentrations of TBT in shellfish tissues have declined in Chesapeake Bay (Unger, 1999).

Surface Water: There are no New Jersey surface water quality criteria for tin or TBT. EPA's water quality criteria (as guidance) for TBT for freshwater are 0.46 µg/L (acute) and 0.063 µg/L (chronic); and for saltwater the criteria are 0.37 µg/L (acute) and 0.010 µg/L (chronic) (U.S. EPA. 1999).

Little data was found for freshwater systems (inland waters), as tin is not routinely monitored. Sn was not detected at two sites, a tributary to the Whippany River, and East Lake in Cumberland County investigated as part of DEP's Site Remediation Program.

No statewide average Sn or TBT concentration was readily found for NJ marine/estuarine waters. Cardwell et al. (1999) conducted a study at more than 50 sites across the United States on the aquatic risks posed by TBT. TBT concentrations in surface waters were generally less than 5 ng/L in commercial harbors, shipyards, and fish & shellfish habitats. TBT concentrations in marinas were generally greater than 10 ng/L. Cardwell et al. (1999) concluded that chronic risks to aquatic life remained highest in marinas compared to other sites, but has declined from a risk involving 25% of the species prior to 1989 to a risk involving 6% of the species in 1996. TBT concentrations in Chesapeake Bay have decreased by a factor of five in surface water (Unger, 1999).

Fish Tissue: Shephard (1998) recommended the use of tissue screening concentrations (TSCs) to assist in determining the risk of bioaccumulated contaminants. TSCs were defined as "whole body, wet weight tissue residues of chemicals, which if not exceeded, pose little chance of causing adverse toxicological or ecological harm to aquatic biota." These values were derived from bioconcentration factors and the ambient water quality criteria. TSCs were compared to effect tissue concentrations for over 150 chemicals including TBT based on an extensive literature review (1400 records). The TSC for TBT is 6 µg/kg (Shephard, 1998). Meador (2000) listed a critical body residue (CBR) of 3 mg/kg that equated to the lowest-observed-effect tissue residue that impaired growth.

HQs were generated by comparing NJ fish tissue concentrations of TBT to the TSC value.

TBT concentrations in tissue from the Passaic River (BBL, 2000):

<u>Tissue</u>	<u>Average* (µg/kg)</u>	<u>Range (µg/kg)</u>	<u># of detects (total # of Samples)</u>
Mummichog	2.4	non-detect (ND) – 3.92	19 (45)
Silversides	3.2	ND – 9.15	1 (9)
White Perch	15.1	10.8-20.7	9 (9)
Striped Bass (Juv.)	8.0	ND-11.6	2 (3)
Crab (whole)	3.8	ND-12.6	4 (15)
Bivalve	8.2	ND-11.7	13 (14)

* Average includes use of ½ the detection limit (DL) for those samples below the DL.

In the Mullica River, all samples had non-detect concentrations of TBT (BBL, 2000).

Specific population(s) at increased risk	Aquatic biota in proximity to dry docks and heavy usage areas (e.g., commercial docks and berths) are at increased risk due to the observed elevated concentrations of organotins in the sediments. In addition, mollusk populations in Upper New York Bay may be at increased risk due to the observed tissue residues of organotins in blue mussel.													
Quantification of exposure levels to population(s) at increased risk	These populations will continued to be exposed by organotins leaching from incoming ship traffic, release from sediments, and bioaccumulation from the food chain. No quantification is possible due to lack of data. Qualitatively, the exposure is expected to decrease over time due to the lack of active use (e.g., painting at dry docks) in the U.S. and the planned international ban that is being considered.													
Dose/Impact-Response Assessment														
Quantitative impact-assessment employed	<p>Statewide average concentrations of TBT (where available) were compared to screening benchmarks to assess the risk to organisms living in NJ's ecosystems. Benchmarks for sediment, surface water, and fish tissue were compared to available data on TBT concentrations in NJ for these matrices.</p> <p>The hazard quotient (HQ) method was used to conduct a screening level risk assessment where:</p> $HQ = \frac{\text{Estimated Environmental Concentration}}{\text{Benchmark Concentration}}$ <p>The HQ was used as a measure of potential risk for the various ecosystems. This assumed that concentrations of the contaminant in the various environmental matrices (e.g., soil, water, sediment) were representative of the exposure to biota in those systems. HQ values <1 indicate Sn or TBT concentrations are at a level where adverse effects are not expected and little or no potential risk; values >1 indicate that concentrations are at a level where adverse effects may potentially occur, and there is potential risk to ecological receptors.</p>													
Risk Characterization														
Risk estimate(s) by population at risk		Score												
Assessment of severity/irreversibility	<p>The following scale was used when assigning a score to the generated HQs:</p> <table border="1" data-bbox="485 1239 703 1421"> <thead> <tr> <th><u>HQ</u></th> <th><u>Score</u></th> </tr> </thead> <tbody> <tr> <td><1</td> <td>1</td> </tr> <tr> <td>1-5</td> <td>2</td> </tr> <tr> <td>6-10</td> <td>3</td> </tr> <tr> <td>4</td> <td></td> </tr> <tr> <td>25+</td> <td>5</td> </tr> </tbody> </table> <p>1) STATEWIDE:</p>	<u>HQ</u>	<u>Score</u>	<1	1	1-5	2	6-10	3	4		25+	5	2-3
<u>HQ</u>	<u>Score</u>													
<1	1													
1-5	2													
6-10	3													
4														
25+	5													

Sediment: This data results in HQs of (for SEL) to (for LEL) for freshwater stream sediments. For lakes the HQs ranged from (LEL), and from (SEL).

For marine/estuarine sediments the HQs range from (ER-L) and (ER-M).

Freshwater: 1995-1997 Data: HQ = (acute) and (chronic).

DSRT Data: HQs = to (acute), and to (chronic).

Marine/Estuarine Water: HQs for the average concentration in the waterbodies detailed.

Fish Tissue: HQs for the range in tissue concentrations by species was:

<u>Tissue</u>	<u>HQ – Passaic River</u>	<u>HQ –</u>
<u>Mullica River</u>		
Mummichog	0.4	
0.3		
Silversides	0.5	
NA		
White Perch	2.5	
NA		
Striped Bass (Juv.)	1.3	
NA		
Crab (whole)	0.6	
0.3		
Bivalve	1.4	
0.3		

Passaic R. = 2
 Mullica R. = 1

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Assessment of frequency of effect(s): 5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)	Frequency of effects varies by ecosystem and location. The greatest frequency of effects would be expected in marine/estuarine waters at dry docks and areas with high ship usage (e.g., berths, docks). Therefore, scores for marine and inland waters would be rated a 3. Scores for terrestrial systems would be rated a 2 (rare), as most effects would probably be related to other uses and spills of organotins.	2-3
Size of population(s) and/or extent of the State/habitat affected (impacted): 5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted	The extent of impacts varies by ecosystem and location in the State. The greatest extent of impacts would be expected in high ship usage areas such as NY-NJ Harbor and commercial docks along the Delaware River. However, the impacts may be limited to dry docks and areas with high ship usage (e.g., berths, docks). Therefore, scores for marine waters would be rated a 2. Scores for inland waters and wetlands would be rated a 1. Terrestrial systems would also be rated a 1.	1-2
	Total	4-18
Assessment of uncertainties in this assessment (H,M,L) and brief description	There is moderate uncertainty with this assessment. There is very limited data available on soil, surface water, sediment, and fish tissue concentrations. In addition, these data do not cover all habitats in the State. Therefore, the risk to aquatic life may be underestimated.	
Potential for additional data to result in a significant future change in this risk estimate (H,M,L) and brief description (Data Gaps; highlight significant data needs)	Currently there is no comprehensive or regular monitoring of TBT levels in biological tissues in the State. Ecosystem-level effects are poorly understood, and additional research is needed to better characterize the long-term and chronic effects of TBT discharge to the environment. Additional data may moderately change this risk estimate. More concise estimates of risk will be possible as more data becomes available on TBT levels in the State, and as national/international research directed at ecosystem-level effects is completed.	
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, -, =, ≡ where + is improvement), and brief description.	The potential for significant future change in the underlying risk from TBT is rated (++). In 1999, the International Maritime Organization (IMO; a specialized agency of the United Nations) passed a resolution that banned the application of organotin compounds (e.g., TBT) starting in 2003, and a prohibition on the presence of organotin compounds on ships by 2008. The Marine Environment Protection Committee (MEPC) of the IMO is to develop an instrument, legally binding throughout the world, to address the harmful effects of anti-fouling systems (IMO, 2000). If enacted, this ban will reduce the usage and contribution of organotins from international shipping.	

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Potential for catastrophic impacts (H,M,L) and brief description	The potential for catastrophic impacts to NJ's ecosystems and associated biota is considered to be low based on current release rates, the fate of Sn and TBT compounds, and current knowledge of the severe adverse effects of Sn and TBT.
Link to other Work Groups (e.g., socioeconomic impacts)	Potential socioeconomic impacts of Sn and TBT contamination include the economic costs of remediating hazardous waste sites and contaminated sediments to protect ecosystems. The ban of TBT as an anti-fouling agent could result in additional fuel consumption by marine transporters; and a resultant increase in shipping costs and the cost of goods.
Extent to which threat is currently regulated or otherwise managed	NJDEP's Site Remediation Program (SRP) regulates Sn in the form of administrating or conducting the cleanup of Sn contaminated sites to levels that are safe to human health and the environment. TBT is regulated as a restricted use pesticide under state and federal laws (Pesticide Control Code, N.J.A.C. 7:30-2.3; Federal Insecticide, Fungicide, and Rodenticide Act, 7 U.S.C. 121). In NJ, TBT paint can only be applied by certified applicators to vessels 25 meters (82 feet) or larger, or to aluminum hulls.
Barriers to restoration	The continued use of TBT in marine paints will continue to cause aquatic impacts. Delays in finding and using an effective alternative to TBT will be a barrier to restoration.
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	The primary sources of TBT to the aquatic environment include: Paint leaching from boat hull surfaces Runoff from sites where boats are painted Spills Other sources include use of biocides in cooling towers and industrial discharges (Ruppel et al., 1988) and historical industrial discharges (as evidenced by sediment concentrations).
NJ Primary Sources	
Large business/industry	M: shipping industry (i.e., all products imported or exported from local ports such as NY-NJ Harbor which use vessels treated with TBT paint)
Small business/industry	M: shipping industry (same as large business) and ship repair businesses (e.g., dry docks)
Transportation	M: shipping industry which uses TBT paint on their vessels
Residential	L
Agriculture	L
Recreation	M: recreational vessels larger than 25 m
Resource extraction	L
Government	L: potential use of TBT paints on government vessels
Natural sources/processes	L
Orphan contaminated sites	L
Diffuse Sources	
Sediment	M: Historical activity including leaching of marine paints and water discharges have resulted in elevated concentrations of TBT in sediments adjacent to marinas and ports. These sediments can act as a source of TBT to aquatic ecosystems.

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Soil	L
Non-local air sources incl. Deposition	L
Biota	L: aquatic organisms will bioaccumulate TBT and other organotins, however biomagnification through the food chain has not been shown to occur.

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Summary Statement:

(This statement should include a brief description of the stressor, exposure pathway(s), populations/ecosystems exposed, effects/impacts, and reason for the score given).

Statewide Analysis of Threat

Threat = Tributyltin (TBT)

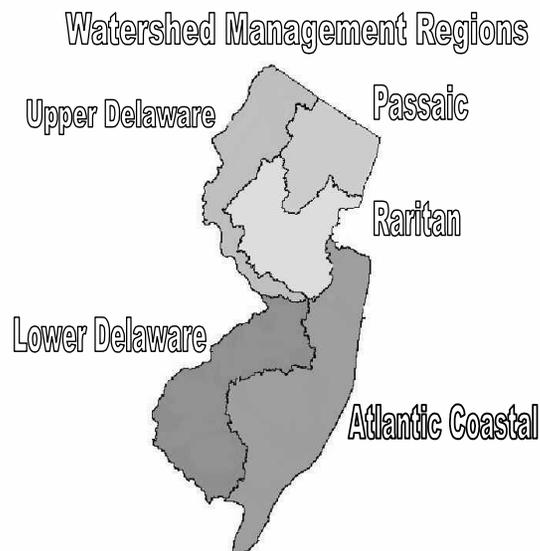
Ecosystem	Severity	Irreversibility	Frequency	Magnitude	Score
Inland Waters	2		3	2	12
Marine Waters	3		3	2	18
Wetlands	2		3	2	12
Forests	1		2	1	2
Grasslands	1		2	1	2
Total Score					46
Average Score (Total ÷ 5)					9.2

Risk by Watershed Management Region

THREAT = TBT	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L	NA	L	L	L
Passaic	L	M	L	L	L
Raritan	L	M-H	L	L	L
Atlantic	L	M	L	L	L
Lower Delaware	L	M	L	L	L
Region/Watershed (secondary)					
Urban	L	M-H	L	L	L
Suburban	L	M	L	L	L
Rural	L	M	L	L	L

H=high, M=medium, L=low, NA = not applicable

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New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	Ultraviolet Radiation
Description of stressor	<p>Ultraviolet Radiation (UV) is an electromagnetic wave, produced in nature by the sun, which has been divided (somewhat arbitrarily) into three categories: UV-A, with wavelengths from 320-400, UV-B, with wavelengths from 280-320 nanometers^[1]; and UV-C, with wavelengths from 100-280 nanometers. The presence of the ozone layer determines what UV reaches the lower atmosphere and surface of the earth. UV-A is not absorbed by the ozone layer, but is considered less hazardous than UV-B, which is mostly absorbed by the ozone layer. UV-C is extremely hazardous, but it is completely absorbed by ozone and normal oxygen (O₂).</p> <p>The amount of UV radiation reaching the earth's surface is partially determined by the effectiveness of the ozone layer. High altitude ozone depletion has been shown to have an effect on both terrestrial and aquatic ecosystems, and studies indicate that additional and increasing impacts will occur as the ozone layer continues to deteriorate. While the public is aware of the "Ozone Holes" in the Arctic regions, ozone depletion is not limited to polar regions. "In general, ozone depletion is greater at higher latitudes, thus, the decrease near Seattle will be greater than near Los Angeles, while Miami will see the smallest depletion of the three cities. However, southern cities also have much higher incidence of UV-B light; even with less depletion, the net increase in UV-B can be greater." ^[2]</p>
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	<p>Terrestrial Ecosystems</p> <p>"Increased UV-B can be damaging for terrestrial organisms including plants and microbes. UV-B effects manifest themselves in many ways including changes in life-cycle timing, changes in plant form, and production of plant chemicals which play a role in protecting plants from pathogens and insect attack." ^[3]</p> <p>"Responses to increased UV-B are evident primarily in interactions among species, rather than in the performance of individual species. Recent experimentation indicates that increased UV-B affects the balance of competition among higher plants, the degree to which higher plants are consumed by insects, and susceptibility of plants to pathogens." ^[3]</p> <p>"Effects of increased UV-B radiation may accumulate from year to year in long-lived perennial plants and from generation to generation in annual plants. This effect has been shown in a few recent studies, but the generality of this accumulation among species is not presently known. If this phenomenon is widespread, this would amplify otherwise subtle responses to UV-B seen in a single growing season, for example, in forest trees." ^[3]</p> <p>Aquatic Ecosystems</p> <p>Studies continue to demonstrate that solar UV-B and UV-A have adverse effects on the growth, photosynthesis, protein and pigment content, and reproduction of phytoplankton, thus affecting the food web. ^[3]</p> <p>"Macroalgae and seagrasses show a pronounced sensitivity to solar UV-B. They are important biomass producers in aquatic ecosystems. Most of these organisms are attached and so cannot avoid being exposed to solar radiation at</p>

	<p>their growth site. Effects have been found throughout the top 10-15 m of the water column."^[3] "Zooplankton communities as well as other aquatic organisms including sea urchins, corals, and amphibians are sensitive to UV-B."^[3] "UV-B radiation is absorbed by and breaks down dissolved organic carbon (DOC) and particulate organic carbon (POC) and makes the products available for bacterial degradation and remineralization. This can lead to the increase in UV-B and UV-A penetration into the water column."^[3] "Potential consequences of enhanced levels of exposure of aquatic ecosystems to UV-B radiation include reduced uptake capacity for atmospheric carbon dioxide, resulting in the potential augmentation of global warming."^[3]</p>
<p>Key impacts selected (critical ecological effects)</p>	<p>Terrestrial Ecosystems The major anticipated effects of increased solar UV-B on terrestrial ecosystems may result from direct UV-B radiation. "Nonbiological UV-B effects can influence other processes in ecosystems. For individual organisms, there are several potential pathways of UV-B action in damage and regulatory processes that affect whole organism performance, such as growth and reproduction."^[4] Even if plant production per say is not affected by increased UV-B, changes in plant form can result in changes in which species can more effectively compete for sunlight. "In a six-year field study using modulated UV-B lamp systems, the competitive balance of two species (wheat and a common weed, wild oat) could be changed even though increased UV-B radiation had no effect on production and growth of these species if grown by themselves. Subtle changes in plant form of the two species were sufficient to change the balance of competition for sunlight. In this experiment, the wheat benefited from the increased UV-B and the weed suffered. However, in other mixtures of crop and weeds the situation might be reversed."^[5] One study has suggested that monoculture production, such as what we might expect from modern farming practices, might not be altered by an increase of UV-B. "But, in the presence a more diverse ecosystem, changes in competitive balance may be expected with an increase in UV-B, specifically, when mixtures where monocots are involved, rather than only dicots."^[6] "Both plants and phytoplankton vary widely in their sensitivity to UV-B. When over 200 agricultural plants were tested, more than half showed sensitivity to UV-B light. Other plants showed negligible effects or even a small increase in vigor. Even within a species there were marked differences; for example one variety of soybean showed a 16% decrease in growth while another variety of the same soybean showed no effect [R. Parson]. An increase in UV-B could cause a shift in population rather than a large die-off of plants."^[7] "Increased UV-B has been shown to advance or delay (depending on the species) the time of flowering in plants. This shift can mean that a plant species may not have sufficient insect pollinators available at the new time of flowering, either because the insects are not present or because other plant species are attracting these pollinators." Additional study on flowering responses is necessary for a better understanding of this issue.^[4]</p> <p>Aquatic Ecosystems There is increased consensus, covering a wide range of aquatic ecosystems, that environmental UV-B, independent of ozone-related increases, is an important ecological stress that influences the growth, survival and distribution of phytoplankton. Polar ecosystems, where ozone-related UV-B increases are the greatest and which are globally significant ecosystems, are of particular concern.^[8] Evidence of detrimental effects on bacterioplankton was examined in one study of UV-B in marine environments. It was found that decreased bacterioplankton activity may lead to an increase in dissolved organic matter (DOM) in ocean waters as bacterial assimilation of DOM is reduced. Cyanobacteria, an organism important in nitrogen fixation,</p>

	<p>is also at risk.^[9]</p> <p>"Within freshwater ecosystems, the interactions of UB-B and heavy-metal concentrations resulted in synergistic inhibition of nutrient uptake, enzyme activity, carbon fixation, ATP synthesis and oxygen evolution in a number of phytoplankton species. 67 freshwater species of algae (<i>Chlorophyta</i> and <i>Chromophyta</i>) were screened in an experiment to determine their UV-B sensitivity. The algae were selected to represent different ecosystems ranging from high-altitude lakes to thermal springs. The most sensitive species lost 30-50% of their oxygen-evolving capacity during a 2h UV-B exposure (2 W m⁻²). It should be noted that predictions of responses by ecosystems to elevated UV-B exposure should not be based solely on single species assessments."^[8]</p> <p>"Potential consequences of enhanced levels of exposure to UV-B radiation include loss of biomass, such as food sources for humans, changes in species composition decrease in availability of nitrogen compounds, and reduced uptake capacity for atmospheric carbon dioxide, resulting in the potential augmentation of global warming."^[8]</p>
Exposure Assessment	
Exposure routes and pathways considered	UV radiation in the environment is primarily solar radiation. Factors that alter the exposure levels to natural UV include: latitude and elevation, cloud cover, proximity to an industrial area, and albedo (reflection).
Population(s)/ecosystem(s) exposed statewide	Terrestrial Ecosystems Agricultural lands, forests, grasslands, etc. Aquatic Ecosystems Coastal areas and inland waterways
Quantification of exposure levels statewide	Exposure levels determined by system driven variables.
Specific population(s) at increased risk	Terrestrial Ecosystems Aquatic Ecosystems
Quantification of exposure levels to population(s) at increased risk	Terrestrial Ecosystems Dependant on numerous plant dependant variables. Research available for only certain plants. Additional research is required. Aquatic Ecosystems Dependant on organism. Research available for only certain organisms. Additional research is required.
Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	
Risk Characterization	
Risk estimate(s) by population at risk	
Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)	Score

<p>Assessment of severity/irreversibility</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage: <ul style="list-style-type: none"> • many species threatened/endangered • major community change • extensive loss of habitats/species Long time for recovery</p> <p>3 - Adverse affect on structure and function of system: <ul style="list-style-type: none"> • all habitats intact and functioning • population abundance and distributions reduced Short time for recovery</p> <p>2 - Ecosystem exposed but structure and function hardly affected</p> <p>1 - No detectable exposure</p>	<p>Terrestrial Ecosystems Aquatic Ecosystems</p>	<p>3 3</p>
<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade)</p> <p>5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)</p>	<p>Terrestrial Ecosystems Aquatic Ecosystems</p>	<p>4 4</p>
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude)</p> <p>5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>Terrestrial Ecosystems Aquatic Ecosystems</p> <p>*See "Key impacts selected (critical ecological effects)" for materials.</p>	<p>4 4</p>
	<p>Total</p>	<p>48</p>
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>(L) There is little doubt that UV affects the environment. Questions remain however regarding the extent of the damaging effects and quantifying the potential for increased hazard due to increased levels of UVR due to ozone depletion.</p>	
<p>Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief</p>	<p>(M) Additional research is required, especially in the area of terrestrial ecosystems in order to more accurately predict the effects of increased UVR. In addition, State specific data should be acquired in terms of UVR exposures and NJ</p>	

description. (Data Gaps; highlight significant data needs)	specific terrestrial and aquatic communities affected in order to better interpret national and international research.
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, !, =, ≡; where + is improvement), and brief description.	(-) Thinning of atmospheric ozone contributes to an increase in solar UV penetration to the earth's surface. It is expected that ozone depletion will continue for an undetermined length of time. Unchecked, this will result in an increase in UV exposure and the likelihood of an increase in the negative effects associated with such an increase.
Potential for catastrophic impacts (H,M,L) and brief description	(L) A catastrophic reduction in atmospheric ozone sufficient to allow penetration of significant amounts of UVB and UVC would likely result in the severe disruption of life within all ecosystems. While it is extremely unlikely that such an event might occur, continued ozone depletion is and should be of grave concern.
Link to other Work Groups (e.g., socioeconomic impacts)	
Extent to which threat is currently regulated or otherwise managed	<p>Background (Source: EPA) ^[10]</p> <p>In July 1992, EPA issued its final rule implementing section 604 of the Clean Air Act Amendments of 1990. That section limits the production and consumption of a set of chemicals known to deplete the stratospheric ozone layer. EPA controls production and consumption by issuing allowances or permits that are expended in the production and importation of these chemicals. These allowances can be traded.</p> <p>The July, 1992 rule required producers of class I substances (chlorofluorocarbons (CFCs), halons, carbon tetrachloride, and methyl chloroform) to gradually reduce their production of these chemicals and to phase them out completely as of January 1, 2000 (2002 for methyl chloroform). In addition to these production limits, the rule required a similar reduction in consumption, defined as production plus imports minus exports.</p> <p>On February 11, 1992, the United States, responding to recent scientific findings, announced that the phaseout of the production of CFCs, halons, carbon tetrachloride, and methyl chloroform would be accelerated and that these substances would be phased out by December, 31, 1995. It was also stated that the U.S. will consider recent evidence suggesting the possible need to phase out methyl bromide. At the same time, the Agency received petitions from environmental and industry groups to accelerate the phaseout of these chemicals.</p> <p>The recent United Nations Environmental Program (UNEP) Scientific Assessment identified methyl bromide, widely used as a soil fumigant, as a significant ozone-depleting compound. The parties set an ozone depletion potential of 0.7 for this chemical. Starting in 1994, this regulation freezes the production and consumption of this chemical at 1991 levels through the year 2000. The Agency is obligated under the Clean Air Act to phase out this chemical by the year 2001.</p> <p>HCFCs (Source: NOAA) ^[11]</p> <p>HCFCs are compounds containing carbon, hydrogen, chlorine and fluorine. Certain chemicals within this class of compounds are viewed by industry and the scientific community as acceptable alternatives to chlorofluorocarbons. The HCFCs have shorter atmospheric lifetimes than the CFCs and a much smaller capacity to deliver reactive chlorine to the stratosphere where the ozone layer is found. Consequently, it is expected that these chemicals will contribute much less to stratospheric ozone depletion than CFCs. Because they still contain chlorine and have the potential to destroy stratospheric ozone, they are viewed only as temporary replacements for the CFCs. Current international</p>

	legislation has mandated production caps for HCFCs in the future; production in developed countries is prohibited after 2030. See the Summary Statement for a table with the phase-out schedule for HCFC's.
Barriers to restoration	Delays in the phase out of HCFC's and the continued use of stockpiled CFC's.
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	(M) Air Conditioners: Use of HCFC's and CFC's in older units. Industrial Processes.
Small business industry	(M) Air Conditioners: Use of HCFC's and CFC's in older units. Industrial Processes.
Transportation	(M) Air Conditioners: Use of HCFC's and CFC's in older units.
Residential	(M) Air Conditioners: Use of HCFC's and CFC's in older units.
Agriculture	N/A
Recreation	N/A
Resource extraction	N/A
Government	N/A
Natural sources/processes	N/A
Orphan contaminated sites	N/A
Diffuse Sources	
Sediment sinks	N/A
Soil sinks	N/A
Non-local air sources incl. deposition	(L) Continued use of atmospheric ozone depleting worldwide.
Biota sinks	N/A

Summary Statement

Increases in UV will have wide ranging impacts on both terrestrial and aquatic ecosystems, including species composition and biomass productivity. Should trends on depletion of the ozone layer continue, effects on the food web and contribution to global warming are two of the potential impacts we could expect to see. Phaseout of ozone depleting chemicals continues, but because these chemicals take time to reach the altitude where destruction of the ozone occurs, the full impact of these chemicals remains to be seen.

"The following table* shows the U.S. schedule for phasing out its use of HCFCs in accordance with the terms of the Protocol. The Agency intends to meet the limits set under the Protocol by accelerating the phaseout of HCFC-141b, HCFC-142b and HCFC-22. These are the most damaging of the HCFCs. By eliminating these chemicals by the specified dates, the Agency believes that it will meet the requirements set by the Parties to the Protocol. The third and fourth columns of the table show how the U.S. will meet the

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 international obligations described in the first two columns." - USEPA"

{PRIVATE}Montreal Protocol		United States	
Year by which Developed Countries Must Achieve % Reduction in Consumption	% Reduction in Consumption, Using the Cap as a Baseline	Year to be Implemented	Implementation of HCFC Phaseout through Clean Air Act Regulations
2004	35.0%	2003	No production and no importing of HCFC-141b
2010	65.0%	2010	No production and no importing of HCFC-142b and HCFC-22, except for use in equipment manufactured before 1/1/2010 (so no production or importing for NEW equipment that uses these refrigerants)
2015	90.0%	2015	No production and no importing of any HCFCs, except for use as refrigerants in equipment manufactured before 1/1/2020
2020	99.5%	2020	No production and no importing of HCFC-142b and HCFC-22
2030	100.0%	2030	No production and no importing of any HCFCs

*Source: USEPA

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Statewide Analysis of Threat

Threat =

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	3	4	4	48
Marine Waters	3	4	4	48
Wetlands	3	4	4	48
Forests	3	4	4	48
Grasslands	3	4	4	48
Total Score				240
Average Score (Total ÷ 5)				48

Watershed Management Regions

The map shows five distinct watershed management regions in New Jersey, each shaded in a different tone of gray. The regions are labeled as follows: Upper Delaware (top north), Passaic (top east), Raritan (middle east), Lower Delaware (bottom west), and Atlantic Coastal (bottom east).

Risk by Watershed Management Region

THREAT =	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L	NA	L	L	L
Passaic	L	M	L	L	L
Raritan	L	M	L	L	L
Atlantic	L	M	L	L	L
Lower Delaware	L	M	L	L	L
Region/Watershed (secondary)					
Urban	N/A	M	L	L	L
Suburban	L	M	L	L	L
Rural	L	M	L	L	L

H=high, M=medium, L=low, NA = not applicable

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- [1] The EPA has established a definition of UVB as wavelengths from 280-320nm. The World Health Organization (WHO) measures UVB as between 280 - 315nm
- [2] EPA: Stratospheric Protection Division
- [3] Environmental Effects of Ozone Depletion: 1998 assessment: "Executive Summary" Journal of Photochemistry and Photobiology, B:Biological 46 (1998) 1-4
- [4] M.M. Caldwell, L.O. Bjorn, J.F. Bornman, S.D. Flint, G. Gulandaivelu, A.H. Teramura, M. Tevini; "Effects of increased solar ultraviolet radiation on terrestrial Ecosystems" Journal of Photochemistry and Photobiology, B:Biological 46 (1998) 40-52
- [5] P.W.Barnes, P.W.Jordan, W.G.Gold, S.D. Flint, M.M. Caldwell, Competition, morphology and canopy structure in wheat (*Triticum aestivum* L.) and wild oat (*Avena fatua* L.) exposed to enhanced ultraviolet-B radiation, Functional Ecology 2 (1988) 319-330
- [6] Barnes, P. W., S. D. Flint, and M. M. Caldwell. 1990. Morphological responses of crop and weed species of different growth forms to ultraviolet-B radiation. *American Journal of Botany* 77: 1354-60
- [7] NASA R.Parson *FAQ 111* ,UV and biological effects of UV
- [8] D.P.Hader, H.D.Kumar, R.C.Smith, R.C.Worrest; "Effects on Aquatic Ecosystems" Journal of Photochemistry and Photobiology, B:Biological 46 (1998) 53-68
- [9] Herndl, G. J., G. Muller-Niklas, and J. Frick. 1993. Major role of ultraviolet-B in controlling bacterioplankton growth in the surface layer of the ocean. *Nature* 361: 717-19
- [10] EPA - The Accelerated Phase-out of Class I Ozone-Depleting Substances (<http://www.epa.gov/ozone/title6/phaseout/acfact.html>)
- [11] NOAA - (<http://www.cmdl.noaa.gov/noah/flask/hcfc.html>)

New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	Volatile Organic Compounds (VOCs)
Description of stressor	<p>The term volatile organic compounds (VOCs) represents a group of stressors that includes a wide range of naturally-occurring and synthetic compounds. While VOCs can contaminate any media, their volatility makes them primarily an air problem, with inhalation the primary route of exposure for humans. Some concentrations are the result of emissions (e.g., benzene); others can form as the by-products of reactions among other compounds in the air (e.g., chloroform). Others, such as formaldehyde, can result from both processes.</p> <p>Benzene and others are classified as hazardous air pollutants because of their potential to cause cancer and other serious health effects in humans. But the risks from VOCs do not end with exposure to the VOCs themselves. Once in the ambient air, a number of VOCs react in the presence of sunlight with oxides of nitrogen (NOx) to form ground-level ozone. The focus of this review is on the risks associated with VOCs in outdoor air: both as hazardous air pollutants and as precursors to ground-level ozone formation.</p>
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	In general VOCs pose the greatest threat to habitat/ecosystem health. It is unlikely that VOCs would adversely affect biological integrity, biodiversity, or ecosystem function on a large scale because animals can and are likely to flee an area contaminated with irritating VOCs. While plants do not have this ability, VOCs dissipate quickly enough where they would not pose a substantial threat.
Key impacts selected (critical ecological effects)	Because VOCs volatilize quickly, most terrestrial systems are not at high risk from releases of this stressor group. Aquatic systems are more at risk because VOCs can dissolve in water, although most rapidly volatilize. There are no documented examples of particular aquatic communities or species that are at risk at this time from VOC emissions.
Exposure Assessment	
Exposure routes and pathways considered	VOCs are easily transported through all media: land, air, and surface water and groundwater. Some, such as acetone and to a lesser extent chloroform, dissolve in water and can be conveyed in water and wastewater discharges. In groundwater, high concentrations of VOCs can become DNAPLs (dense non-aqueous phase liquids). In this setting, VOCs moving under the influence of gravity can separate from the main part of a contamination plume.
	VOCs will sorb to organic matter in sediment and soil proportionately with their K _{OW} (octanol-water partitioning coefficient). The higher the K _{OW} , the greater potential for a VOC to bioaccumulate. Chlorinated alkanes such as carbon tetrachloride, chloroform, and 1,1,2 trichloroethane are not readily metabolized to CO ₂ and water, but have toxic intermediates. Some aromatic VOCs may be subject to

	proportion to nutrient caloric intake). The greater the intensity and duration of VOC exposure, the more likely that the mix of agents will cause chronic problems, but these have not been documented in the environment for animals or people outside of occupational exposures.	
Specific population(s) at increased risk	Plants as well as bird populations nesting along New Jersey's highways (which are throughout the state) and in New Jersey's industrial areas (which are highly concentrated in Northern New Jersey) are at greatest risk of exposure.	
Quantification of exposure levels to population(s) at increased risk	Not available.	
Dose/Impact-Response Assessment		
Quantitative impact-assessment employed	<p>The longer the exposure, the lower the toxicity threshold. For example, the carbon tetrachloride 24 hour LC₅₀ for <i>Daphnia magna</i> is > 770,000 µg/l, but the 48 hour LC₅₀ is 35,000 µg/l. Toxicity thresholds for VOCs are fairly high, ranging from 96 hour LC₅₀ of 13300 µg/l for trichloromethane in bluegills to 150,000 µg/l of carbon tetrachloride in inland silversides. Given the high concentrations and long durations needed to cause acute toxicity, impacts are primarily likely to result from large spills into small water bodies, unpermitted discharges, and frank failures of treatment technologies. If not chronic and if uninvolved with other chemicals (i.e., NAPLs are not formed) the VOCs will dissipate and volatilize, so ecosystems will recover. Continuous leaks could result in localized aquatic impacts.</p> <p>Concentrations of chloroform, trichloroethylene, and tetrachloroethylene found in fish in coastal Norwegian waters at levels of 10 - 1000 ppb. Bioaccumulation factors for chloroform and trichloroethylene were found to be 200-500 and 1400 respectively. This reversible accumulation may have chronic consequences similar to those in terrestrial animals, but they have not been documented.</p> <p>Data from toxicity tests show that animals must be exposed to fairly high concentrations for extended periods of time. The lowest value listed, exposure to 280 ppm of PCE for two hours, resulted in a toxic response in mammals (humans). For this type of exposure to occur, an organism would have to be situated near a continuous source of the substance, or in a closed location where the vapors could not dissipate.</p>	
Risk Characterization		
Risk estimate(s) by population at risk		Score
Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)		
Assessment of severity/irreversibility	Individuals or populations are only severely affected by prolonged exposure to high levels of VOCs, which is a relatively unusual event. Even concentrated spills volatilize quickly. There are no documented examples of ecosystem structure or function being adversely impacted, or of individual populations being stressed from VOCs <i>per se</i> . Impacts of	2
5 - Lifeless ecosystems or fundamental change; Irreversible		
4 - Serious damage: many species threatened/endangered		

<p>major community change extensive loss of habitats/species Long time for recovery 3 – Adverse affect on structure and function of system: all habitats intact and functioning population abundance and distributions reduced Short time for recovery 2 – Ecosystem exposed but structure and function hardly affected 1 - No detectable exposure</p>	<p>VOCs via ozone degradation are of concern however, and should continue to be researched and monitored. Their effects may ultimately warrant re-evaluation for this criterion for New Jersey.</p> <p>VOCs volatilize quickly and their effects generated in the lower atmosphere on aquatic and terrestrial ecosystems are relatively short-term.</p>	
<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade)</p> <p>5 - Often and increasing 4 - Often and continuing 3 <input type="checkbox"/> Occasional 2 <input type="checkbox"/> Rare 1 - Possible in the future 0 <input type="checkbox"/> Unlikely (or 0.1)</p>	<p>VOCs have relatively high toxicity thresholds. For toxic exposures to occur, organisms must be exposed to high concentrations in enclosed spaces or to continuous releases. Local impacts to terrestrial and aquatic life are possible in the vicinity of spills or inactive hazardous waste sites. VOCs probably do not cause widespread ecological harm.</p>	2
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude)</p> <p>5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>Although some VOCs are released in all areas across the state, it is likely that effects would occur primarily in areas of significant urbanization.</p>	2
	Total	8
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>(M) Ideally, comparative risk analysis would extend to consideration of the ecosystem effects of ozone, the repercussions of plant injury on other ecological components, and ultimately how it would affect structure and function at the ecosystem level. Estimating the extent of plant injury based on ambient ozone levels and species inventories is confounded by the lack of sufficient monitoring sites for estimating ambient concentrations.</p>	
<p>Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief description. (Data Gaps;</p>	<p>(M) If it is determined at some future time that VOCs significantly contribute to ozone depletion and the ecological effects therefrom are unacceptable, this issue may warrant more attention. The ecological effects of ozone depletion may extend beyond three years.</p>	

highlight significant data needs)	
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, -, =, _; where + is improvement), and brief description.	(0) There are no documented examples of ecosystems with high or moderate ecological significance being adversely impacted by VOCs, nor are there documented examples of sensitive, endangered, or unique species or communities affected, at this time.
Potential for catastrophic impacts (H,M,L) and brief description	(L) The worst-case scenario would result from the release of large volumes of VOCs from solvents in transit, underground storage tanks, or inactive hazardous waste sites. Such sites could result in soil being contaminated as the VOCs diffuse, and ground water being contaminated from liquid VOCs. Concentrations of VOCs could permeate through the soil and enter surface water bodies or wetlands. Aquatic and terrestrial life could be harmed in the general vicinity of the hazardous waste site, but the impacts tend to be limited to that immediate area if only VOCs are involved. Unfortunately, their occurrence in such sites is typically as solvents contaminated or mixed together with PAHs, heavy metals, and pesticides. The formation of non-aqueous phase liquids (NAPLs) from the VOCs may help transport other compounds or prevent their degradation in the environment. When NAPLs are more dense than water, they sink into the bottom sediment as a pool, continuing to release toxicants to aquatic and terrestrial organisms long after the source has been abated or cleaned up.
Link to other Work Groups (e.g., socioeconomic impacts) Extent to which threat is currently regulated or otherwise managed	To reduce ozone levels, the Clean Air Act requires states to determine VOC emissions, and devise a plan to reduce them by a certain amount. This has proved difficult due to population growth and increases in vehicle miles traveled (VMT). In the US, after 25 years of regulation, over 70 million people are living in nonattainment areas. Moreover, recent studies indicate that respiratory effects may be occurring at levels below the current standard. During this same time period, U.S. population increased by 28%, and VMT increased 116% (U.S. EPA numbers). The challenges encountered in characterizing and controlling the VOC/ozone problem will intensify under a scenario of tightening standards along with increasing population and sprawl.
	Significant data gaps and unanswered questions remain, but the information gathered by comparative risk projects around the country suggests that we may already know enough about the problem, specifically the driving forces, to pursue an effective risk reduction strategy. Reducing VOC emissions via large-scale reductions in VMT is the next logical policy goal. As the Columbus project concluded in its quality of life report, "the most dramatic and realistic improvements in air quality will be obtained through a mode-shift to transit services and a reduction in vehicle miles traveled by the personal automobile."
Barriers to restoration	While accurate emissions inventories have been elusive, a major culprit distinctly emerges. Motor vehicles, according to EPA numbers cited by several projects, account for 35% of VOCs and 56% of air toxics. California estimates 100 to 1,000 cancer cases per year can be attributed to mobile sources of air pollution, with VOCs like benzene major constituents of the problem. Michigan reported 48% of its VOC emissions attributable to mobile sources. Overall, motor vehicles account for roughly one-third to one-half or more of the problem. Air quality improvements that have been achieved via emissions controls will

	likely be counteracted in the near future if sprawl and increased VMT continues.
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	H
Natural sources/processes	L
Orphan contaminated sites	M
Diffuse Sources	
Sediment sinks	L
Soil sinks	L
Non-local air sources incl. deposition	L
Biota sinks	L

Summary Statement:

The term volatile organic compounds (VOCs) represent a group of stressors that includes a wide range of naturally occurring and synthetic compounds. While VOCs can contaminate any media, their volatility makes them primarily an air problem, with inhalation the primary route of exposure for humans. Some concentrations are the result of emissions (e.g., benzene); others can form as the by-products of reactions among other compounds in the air (e.g., chloroform). Others, such as formaldehyde, can result from both processes. Benzene and others are classified as hazardous air pollutants because of their potential to cause cancer and other serious health effects in humans. But the risks from VOCs do not end with exposure to the VOCs themselves. Once in the ambient air, a number of VOCs react in the presence of sunlight with oxides of nitrogen (NO_x) to form ground-level ozone. The focus of this review is on the risks associated with VOCs in outdoor air: both as hazardous air pollutants and as precursors to ground-level ozone formation. Because VOCs volatilize quickly, most terrestrial systems are not at high risk from releases of this stressor group. Aquatic systems are more at risk because VOCs can dissolve in water, although most rapidly volatilize. There are no documented examples of particular aquatic communities or species that are at risk at this time from VOC emissions.

VOCs are easily transported through all media: land, air, surface water and groundwater. Some, such as acetone and to a lesser extent chloroform, dissolve in water and can be conveyed in water and wastewater discharges. In groundwater, high concentrations of VOCs can become DNAPLs (dense non-aqueous phase liquids). In this setting, VOCs moving under the influence of gravity can separate from the main part of a contamination plume.

Because sewage treatment plants convey large quantities of water to surrounding water bodies, they have long been suspected as the source of a variety of toxics in the environment but also present a real opportunity for pollution prevention and reduction. VOCs are not destroyed by the sewage treatment process and are released into the environment via air and water pathways. Storm water runoff may be another significant pathway for carrying toxic contaminants into waterways. Air is the most likely avenue of exposure for terrestrial wildlife, but this direct exposure is limited. Plants as well as bird populations nesting along New Jersey's highways (which are throughout the state) and in New Jersey's industrial areas (which are highly concentrated in Northern New Jersey) are at greatest risk of exposure. Therefore, because VOCs often volatilize before they can create a long-term effect on an ecosystem, the threat scores are low.

Issue: Volatile Organic Compounds (VOCs)
 Author: Gillespie
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Statewide Analysis of Threat

Volatile Organics Compounds (VOCs)
Threat =

{ "Threat = Volatile Organic Compounds" }

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	2	2	2	8
<i>Marine Waters</i>	2	2	1	4
<i>Wetlands</i>	2	2	1	4
<i>Forests</i>	2	2	1	4
<i>Grasslands</i>	2	2	1	4
			Total Score	24
			Average Score	4.8

Risk by Watershed Management Region

<i>THREAT = VOCs</i>	ECOSYSTEM				
<i>Watershed Management Region</i>	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
<i>Upper Delaware</i>	L	NA	L	L	L
<i>Passaic</i>	L	L	L	L	L
<i>Raritan</i>	L	L	L	L	L
<i>Atlantic</i>	L	L	L	L	L
<i>Lower Delaware</i>	L	L	L	L	L
<i>Region/Watershed (secondary)</i>					
<i>Urban</i>	H	M	M	M	M

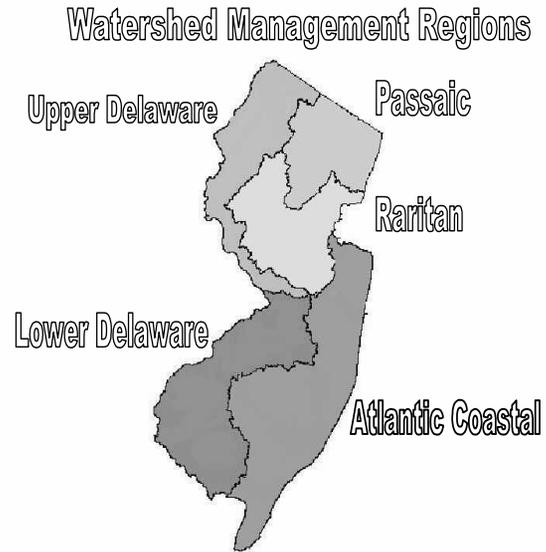
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Suburban	L	L	L	L	L
Rural	L	L	L	L	L

H=high, M=medium, L=low, NA = not applicable



New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

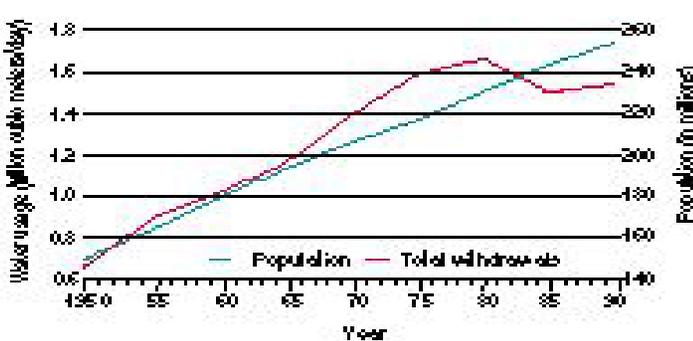
Risk Assessment Framework	Findings																														
Hazard Identification																															
Stressor	Water Overuse																														
Description of stressor	<p>Overuse of water resources, including surface water and ground water, can have profound impacts on ecological systems. This is especially evident in aquatic systems where overuse may result in reduction or loss of stream flow. Reduction or alteration in stream flow can also be caused by other stressors such as increase in impervious cover or channelization (see separate assessments). Loss of or impacts to wetlands can occur due to reduced or loss of hydrologic connection with surface/groundwater due to over pumping/overuse. Impacts can also occur due to uses that result in water quality degradation</p> <p>This assessment will primarily focus on water uses (whether depletive, consumptive, or non-consumptive) that reduce or alter stream flow. This assessment will examine depletive water use, current and projected water deficits, and water usage by county and water region to assess water use/overuse.</p> <p>Nationally, fresh water is considered a limited ecological (physical and biological) and economical resource (Herrmann et al., 1998). Water use has also increased nationally between 1950 and 1990 (Figure 1). Uses of surface and ground water include drinking water (potable water), irrigation, power generation, cooling (e.g., industrial), mining, commercial, recreation, and other uses. Potable uses accounted for the highest withdrawals for the period 1990-1996 in NJ, followed by power generation, industrial, agriculture, and mining. Irrigation (e.g., golf courses) and commercial were relatively minor withdrawals (Hoffman & Lieberman, 2000).</p>  <table border="1"> <caption>Estimated data for Figure 1: Population and water use in the United States, 1950-1990</caption> <thead> <tr> <th>Year</th> <th>Population (millions)</th> <th>Total withdrawals (billion cubic meters/day)</th> </tr> </thead> <tbody> <tr><td>1950</td><td>150</td><td>0.65</td></tr> <tr><td>1955</td><td>160</td><td>0.85</td></tr> <tr><td>1960</td><td>170</td><td>1.00</td></tr> <tr><td>1965</td><td>180</td><td>1.15</td></tr> <tr><td>1970</td><td>190</td><td>1.30</td></tr> <tr><td>1975</td><td>200</td><td>1.50</td></tr> <tr><td>1980</td><td>210</td><td>1.65</td></tr> <tr><td>1985</td><td>220</td><td>1.50</td></tr> <tr><td>1990</td><td>240</td><td>1.65</td></tr> </tbody> </table>	Year	Population (millions)	Total withdrawals (billion cubic meters/day)	1950	150	0.65	1955	160	0.85	1960	170	1.00	1965	180	1.15	1970	190	1.30	1975	200	1.50	1980	210	1.65	1985	220	1.50	1990	240	1.65
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Fig. 1. Population and water use (billion cubic meters per day) in the United States, 1950-1990 (Solley et al. 1993).

The NJDEP monitors the ecological quality of streams in the state under the Ambient Biomonitoring Network (AMNET) program. This program established 822 sampling stations statewide, where the health of instream benthic macroinvertebrate communities are evaluated using a USEP-developed statistical methodology referred to as Rapid Bioassessment Protocol (RPB). Over 500 of the AMNET stations have indicated some level of biological impairment. This assessment will examine only one of many factors responsible for impairment in these waters.

Water supply in New Jersey is closely managed and controlled under the Water Supply Management Act (N.J.S.A. 58:1a-1 et seq.), Water Supply Bond Act, State and Federal Safe Drinking Water Acts, and other statutory/regulatory authority. Rules implemented by the NJDEP Bureau of Water Allocation under the Water Supply Management Act were designed to:

Manage the State of NJ's ground and surface waters through an allocation permitting program
Ensure that an adequate supply of water is available for present and future uses
Ensure that new diversions will not adversely affect the water resources of the State or existing users of those water resources.

However, there are no specific NJ regulations involving water diversions that require protection of ecological quality. Ecological resources are not currently addressed in water diversion permits (D. Zalaskus, pers. comm.). The regulations do require minimum passing flows in streams/rivers with water diversions. These provide some limited protection of the ecological resources. However, there is no monitoring of the effects of water diversions on upstream or downstream ecological resources.

"In many areas of the country, damage to aquatic ecosystems has resulted from excessive water withdrawals. To date, there has been no hard evidence that similar situations exist in New Jersey, although clear evidence exists that water withdrawals can reduce low flows, especially in smaller watersheds." (NJDEP, 1996)

A few relevant terms concerning water use are defined below (NJWSA, 2000) :

"Consumptive water use" means the use of water in such as way that a portion of the water used is lost to evaporation, transpiration, incorporation in product, etc., and not discharged to any location.

"Critical water supply area" or "critical area" means a water supply area in which it is officially determined by the New Jersey Department of Environmental Protection, after public notice and a public meeting, that adverse conditions exist, related to the ground or surface water, which require special measures in order to achieve the objectives of the Water Supply Management Act.

"Dependable yield of combined surface/ground water sources" means the yield of water by a water system that is available continuously throughout a repetition of the most severe drought of record, without causing undesirable effects.

<p>Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function</p>	<p><i>“Depletive water use”</i> means the withdrawal of water from a water supply resource (ground or surface water) where the water, once used, is not discharged to the same water supply resource in such a manner as to be useable within the same watershed.</p> <p><i>“Drought”</i> means a condition of dryness due to lower than normal precipitation, resulting in reduced stream flows, reduced soil moisture and/or lowering of the potentiometric surface in wells.</p> <p><i>“Safe yield from surface sources”</i> means the yield maintainable by a water system continuously throughout a repetition of the most severe drought of record, after compliance with requirements of maintaining minimum passing flows, assuming no significant changes in upstream or upgradient depletive withdrawals.</p> <p>Biological integrity, biodiversity, habitat/ecosystem health, and ecosystem function were considered. Water use, meaning the withdrawal of water or alteration of water quality, results in watershed changes including changes in biological diversity and modifications to the landscape (Herrmann et al., 1998). Moyle and Leidy (1992) indicated that species diversity of fish communities decreased as human population and water use increased. Water usage by humans has led to the endangerment of freshwater fishes in several regions of the U.S. Human activities and impacts including channelization, dam building, expansion of agriculture, urbanization, erosion of river channels, pollution, logging and clearing of headwaters, and the introduction of nonindigenous species has resulted in the alteration of stream ecology including altered fish migration patterns, water temperature, nutrient levels, water chemistry and biological diversity (Warren and Burr, 1994; Herrmann et al., 1998). The development of freshwater resources has many consequences for aquatic biota, as well as terrestrial and riparian species that utilize aquatic ecosystems for food or habitat (Herrmann et al., 1998), including threatened and endangered species.</p> <p>Flow regulation can benefit nonindigenous species of fish by altering the natural predator-prey relationships in fish (in Herrmann et al., 1998), leading to changes in biodiversity.</p> <p>In estuarine/marine waters, reduced freshwater inflow can result in changes in biodiversity and biological integrity. In the Everglades, reduced freshwater flow resulted in increased salinities in Florida Bay. This caused reduced reproduction and distribution of the aquatic and terrestrial species typically found in the bay (McIvor et al., 1994).</p> <p>Herrmann et al., (1998) summarized the impacts of human use of water on aquatic systems:</p> <p>“In sum, aquatic ecosystems, possibly more than any others, have taken the brunt of human activities that are incompatible with the structure and functions of these ecosystems. Humans have derived tremendous short- and occasionally long-term economic benefits from changing aquatic ecosystems but have caused instability, massive losses of integrity that preclude natural functioning of the systems, and large reductions in species composition. None of these are short-term effects. Restorations have not returned some of these ecosystems to the degree of self-sufficiency and sustainability they possessed before human perturbation.”</p>
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	<p>Overpumping of ground water wells can lead to aquifer depletion, which in turn can impact adjacent streams, other waterbodies, and wetlands. Impacts can include reduction in water levels, loss of stream flow (i.e., dry stream), and saltwater intrusion. Water level reductions and loss of stream flow results in habitat loss for aquatic organisms and alterations to the aquatic community. The magnitude of impact will vary depending on the duration of reductions/loss (e.g., temporary, seasonal, or permanent). Intrusion of saltwater into ground water can be a serious drinking water issue since this can make the water undrinkable. Reduced or lack of stream flow can also lead to salinity increases in tidal streams. This can lead to the stress of freshwater organisms and/or alteration in the aquatic community (e.g., estuarine organisms replacing freshwater organisms).</p> <p>Reduction or lack of stream flow can have serious impacts on stream ecosystems. These can include loss of habitat (e.g., loss of pools), alteration in community structure and/or loss of biota (e.g., fish, benthic invertebrates), reduction in organic matter transport downstream (i.e., alterations in energy transfer and ecosystem function), and impacts on adjacent riparian habitat and associated wildlife. Impacts to wildlife, for example, include greater energy expenditure to reach new sources of water, loss of food, and loss of habitat.</p> <p>There are a number of stressors and factors that influence the health and quality of aquatic ecosystems. Ayers et al. (2000) studied watershed characteristics for 36 streams in NJ and Long Island and found that several were highly related to impairment of fish, aquatic-invertebrate, and algal communities. Annual peak discharge, amount of impervious road area, and population density were related to impairment in all three communities (Table 1). Changes in hydrologic factors, including decreases in base flow, increases in peak discharge and flashiness of stream flow, also were reported to influence the types and condition of aquatic communities, largely by affecting stream habitat (Ayers et al., 2000). Reductions in base flow have great influence on the suitability of streams for organisms (Klein, 1979), and maintenance of base flow was a positive characteristic in healthier aquatic invertebrate communities (Table 1; Ayers et al., 2000).</p>
Key impacts selected (critical ecological effects)	Biological integrity and biodiversity.
Exposure Assessment	
Exposure routes and pathways considered	Water overuse can occur due to current uses, increases in human population (i.e., increased density), and/or increases in water use for agricultural, industrial, and domestic or other uses. Reduced rainfall (e.g., droughts) can exacerbate the impacts of water use on aquatic and terrestrial ecosystems.
Population(s)/ecosystem(s) exposed statewide	Aquatic systems, primarily freshwater ecosystems (inland waters) and wetlands are impacted by water overuse. Estuarine waters can be impacted when drastic alterations to freshwater inflows occur, resulting in changes to water quality (e.g., increased salinity) and ecosystem structure. Adjacent riparian and terrestrial systems can also be impacted.

<p>Quantification of exposure levels statewide</p>	<p>Water withdrawals in NJ have averaged 971 billion gallons (bg) annually from 1990-1999 (Hoffman, 2001). Overall demand for potable water has increased from 124 bg (billion gallons) in 1925 to 418 bg in 1996, with a projection of 463 bg by 2010 (Hoffman & Lieberman, 2000, CH2M Hill et al., 1993). Withdrawals from surface water accounts for 75% of total water withdrawals and 59% of potable water withdrawals in NJ. On average (from 1990-96), approximately 30 billion gallons per year (bgy) of water is withdrawn from reservoirs, 245 bgy from ground water, and 695 bgy from rivers (includes 275 bgy from the Delaware River). (Hoffman & Lieberman, 2000)</p> <p>In general, southern New Jersey is more dependent on ground water supplies and northern New Jersey is more dependent on surface water (NJSWA, 2000). Table 2 (see end of assessment) lists the total water withdrawals by water region and water source for 1990-1999 in NJ, while Table 3 lists the size of each water region (Hoffman, 2001). The Lower Delaware region has the largest surface water and ground water withdrawals and is the 2nd largest region in size. The smallest region in the state (Passaic) had the 2nd largest surface water and total water withdrawal in the state. The largest region in the state (Atlantic Coastal) had the second highest ground water withdrawals, but was ranked 4th in total withdrawals.</p> <p>Table 4 lists surface water, ground water and total water withdrawals for the period 1990-99 (average annual withdrawals) by county. Hunterdon, Morris, Passaic, Somerset and Warren Counties export the most water in the state. Bergen, Essex, Hudson, Middlesex, and Union Counties import the most water (Hoffman, 2001; Hoffman and Lieberman, 2000). (See notes and Figure 2 at end of assessment).</p> <p>Factors such as rainfall, season, temperature, and the economy all affect water use. Potable, irrigation and agricultural water demands have a strong seasonal component, with more water needed during the growing season (Hoffman & Lieberman, 2000). For example potable water use includes lawn watering during the warmer months, whereas agricultural includes crop and golf course watering. Water use is inversely related to rainfall (e.g., more water is used to water crops when there is less rainfall). For each inch of rainfall greater than 8 inches, agricultural withdrawals decreased by 1 bg (Hoffman & Lieberman, 2000).</p> <p>Consumptive uses and out-of-basin diversions (i.e., depletive use) have the most impact on downstream areas since the water is not returned to the source stream (i.e., lost from the stream). “The larger the depletive water use, the greater the potential for negative impacts to these resources as a result of reduced water flow through the natural water systems.” (NJDEP, 1996)</p> <p>Information from the Raritan Basin Watershed Management Project provides an example of a water budget for this region of the state. This project indicates that the Raritan River Basin currently has a water supply availability of 360 mgd (million gallons per day), a demand (in 1990) of 225 mgd, and a predicted need of approximately 350 mgd by 2040. Therefore, there is an adequate supply currently available. It was estimated that more than 900 billion gallons of water in the form of precipitation falls on the basin each year (based on 47 inches of rain). Over half of the precipitation (53%) evaporates from land or water surfaces, while 30% is runoff to streams or water bodies, and only 17% infiltrates into the ground (i.e., ground water recharge). Increases in impervious surfaces (e.g., due to development) decreases infiltration and leads to higher flows after storms, and lower stream flows during dry weather (e.g., less ground water discharge to streams).</p>
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“Continued development of the Basin is likely to require construction of an additional water supply in the first half of the 21st Century. Development, in turn, may have impacts on ground water availability (through recharge reduction), baseflow of streams, safe yields and water quality.” (NJWSA, 2000)

The U.S. Geological Survey maintains a stream gage network in the U.S. including NJ. More than 80 stream gauging stations are located in NJ, and most of these measure both stream flow and gage height. These gages are used for monitoring of passing flows.

Passing flows can be established to provide water for downstream users (e.g., other water diversions), and to control salinity in the estuary (e.g., reservoir releases to the Delaware River). Passing flows for streams with water diversions are specified or set by the NJDEP (N.J.A.C. 7:19-4.6(f)), and are equal to the average daily flow for the driest month based on existing records, or 125,000 gallons/square mile of unappropriated watershed above the point of diversion. In the Raritan River Basin, ten minimum passing flows have been established. The entity responsible for ensuring compliance with the minimum passing flows is also identified (NJSWA, 2000).

Waterbody	Minimum Passing Flow	Entity
Carnegie Lake	5.5 MGD	Princeton University
Stony Brook	0.65 MGD	Hopewell Valley Golf Club
Millstone River at Blackwell’s Mill	32.2 MGD	NJ Water Supply Authority
South Branch Raritan River at Stanton	40 MGD*	NJ Water Supply Authority
Lamington River	9 MGD	Belle Mead Development
Raritan River at Manville	70 MGD*	NJ Water Supply Authority
Raritan River at Bound Brook	90 MGD*	NJ Water Supply Authority
Lawrence Brook below Weston’s Mill Pond	5.6 MGD	City of New Brunswick
Deep Run (tributary to South River)	1.4 MGD	Middlesex Water Company
Matchaponix Brook	4.7 MGD	United Water-Matchaponix

* flow augmentation is necessary

Specific population(s) at increased risk

The following areas have been identified as having water supply deficits (Source: NJDEP, 2002):

Water Supply Critical Areas: The State of New Jersey has designated two areas of water supply concern. These are areas where excessive water usage poses significant threat to the long-term integrity of a water supply source.

Critical Area No. 1 was declared in 1985 and includes portions of Middlesex, Monmouth and Ocean Counties. There are four affected aquifers in this critical area: the Englishtown, the Upper and Lower Potomac-Raritan-Magothy, and the Wenonah –Mt Laurel. Water allocations were reduced 40-50 percent within this critical area.

Critical Area No. 2 was declared in 1994 and includes portions of Burlington, Camden, Gloucester, Atlantic, Cumberland, Salem, Monmouth and Ocean Counties. Water allocations from the Potomac-Raritan-Magothy aquifer system were reduced an average of 22 percent within this region.

	<p>According to the New Jersey State Water Use Plan (NJDEP, 1996), 9 of the 23 Regional Water Resource Planning Areas' (RWRPA) water supplies are at heightened risk due to estimated supply deficits. Based on supply estimates for 1990 and 2010, these include (see also Table 5):</p> <p>Camden/Delaware Tributaries – 70 mgd supply deficit, although existing regulations on aquifer withdrawals are expected to solve this problem.</p> <p>Mullica River region – estimated water supply deficit of 56 mgd. By 2010 projected to increase to 84 mgd. There is some uncertainty with the estimated deficit due to uncertainties regarding the amount of agricultural surface water withdrawals. If estimates of ground water are accurate, overpumping of the Cohansey aquifer may be occurring, resulting in adverse effects such as baseflow reductions in the Mullica River or the inducement of the salt front into the estuaries.</p> <p>South River area – water supply deficit of 27 mgd, projected deficit of 42 mgd by 2010.</p> <p>Metedeconk River and Tom's River areas – deficit of 20 mgd; projected to increase to 34 mgd by 2020. High peak demands in summer may cause streamflow depletion. All regional wastewater treatment plants discharge to the ocean, representing a large scale depletive water use which may have long-term impacts on water supplies. Modeling results indicated that groundwater withdrawals resulted in average base flow depletion of up to 11 percent of predevelopment base flow in some streams.</p> <p>Maurice River area – 8 mgd deficit; projected 18 mgd by 2010. Streams in southern New Jersey derive a large percentage (approximately 80-90%) of their total annual flow from ground water. For this reason, aquifer overuse could result in baseflow reductions.</p> <p>Hackensack River area – small deficit, being remedied without major capital expense.</p> <p>Cape May coastal area – currently has a water supply surplus; current trends indicate a deficit is likely by 2010. Saltwater intrusion is a problem in this area.</p> <p>Lower Passaic/Rahway area – deficit of 9 mgd.</p> <p>In planning area 4 (Upper Passaic River) ground water investigations have indicated significant declines in the water table and stream flow reductions in reaches where ground water is depletively withdrawn (NJDEP, 1996). Potential reductions in stream baseflow were also identified for the Mullica River (Area 18), Metedeconk River (Area 15), Toms River (Area 16), and Maurice River (Area 21) (NJDEP, 1996).</p> <p>In addition, there are potential impacts to streams in areas not anticipated to have water supply deficits by 2010. The expansion of sewers can result in depletive water use and stream flow reductions. The 1996 Statewide Water Supply Plan recommended that “depletive sewerage projects be carefully scrutinized” and wastewater be discharged near withdrawal locations to “compensate” for expected stream flow reductions. These recommendations included RWRPA areas 1, 2, 3, 7, 8, and 9, all in western/northwestern NJ.</p> <p>Both RWRPA 19 (Atlantic Coastal) and 22 (Great Egg Harbor River) were identified as areas sensitive to saltwater intrusion and/or have the potential for stream flow depletion due to potential future depletive uses (e.g., development, sewerage) (NJDEP, 1996).</p>
<p>Quantification of exposure levels to population(s) at increased risk</p>	<p>Qualitative estimates were made using the current and future water supply deficits, depletive use, and water supply usage. The estimate assumed that water withdrawals from either surface water or ground water would similarly impact ecological systems.</p>

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Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	No quantitative methods were employed. Quantitative data are lacking in NJ for the number of miles or streams or acres of habitat that are potentially impacted by water use/overuse. Qualitative estimates were made based on current and projected water supply usage, depletive use, and water supply deficits (see Table 5 for ranking of potentially impacted areas).
Risk Characterization	

<p>Risk estimate(s) by population at risk</p> <p>Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)</p> <p>Assessment of severity/irreversibility</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage: <ul style="list-style-type: none"> • many species threatened/endangered • major community change • extensive loss of habitats/species Long time for recovery</p> <p>3 - Adverse affect on structure and function of system: <ul style="list-style-type: none"> • all habitats intact and functioning • population abundance and distributions reduced Short time for recovery</p> <p>2 – Ecosystem exposed but structure and function hardly affected</p> <p>1 – No detectable exposure</p>	<p>Inland waters, wetlands, and riparian habitat will most likely be impacted to the greatest extent by water overuse. Many of the major rivers and streams are monitored and managed for minimum passing flows, which should provide some level of protection for the associated ecosystems. Smaller streams and tributaries that are not directly managed are probably at increased risk. For example, water overuse may reduce the flows in permanent streams, and reduce the number of days that intermittent streams flow resulting in impacts to aquatic and semi-aquatic species (e.g., benthic invertebrates and amphibians). Wetlands dewatering could potentially occur due to water overuse. Changes in the salt front and saltwater intrusion could occur in coastal areas impacting associated biological communities.</p>	<p>Score</p> <p>2-3</p>
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	<p>It is estimated that Statewide the severity/irreversibility score ranges from 1 to 4: Inland waters: 2-4 Wetlands: 2-4 Grasslands: 1-2 Marine water: 2-3 Forests: 1-2.</p>	
<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade) 5 – Often and increasing 4 – Often and continuing 3 – Occasional 2 – Rare 1 - Possible in the future 0 – Unlikely (or 0.1)</p>	<p>Statewide the Frequency of effects ranges from “rare” (2) (terrestrial systems) to “often and continuing” (4) in some aquatic systems.</p>	3
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude) 5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>	<p>It is difficult to estimate the extent of the state that is affected. Factors such as rainfall, season, temperature, and the economy all affect water use. Since only some of the state’s habitats are impacted by water use (primarily inland waters and wetlands) it is estimated that 5-10% of the state is impacted (2). It is expected that this will increase as water use increases in the future.</p>	2
	Total	12-18
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>M – There is some uncertainty with the data used in this assessment (e.g., water deficits, depletive values, total available water). The data were estimated and data confidence varies for each area due to differences in watershed characteristics and quality of data. A comprehensive survey of instream flow requirements and comparison with seasonal flows is needed to assess the current status of water use on New Jersey’s streams. Impacts on wetlands should also be studied.</p>	
<p>Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief</p>	<p>M – A USGS and NJDEP research project is underway to examine the flow characteristics and basis for developing ecological flow goals and methodologies for NJ streams. Data from this project may help define the current risk and</p>	

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description. (Data Gaps; highlight significant data needs)	impacts of water overuse in the state.
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, !, =, ≡; where + is improvement), and brief description.	“! ” Increased water withdrawal associated with continued development (e.g., sprawl) and population increases in the state will lead to additional impacts. This may be partially offset by potential benefits from including biological needs in water allocation permits.
Potential for catastrophic impacts* (H,M,L) and brief description (*Short-term drastic negative impacts having widespread geographic scope)	L –Impacts are expected to be periodic or gradual (as water use increases). Drought conditions can exacerbate the effects of water overuse leading to more serious short-term widespread impacts.

<p>Link to other Work Groups (e.g., socioeconomic impacts)</p> <p>Extent to which threat is currently regulated or otherwise managed</p>	<p>Water overuse can lead to economic costs including increased costs of developing new water supplies and associated infrastructure (e.g., reservoirs and supply interconnections), increased costs of water treatment, and potential loss of recreational opportunities due to reduced/lack of water flow/water levels.</p> <p>The USGS, NJDEP, NJ Water Supply Authority, and other water suppliers monitor water levels including stream flows. The NJDEP can issue a drought watch or drought warning based on rainfall, stream flows and ground water levels. Then DEP Commissioner Bob Shinn issued a drought warning for the Northwest, Southwest and Coastal South regions of the state on November 21, 2001 due to a rainfall deficit and declines in stream flows and ground water levels. If conditions worsen, the Governor can declare a water emergency for the state, as Governor McGreevey did on March 4, 2002 by issuing an Executive Order. This authorized DEP Commissioner Bradley Campbell to issue mandatory water-use restrictions and conservation measures in March 2002 through the signing of an Administrative Order. These actions give DEP greater authority to control water distribution and transfers among the major reservoir systems, to temporarily modify water allocation permits, and direct water usage and water conservation. Water allocation permits are issued by DEP, however, the biological needs of the aquatic system are not currently included. The Delaware River Basin Commission (DRBC) monitors and regulates flows in the Delaware River.</p> <p>Water supply in NJ is regulated by a number of laws and regulations including: Water Supply Management Act of 1981 (mandated a state planning effort) Water Supply Bond Act of 1981 (created a \$350 million bond fund for capital improvements to water supply) Safe Drinking Water Act of 1977 Water Supply Authority Act (established the NJ Water Supply Authority)</p> <p><u>NJDEP Water Resources Management Section, Bureau of Water Allocation.</u> The section regulates all ground and surface water diversions in New Jersey that are in excess of 100,000 gallons of water per day. This includes water diverted for public water supply, industrial processing and cooling, irrigation, sand and gravel operations, remediation, and power generation.</p> <p>Water Supply Planning: The Water Resources Management Section works with the United States and New Jersey Geological Surveys, the Delaware River Basin Commission, the New Jersey Water Supply Authority and the Department of Environmental Protection's Division of Watershed Management to conduct water supply feasibility studies and water resource evaluations. Such studies are used to evaluate long term water supply planning needs and to provide viable alternatives through the Statewide Water Supply Plan.</p>
	<p>A new water supply law (P.L. 2001, c. 165) authorized the assessment of water supply in the Pinelands and Cape May County. The Pinelands study will be a multi-agency study to evaluate current and projected water supply needs. The law requires the needs be balanced against protection of the area's ecology and the Kirkwood-Cohansey aquifer. The Cape May study will assess the sustainability of water supply options for current and future needs. Increased water withdrawals will only be allowed if they will not accelerate saltwater intrusion, lower existing stream flow or injure the ecosystem. (Sinclair, et al., 2001)</p>
<p>Barriers to restoration</p>	<p>Competing uses and increased usage of water are a significant barrier. Drought conditions, as they occur, exacerbate water supply/water use impacts and make solutions more difficult.</p>

	Recommendations for management include applying techniques that increase recharge and otherwise reduce runoff to increase stream base flow and reduce streamflow variability (Ayers et al., 2001).
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	Note: NJ water withdrawal data for the period 1990-96 were used to rank the uses of water in the state (and percent of total use) (from data in Hoffman & Lieberman (2000))
Large business/industry	H – power generation – ranked #2 for water withdrawals in NJ (36%). Industrial withdrawals ranked #3 (10%).
Small business industry	M – Industrial withdrawals ranked #3 (10%).
Transportation	L
Residential	H – Potable water ranked #1 for water withdrawals in the state (45%).
Agriculture	M – Ranked #4 for water withdrawals in the state (5%).
Recreation	L – Irrigation (e.g., golf course irrigation) ranked #6 in the state (0.4%).
Resource extraction	L-M – Mining was #5 in water withdrawals (3%).
Government	H – Public water supplies (potable water ranked #1; 45%).
Natural sources/processes	H – the majority of water that reaches the state through rainfall is lost from natural processes (e.g., evaporation, transpiration, and river flow to coastal saline waters)
Orphan contaminated sites	L
Diffuse Sources	
Sediment sinks	NA
Soil sinks	NA
Non-local air sources incl. Deposition	NA
Biota sinks	NA

Summary Statement: Water use/overuse can have profound impacts on ecosystems from which the water was withdrawn or withheld. Water use results in watershed changes including changes in biological diversity and modifications to the landscape. Water overuse can lead to loss or reduction in stream flow, saltwater intrusion, changes in estuarine salinities and associated loss of habitat, and loss of biological diversity and biological integrity. Data on direct impacts to ecosystems in New Jersey are sparse. Indirect data such as depletive water use, total water usage, and water supply deficit estimates were used to estimate impacts to New Jersey’s ecosystems. Several regions and ecosystems of the state are at potential risk. Ecosystems include inland waters, wetlands, and marine (estuarine) waters. Potential reductions in stream baseflow were identified for the Mullica River, Metedeconk River, Toms River, and Maurice River. In planning area 4 (Upper Passaic River) ground water investigations have indicated significant declines in the water table and stream flow reductions in reaches where ground water is depletively withdrawn. In addition, there are potential impacts to streams in areas not anticipated to have a water supply deficit by 2010. The expansion of sewers can result in depletive water use and stream flow reductions. The 1996 Statewide Water Supply Plan recommended that “depletive sewerage projects be carefully scrutinized” and wastewater be discharged near withdrawal locations to “compensate” for expected stream flow reductions. These recommendations included RWRPA areas 1, 2, 3, 7, 8, and 9, all in western/northwestern NJ. Both RWRPA 19 (Atlantic Coastal) and 22 (Great Egg Harbor River) were identified as areas sensitive

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to saltwater intrusion and/or have the potential for stream flow depletion due to potential future depletive uses (e.g., development, sewerage). Overall the statewide risk is deemed “Low-Medium”, while the risk is estimated to be “Medium” for inland waters and wetlands.

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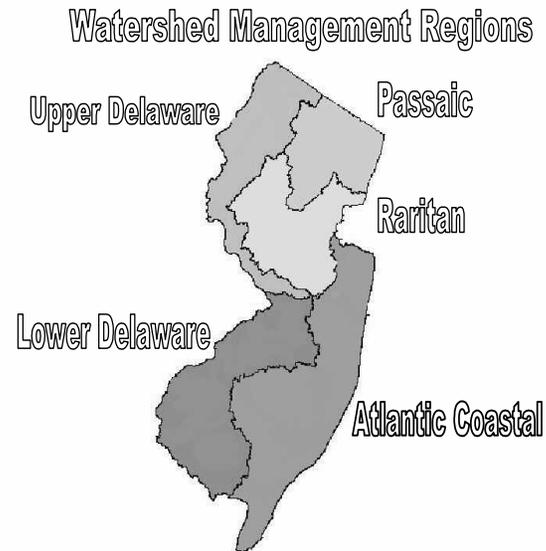
Statewide Analysis of Threat **Threat = Water Overuse**

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	2-4	3-4	3	33 (18-48)
Marine Waters	2-3	3	2	15 (12-18)
Wetlands	2-4	3-4	3	33 (18-48)
Forests	1-2	2	1	3 (2-4)
Grasslands	1-2	2	1	3 (2-4)
Total Score				87
Average Score (Total ÷ 5)				17.4

Risk by Watershed Management Region

THREAT = Water Overuse	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L-M	NA	L-M	L	L
Passaic	L-M	L	L-M	L	L
Raritan	M	L	M	L	L
Atlantic	M-H	M	M-H	L	L
Lower Delaware	M-H	M	M-H	L	L
Region/Watershed (secondary)					
Urban	M	L-M	M	L	L
Suburban	M	L-M	M	L	L
Rural	M	L-M	M	L	L

H=high, M=medium, L=low, NA = not applicable



Notes from Hoffman (2001):

Withdrawals are reported on the basis of water source, not the location of eventual water use. For example, Hudson County is supplied almost entirely by surface water withdrawn in Passaic and Morris Counties (Nawyn and Clawges, 1995). Thus, water consumed in Hudson County is reported under Passaic and Morris counties in this report. In general, those counties and watersheds from which water is exported show withdrawals that are disproportionately high for their population. Conversely, counties which import water report water withdrawals that are disproportionately low for their population.

Water withdrawn is not equivalent to water consumed. Many purveyors, especially thermoelectric generators and in particular instream hydroelectric plants, are wholly or partially nonconsumptive. In 1988 approximately 79% of surface-water use was nonconsumptive (Saarela, 1992). Thus, the reported volume of surface-water withdrawals does not imply an equivalent reduction in stream-flow volumes.

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Table 1. Environmental factors that were highly related to impairment of fish, aquatic-invertebrate, and algal communities along an urban land-use gradient. [NS, No statistically significant effect on aquatic community] (From Ayers et al., 2000)

Watershed characteristic	Response of aquatic community		
	Fish	Aquatic invertebrates	Algae
Area of forest and wetlands	NS	Positive	NS
Ability to maintain base flow	NS	Positive	NS
Percentage of cobble substrate	Positive	Positive	NS
Median sulfate concentration	NS	Positive	Positive
Median total phosphorus concentration	Negative	NS	Positive
Mean annual flood	Negative	Negative	Negative
Flashiness of streamflow	Negative	NS	NS
Impervious area, road area only	Negative	Negative	Negative
Impervious area, nonroad area only	NS	Negative	NS
Population density	Negative	Negative	Negative
Total urban area in 1986	Negative	NS	NS
Urban area growth from 1986 to 1995	NS	Negative	NS
Commercial and industrial area in 1986	NS	Negative	Negative
Total point-source flow	NS	Negative	NS

Table 2. Total water withdrawals in 1990-1999 by water region and water source (millions of gallons) (Hoffman, 2001)

Water region	Average total water withdrawals 1990-1999 (millions of gallons)		
	Source		
	Surface Water	Ground Water	Total
Passaic	234,437	40,646	275,083
Raritan	46,615	39,399	86,014
Atlantic Coastal	47,036	65,898	112,934
Upper Delaware	134,099	21,730	155,830
Lower Delaware	263,598	77,486	341,084
Totals	725,785	245,159	970,945

Table 3. Size of water regions (Hoffman & Lieberman, 2000)

Water Region	Area (mi ²)	Percent of state
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Passaic	953	12.2%
Raritan	1,284	16.5%
Atlantic Coastal	2,453	31.5%
Upper Delaware	1,226	15.7%
Lower Delaware	1,872	24.0%

Table 4. Average Total Water Withdrawals in 1990-1999 by County and Water Source (millions of gallons)
 (Hoffman, 2001)

County	Source		Total
	Surface Water	Ground Water	
Atlantic	1,311	17,178	18,488
Bergen	35,789	10,973	46,762
Burlington	54,456	25,612	80,068
Camden	1,346	25,170	26,516
Cape May	864	9,388	10,252
Cumberland	17,501	15,780	33,281
Essex	19,839	9,477	29,316
Gloucester	18,592	14,725	33,317
Hudson	0	176	176
Hunterdon	68,963	5,175	74,137
Mercer	203,832	5,342	209,175
Middlesex	1,556	18,827	20,383
Monmouth	17,793	9,340	27,133
Morris	19,414	22,613	42,027
Ocean	2,286	21,926	24,212
Passaic	16,0736	4,787	165,523
Salem	6,253	4,772	11,026
Somerset	40,125	4,028	44,154
Sussex	571	6,520	7,091
Union	3,626	5,263	8,889
Warren	50,929	8,118	59,047
Yearly total:	725,787	245,190	970,973

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Table 5. Comparison of Net Depletive/Net Gain Water Use for Regional Water Resource Planning Areas (RWRPA) and Estimated and Projected Water Supply Surplus or Deficit for 1990 and 2010 (Zripko and Hasan, 1994). The RWRPA are ranked by potential impact due to water overuse (from most impacted to least by the sum of the four columns).

Regional Water Resource Planning Areas	Description	Surface Water Net Depletive (-) or Net Gain (+) (MGD)	Ground Water Net Depletive (-) or Net Gain (+) (MGD)	1990 Surplus (+) or Deficit (-) (MGD)	2010 Projected Surplus (+) or Deficit (-) (MGD)
18	Mullica River	-20.9	-48.6	-55.8	-84.0
17	Camden Delaware Tributaries	-0.4	+0.2	-69.8	-79.6
11	South River	-0.8	-34.0	-27.2	-41.9
16	Toms River	-7.9	-15.4	-16.2	-26.8
5	Lower Passaic River – Rahway River	-17.0	-22.1	-9.0	+7.4
21	Maurice River	-1.1	-10.3	-8.5	-17.8
15	Metedeconk River	-1.2	-13.9	-3.6	-7.6
12	Navesink/Swimming River	-29.6	-20.1	+13.5	+13.4
6	Hackensack River	+5.5	+0.5	-7.9	-15.8
23	Cape May Coastal	-0.2	-10.9	+4.1	-0.8
20	Salem River	-12.8	-9.9	+12.4	+9.1
4	Upper Passaic River & Tributaries	-226.7	-16.3	+127.9	+121.0
10	Raritan River	-132.7	-17.9	+95.5	+71.9
14	Rancocas River	-9.4	-0.5	+35.5	+16.3
	Remaining RWRPA (9)	-0.1	-18.8	+165.7	+143.5
	Statewide Totals	-455.2	-238.0	+256.6	+108.4

MGD = million gallons per day

Average Freshwater Withdrawals in New Jersey, 1990-1999, by County and Source of Water

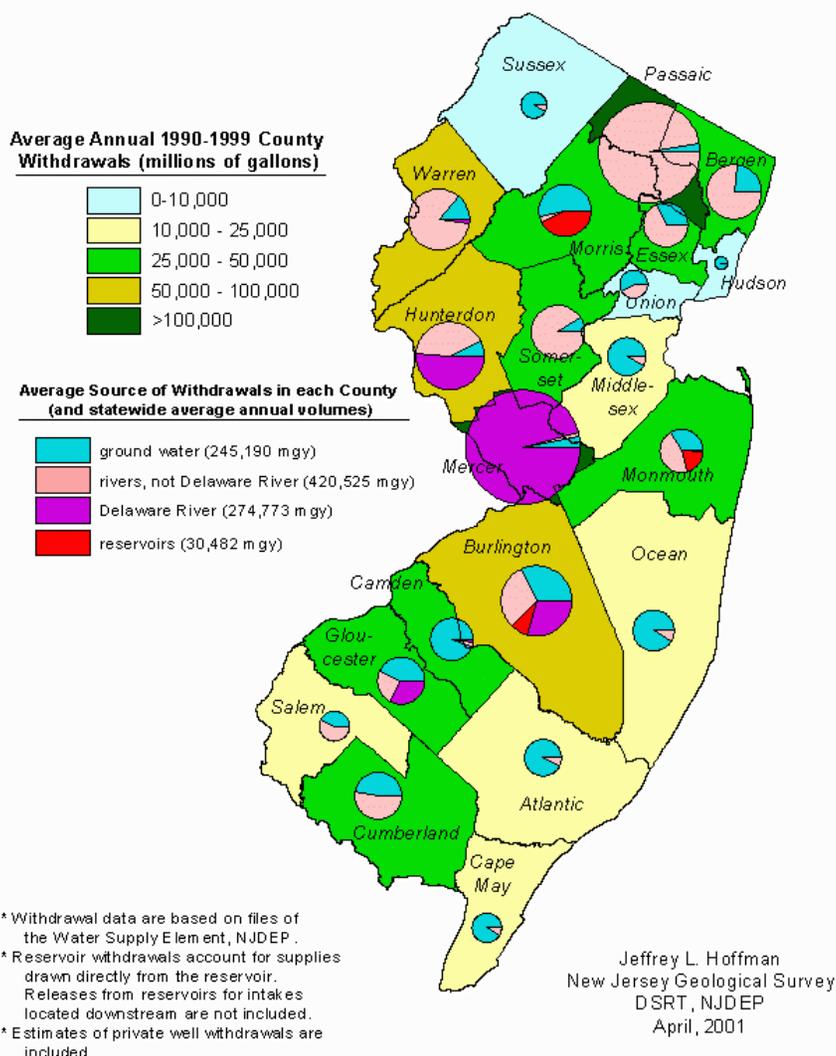


Figure 2. (source: Hoffman, 2001)

Issue: West Nile Virus
Author: Mary Downes-Gastrich
Version:

New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Stressor:

West Nile Virus

Issue: Data for NJ indicate presence of WNV in crows in several counties

Overall Ranking: Medium-Low

Severity: 2 - Ecosystem exposed but structure and function hardly affected

Frequency: 3– Occasional

Size of Population/size of state habitat affected: 3 -because 30% of the state has shown WNV activity.

Confidence/Uncertainty: Medium level of certainty concerning the threat

West Nile Virus (WNV) can cause a form of encephalitis in humans, which is an infection which causes a swelling or inflammation of the brain. In some cases, if not treated quickly, the infection can result in death (Burlington County Health Department, 2000). The West Nile Virus (WNV) is a flavivirus, an RNA virus, which is approximately 40-60 nm, spread by the bite of the infected mosquito, *Culex pipiens* and possibly, other species (e.g., *Culex restuans*, *Culex salinarius*, *Aedes japonicus*, *Aedes vexans*). The virus can be transmitted to people and animals when being bitten by a mosquito infected with the virus (Burlington County Health Department, 2000; Center for Disease Control, 2000). WNV does not appear to cause extensive illness in dogs or cats and no documented evidence of person-to-person or animal-to-person transmission of the virus.

Geographically, the WNV has been diagnosed in birds from NYC, NYS, NJ, CT, RI, MA and MD (USGS, 2000b). Recent crow deaths suggest that WNV emerged in NY in late August, 1999 and could be more deadly to North American bird species (e.g. American and Fish Crows) than to species in Africa, the Middle East and Europe, where the virus is normally found (USGS, 2000c). It was reported that the WNV caused deaths in birds in the Bronx zoo in 1999, including 20 flamingoes, herons, and bald eagles (USGS, 2000a). Over 190 bird specimens (mostly American crows) have tested positive for WNV (Center for Disease Control, 2000). Wild birds infected with WNV can die or become ill when infected but there is no evidence that people handling dead birds will become infected. But there are precautions when handling these birds (Center for Disease Control, 2000).

1999

In 1999, the WNV had human health and ecological impacts. WNV was responsible for the outbreak of encephalitis causing seven deaths in people from the New York Metropolitan area, and causing illness in another 55 (Eidson and Huntley, 2000; Morbidity & Mortality Weekly Report, Center for Disease Control Prevention, 1999). The Center for Disease Control (CDC) in Fort Collins, Co. confirmed that 17 native bird species have tested virus positive for WNV; CT has added a Cooper's Hawk as the 18th species (USGS, 2000b). In 1999 in NJ, the WNV was detected in birds from 16 of 21 counties, with the majority in the north central part of the state (Faye Sorhage, NJDHSS, pers. comm.). There were horse mortalities in 1999 in Long Island (Doug Roscoe, pers. comm.; Center for Disease Control, 2000).

2000

WNV has re-emerged in NJ. As of August 31, 496 crows have tested positive in similar areas as 1999, with positive mosquito pools in Bergen County. One case of human illness in a Jersey City resident occurred with the onset on August 6. There have been eight cases of human WNV in NYC, with most cases in Staten Island. The CDC confirmed the virus in mosquitoes in NYC and CT (USGS, 2000). Positive crows and/or mosquito pools have also been found in upstate NY and in the states of MA, CT, RI, and NH.

The NJ Department of Health and Senior Services, NJ Department of Environmental Protection and the NJ Dept. of Agriculture have developed a comprehensive WNV plan that includes enhanced mosquito control efforts, increased human, animal and mosquito surveillance, a system for monitoring and testing, and a public education program (NJ Department of Health and Senior Services, 2000).

Issue: West Nile Virus
Author: Mary Downes-Gastrich
Version:

For more information on WNV, contact:

Dr. Doug Roscoe, Supervisor of NJDEP's Bureau of Fish and Wildlife Health and Forensics at (908) 735-6390.

Dr. Faye Sorhage, State Veterinarian with the NJ Department of Health and Senior Services at (609) 588-3121.

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New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	Zebra Mussels (<i>Dreissena polymorpha</i>)
Description of stressor	
Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	<p>Biological Integrity:</p> <p>Biodiversity: Epizoic colonization (infestation) of snails, crayfish and especially unionid bivalve molluscs is believed to be the most direct and ecologically destructive characteristic of zebra mussels. All infested unionid species (23) in the Great Lakes have become infested and exhibit mortality within two to four years after heavy infestation (Schloesser, Nalepa and Mackie, 1996). A key impact would be on the fish population of an infested body of water. The expected reduction of phytoplankton and zooplankton would have a negative effect on larval fishes as well as the young of pelagic species because of their dependence on those groups as a food source. Also a number of species whose adults depend on the benthic substrate and its associated food sources such as insects and other invertebrates will prosper and outcompete those species that are pelagic in their existence such as alewives, walleyes and trout.</p> <p>Ecosystem function: Zebra mussels dramatically affect the food web structure and habitat due to their efficiency as filter feeders. In the Hudson River zebra mussels filter all the water in the tidal-freshwater part of the river every two to three days. Prior to the invasion all other filter feeders combined filtered the water about once every 50 days. Phytoplankton have declined by about 90%, causing a change in photosynthesis. The smaller zooplankton have declined in a way similar to the phytoplankton. Planktonic bacteria have increased in abundance presumably as a result of reduced predation from the small zooplankton.</p> <p>Water clarity has increased dramatically from 0.6 meters to 1.2 meters at the depth of the 1% light level. This may have increased the growth of benthic plants (Hudson River Ecosystem Study). Invasion of shallow lakes and ponds may divert production and biomass from pelagic to benthic food webs, shifting ecosystems to an alternative state (MacIsaac, 1996).</p>
Key impacts selected (critical ecological effects)	Extinction of Unionid mollusks Habitat modification Trophic level effects

Exposure Assessment	
Exposure routes and pathways considered	<p>Adult mussels attach to solid substrates and can be unintentionally transported overland by recreational boaters.</p> <p>Commercial barges have transported zebra mussels from the Great Lakes throughout the Mississippi River system all the way to New Orleans).</p> <p>Free living larvae or veligers are microscopic and can be transported by virtually any removal of water. Any object or samples collected in zebra mussel area should be assumed to contain veligers. They are pelagic and disperse effectively with current (www.nsgo.seagrant/research/nonindigenous/zmlifehistory.html).</p>
Population(s)/ecosystem(s) exposed statewide	None currently (see below)
<p>Quantification of exposure levels statewide</p> <p>Specific population(s) at increased risk</p>	<p>None possible at this time</p> <p>Unionid bivalve molluscs and other lifeforms which are subject to attachment by zebra mussels.</p> <p>Phytoplankton “Although freshwater mussels are most diverse in North America, where there are 281 species and 16 subspecies, they are among the most rapidly declining animal groups on the continent. The Nature Conservancy recognizes 55% of North America's mussel species as extinct or imperiled compared to only 7% of the continent's mammal and bird species. In North America above Mexico, there are 19 taxa presumed extinct, 44 species listed or proposed as federally endangered, and 60 species that may be endangered. In addition, a number of the endangered species are functionally extinct, with individuals of a species surviving but not reproducing. Freshwater mussel extinctions and declines can be attributed to several factors, including habitat degradation, construction of dams and subsequent loss of host fishes, and expansion of exotic mollusks such as the zebra mussel and Asian clam.”</p> <p>New Jersey is home to twelve native species of freshwater mussels, including the federally and state endangered dwarf wedgemussel (<i>Alasmidonta heterodon</i>) and three federal species of special concern - the brook floater (<i>Alasmidonta varicosa</i>), green floater (<i>Lasmigona subviridis</i>) and yellow lampmussel (<i>Lampsilis cariosa</i>). Although the dwarf wedgemussel was thought to be extirpated in the state, findings of live individuals in the Paulins Kill and Pequest River have prompted intensive surveys of the waterways.</p>
	<p>The brook floater is only known from small occurrences in four areas of the state, whereas the yellow lampmussel is restricted to the Delaware River. The green floater is by far the most endangered mussel in the state, represented only by a single individual in the Stony Brook. The brook floater and green floater are now being listed as state endangered. The yellow lampmussel, along with the eastern lampmussel, eastern pondmussel, tidewater mucket (<i>Leptodea ochracea</i>), and triangle floater (<i>Alasmidonta undulata</i>) will have a state status of threatened (slated for late 2001).” (NJDEP DFW, 1999; Personal comm., J. Bowers-Altman, 2001)</p>
Quantification of exposure levels to population(s) at increased risk	

Issue: Zebra Mussels
 Author: Tim Casey
 Version:

Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	N/A
Risk Characterization	
<p>Risk estimate(s) by population at risk</p> <p>Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)</p> <p>Assessment of severity/irreversibility</p> <p>5 - Lifeless ecosystems or fundamental change; Irreversible</p> <p>4 - Serious damage:</p> <ul style="list-style-type: none"> • many species threatened/endangered • major community change • extensive loss of habitats/species <p>Long time for recovery</p> <p>3 - Adverse affect on structure and function of system:</p> <ul style="list-style-type: none"> • all habitats intact and functioning • population abundance and distributions reduced <p>Short time for recovery</p> <p>2 - Ecosystem exposed but structure and function hardly affected</p> <p>1 - No detectable exposure</p>	<p>Score</p> <p>4</p> <p>Severe ecological damage via change in ecosystem function (productivity and biomass), alteration of community structure and foodweb dynamics. Local extinction of molluscs.</p>

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<p>Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade)</p> <p>5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)</p>	<p>Since it was accidentally introduced into the Great Lakes (most likely by discharge of larvae in ballast water) into Lake St. Claire in 1986 the zebra mussel has invaded 20 of 38 states East of the Rocky Mountains (Ram and McMahon, 1996). It is already well established in the Hudson River and the headwaters of the Susquehanna river. Maryland researchers project that it will invade the Chesapeake Bay within a year (www.mdsg.umd.edu/extension/zebramussel.html). It is probable that zebra mussels will pose a significant threat in the near future.</p>	<p>1 (4)</p>
<p>Size of population(s) and/or extent of the State/habitat affected (magnitude)</p> <p>5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted</p>		<p>3</p>
	<p>Total*</p> <p>*This score is based on the assumption that Zebra mussels are not presently found in New Jersey. Once established in our watersheds the frequency category could change to 4 , yielding a new score of 48</p>	<p>12 (48)</p>
<p>Assessment of uncertainties in this assessment (H,M,L) and brief description</p>	<p>Medium: The uncertainty lies in when and if zebra mussels will invade New Jersey watersheds. Assessment of the uncertainty of the effects of the invasion is low as these effects are well documented by a number of states.</p>	
<p>Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief description. (Data Gaps; highlight significant data needs)</p>	<p>Low. The zebra mussel and its environmental consequences are well studied. It is unlikely that new data obtained in New Jersey will significantly change the risk estimate. However, recent study by the New York State Museum has found a bacterial species that appears to kill zebra mussels without harming species of fish (Star-Ledger, 2001). This may be an alternative control method to using harsh chemicals.</p>	

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Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, !, =, ≡; where + is improvement), and brief description.	≡: The zebra mussel's rapid spread throughout the US has not been checked. In only 14 years it has spread throughout the Eastern and Central US. It seems very likely that it will soon invade New Jersey.
Potential for catastrophic impacts* (H,M,L) and brief description (*Short-term drastic negative impacts having widespread geographic scope)	M: Immediate short-term impacts are low due to the perceived absence in New Jersey. However, if the bivalve does become established in the near future, impacts would be rapid and severe to aquatic systems (rated H). Combined potential equates to a M rating.
Link to other Work Groups (e.g., socioeconomic impacts)	Severe damage to power plants, water treatment facilities, irrigation systems and municipal water delivery systems due to large concentrations of zebra mussels clogging intake and outflow pipes. An estimate of cost of scraping mussels from pipes in the Great Lakes region is 50 to 100 million dollars per year (Maryland Sea Grant Extension).
Extent to which threat is currently regulated or otherwise managed	The zebra mussel is currently not under control. NJ has a Zebra Mussel Watch program that requests public assistance in reporting zebra mussel sightings. In addition, NJDEP formed a planning task force to manage and mitigate potential infestations.
Barriers to restoration	
Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
NJ Primary Sources	
Large business/industry	High – shipping and other maritime commerce
Small business industry	High
Transportation	High
Residential	Low
Agriculture	Low
Recreation	High
Resource extraction	Low
Government	Low
Natural sources/processes	High
Orphan contaminated sites	Low
Diffuse Sources	
Sediment sinks	N/A

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Soil sinks	N/A
Non-local air sources incl. deposition	N/A
Biota sinks	N/A

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Statewide Analysis of Threat

Threat = Zebra Mussel

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	4	1 (4) *	3	12 (48)
Marine Waters	N/A	N/A	N/A	N/A
Wetlands	N/A	N/A	N/A	N/A
Forests	N/A	N/A	N/A	N/A
Grasslands	N/A	N/A	N/A	N/A
Total Score				12 (48)
Average Score (total score ÷ 5)				2.4 (9.6)

*Number in parenthesis is if the zebra mussel is introduced to NJ

Watershed Management Regions

The map shows the state of New Jersey divided into five distinct regions. Upper Delaware is in the north, Passaic is in the northeast, Raritan is in the east-central part, Lower Delaware is in the southwest, and Atlantic Coastal is along the southern coast.

Risk by Watershed Management Region

THREAT =	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	High	NA	N/A	N/A	N/A
Passaic	High	N/A	N/A	N/A	N/A
Raritan	High	N/A	N/A	N/A	N/A
Atlantic	High	N/A	N/A	N/A	N/A
Lower Delaware	High	N/A	N/A	N/A	N/A
Region/Watershed (secondary)					
Urban	High	N/A	N/A	N/A	N/A
Suburban	High	N/A	N/A	N/A	N/A
Rural	High	N/A	N/A	N/A	N/A

H=high, M=medium, L=low, NA = not applicable

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www.nsgo.seagrant/research/nonindigenous/zmlifehistory.html

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<http://biology.usgs.gov/s+t/noframe/x274.htm>

New Jersey Comparative Risk Project
Ecological Technical Work Group
Stressor-Specific Risk Assessment

Risk Assessment Framework	Findings
Hazard Identification	
Stressor	Zinc
Description of stressor	<p>Zinc, atomic number 30, and atomic weight 65.38 is a malleable, ductile, gray metal. Because of chemical similarities among zinc, cadmium, and mercury, these three metals are classed together in a transition-elements subgroup of the periodic table. Zinc is known to have been present in ancient brass long before it was recognized as a separate element. The use of zinc in metal alloys and medicinal lotions dates back before the time of Christ. Currently, most of the commercial production of zinc involves the galvanizing of iron and the manufacture of brass. Zinc oxide is used in large quantities as an accelerant in the vulcanizing process in making tires; wearing of tires in use is a major contributor to environmental contamination by zinc, particularly by road-runoff to aquatic ecosystems (Callender & Rice, 2000). Zinc is a relatively common metal with an average concentration of 50 mg/kg soil and a range of 10-300 mg/kg soil (Barceloux, 1999). As an indication of the anthropogenic contribution to zinc contamination, in urban soils, Zn, along with Pb, Cr, and Cu are dominant metals and highly associated, with a correlation coefficient ranging from 0.83 to 0.98. Overall, significantly higher metal values occur in the inner city and lower values occur in outlying areas. Given common technical development, especially of traffic flows in cities, similar patterns of soil metals are expected for all US cities. The mobility of zinc in anaerobic environments is poor and therefore severe zinc contamination occurs primarily near points sources of zinc release. Primary prevention of urban metal accumulations is necessary to enhance and sustain the development of urban culture by lessening threats to public health and ecological integrity. (Mielke et al., 2000).</p> <p>Zinc is an essential co-factor in a variety of cellular processes including DNA synthesis, behavioral responses, reproduction, bone formation, growth, and wound healing. Meat, seafood, dairy products, nuts, legumes, and whole grains contain relatively high concentrations of zinc. Zinc compounds can produce irritation and corrosion of the gastrointestinal tract, along with acute renal tubular necrosis and interstitial nephritis. (Barceloux, 1999)</p> <p>Microorganisms: Brim, et al. (1999) studied the bacterial community of a zinc-contaminated soil. Colony-forming units were determined by plating on media with or without metals. Dominant isolates were characterized by PCR fingerprinting. The majority of the isolates were identified by FAME (fatty acid methyl ester analysis) as <i>Arthrobacter</i> spp. (18 out of 23). Most of the isolates were zinc tolerant but only seven could be considered zinc resistant. Metalloresistant <i>Rastomia eutropha</i> like strains previously shown to be dominant in the analyzed biotope were no longer observed. This may suggest a population replacement by less resistant bacteria, concomitant with a progressive decrease of the zinc toxicity in the soil. (Brim, et al., 1999).</p>

The degree of sliminess of a range of fungal isolates which form endomycorrhizas with *Calluna vulgaris* L. was investigated and related to their performance in the presence of potentially toxic concentrations of zinc. Some fungal isolates possess, to a greater or lesser extent, a mass of loosely adherent slime (LAS). The amount of this was proportional to the ability of the fungal mycelium to take up zinc from solution, the zinc tolerance of the fungal isolates, and the amount of growth enhancement exhibited by the associated *C. vulgaris* plants in the presence of potentially toxic concentrations of zinc. Furthermore the amount of LAS was inversely proportional to the concentration of zinc found in the shoots of the associated *C. vulgaris* plants. These observations support the hypothesis that LAS plays a central role in removing zinc ions from solution, removal which in turn determines the zinc tolerance of the fungus itself and the degree of amelioration of zinc toxicity to the host plant (Denny & Ridge, 1995).

Plants: Limited areas of cropland in Georgia have been amended with a high-Zn flue-dust material, which is causing Zn toxicities to crops. Shuman & Li, (1997). Determined the effects of lime and mushroom compost in reducing Zn uptake by cotton (*Gossypium hirsutum* L.). Experiments were carried out using a high-Zn toxic soil and a low-Zn non-toxic soil. There was a steady decline in plant Zn with increasing lime rates for the low-Zn soil. For the high-Zn soil, the lime did little to increase growth, but the compost and compost plus lime resulted in normal plants (soil pH 6.9). Liming and compost amendments redistributed Zn from the exchangeable fraction to the Mn oxide fraction, making Zn less plant available. It was concluded that for soils having mild Zn-toxicity, liming is the preferred remediation technique, but for soils having severe Zn-toxicity, organic amendments plus liming may be necessary

Petunia "Stereo Red" and Impatiens "Impulse Violet" were grown in soilless potting media that contained either 0, 5, 10, or 20% by volume of ground tire rubber. The pH of half of each medium was adjusted to 5.0 and that of the other half to 6.5. Shoot weights declined with increasing rubber content, due to increasingly severe zinc (Zn) toxicity. Extrapolation from the experimental data indicated that Petunia "Stereo Red" would have been damaged by as little as a 2% inclusion of tire rubber (Handreck, 1996).

Aquatic plants: Industrial pollution is a problem very often faced by aquatic systems. Among the industrial pollutants, heavy metals are a major problem. Zinc is a major effluent from the industries such as soft drink flavoring, fur dressing and dyeing, fish processing, laundry, etc.(Dinesan & George, 1993).

Stauber & Florence, (1989) studied the toxicity of ionic copper and zinc to the marine diatom *Nitzschia closterium*. To overcome the problems of metal complexation by chelators such as silicate, iron and EDTA in the assay medium, unsupplemented seawater and buffered synthetic softwater, enriched only with nitrate and phosphate, have been used for the study of metal toxicities to marine and freshwater algae.

Long-term growth and metal (Mn and Cd) uptake experiments and short-term uptake kinetic experiments with the coastal diatom *Thalassiosira pseudonana* revealed antagonistic interactions between toxic metals (Zn and Cd) and the micronutrient manganese. Cd and Zn inhibited algal growth rate only at low Mn ion concentrations, an effect that could be totally accounted for by an inhibition of cellular Mn uptake by the toxic metals and the resultant generation of Mn-deficient cells. Mn and Zn inhibited cellular Cd uptake, indicating reciprocal effects among the metals with respect to uptake. Saturation kinetics modeling of the uptake data was consistent with all three metals competing with each other for binding to the Mn uptake system. (Sunda & Huntsman, 1996).

Terrestrial Invertebrates: In aerially contaminated field sites adjacent to primary zinc smelters, zinc is invariably present in surface soils at concentrations of at least 50 times those of cadmium. EC50s for cadmium and zinc were determined for juvenile production of *Folsomia candida* Willem, 1902, at 20, and 15 degrees C in a standard laboratory test system. The EC50-reproduction values (micrograms g-1) for cadmium and zinc were similar at both 20 and 15 degrees C (20 degrees C: Cd, 590; Zn, 900; 15 degrees C: Cd, 540; Zn, 590). The similarity of the EC50-reproduction values for cadmium and zinc in *F. candida* at 20 and 15 degrees C determined in this study strongly suggests that deleterious effects of mixtures of these metals on populations of Collembola in such sites can be attributed to zinc rather than cadmium. (Sandifer & Hopkin, 1997)

Soil properties are a major influence on the bioavailability and toxicity of metals and represent one of the important factors that complicate the extrapolation of results from laboratory tests to field situations. The influence of soil characteristics and way of contamination on the bioaccumulation and toxicity of zinc and the influence of food supply on the toxicity of zinc were investigated for the springtail *Folsomia candida* in an attempt to identify factors that might modulate ecotoxicological effects of soil contaminants. It was concluded that both zinc stress and food shortage have a major influence on population development of *Folsomia candida*, whereas the interaction between food supply and zinc toxicity is small (Smit, et al., 1998). In freshly contaminated soils, zinc toxicity was related to organic matter and clay content of the soil; however, the use of these soils overestimated the effects of zinc for *F. candida* by a factor of 5 to 8 compared to a test soil that was subjected to aging under field conditions for 1.5 years. Equilibration of the zinc contamination by percolating the soils with water before use in the toxicity experiment strongly reduced the difference in zinc toxicity between laboratory-treated and aged soils (Smit, & Van Gestel, 1998).

Toxicity tests were conducted using a procedure based on the OECD (1984) artificial soil earthworm toxicity test. Toxic effects of zinc on *Eisenia fetida* as measured by reductions in survival and cocoon production were related more closely to soluble than total metal concentrations in soil. In a previous series of tests, zinc toxicity for *Eisenia fetida* was found to be at least ten times greater in OECD artificial soil when compared to contaminated soils collected from a polluted field site. It was concluded that this was due to the greater bioavailability of the metals in the OECD soil (Spurgeon & Hopkin, 1996).

Young garden snails (*Helix aspersa*) set as sentinels in cages laid on the soil for four weeks, give data for biomonitoring the environmental impact of chemicals on soil ecosystems in the field. The survival and the growth of the snails are influenced by the nature of the biotope and the level of the pollutants. Assay of zinc bioaccumulated in the tissues of the sentinel snails provides information on the bioavailability of this metal in the environment. The engagement model, little used for terrestrial species, can be useful in monitoring (specific and global endpoints) metal pollution of the environment in reference to the trophic level of the primary consumers. (Gomont de Vaulfleur & Pihan, 2000.)

Aquatic invertebrates: In chronic toxicity tests, *Ceriodaphnia dubia* were exposed to water spiked with Zn from two different wetlands. These tests indicated higher assimilative capacity in the wetland with values of dissolved organic carbon (DOC) and suspended solids that were significantly greater, suggesting that they may have influenced differences in bioavailability of Zn in wetland water. To test the influence of suspended solids on contaminant assimilation, *C. dubia* were exposed to filtered and unfiltered Zn-spiked wetland water. The greater toxicity of Zn to *C. dubia* in filtered water indicated that suspended solids were important in ameliorating impacts of Zn. Results from this study suggest that from the suite of variables assessed, DOC and suspended solids were the most important physicochemical influences on Zn toxicity within the water column of these two wetlands. (Polonsky & Clements, 1999)

Two microcosm-scale wetlands (570-liter containers) were integratively designed and constructed to investigate transfers and transformations of zinc associated with an aqueous matrix, and to provide future design parameters for pilot-scale constructed wetlands. The fundamental design of these wetland microcosms was based on biogeochemical principles regulating fate and transformations of zinc (pH, redox, etc.). Each wetland consisted of a 45-cm hydrosol depth inundated with 25 cm of water, and planted with *Scirpus californicus*. Zinc (approximately 2 mg/liter) as ZnCl₂ was amended to each wetland for 62 days. Individual wetland hydraulic retention times (HRT) were approximately 24 h. Total recoverable zinc was measured daily in microcosm inflow and out flows, and zinc concentrations in hydrosol and *S. californicus* tissue were measured pre- and post-treatment. *Ceriodaphnia dubia* and *Pimephales promelas* 7-day aqueous toxicity tests were performed on wetland inflows and outflows, and *Hyalella azteca* whole sediment toxicity tests (10-day) were performed pre- and post-treatment.

Approximately 75% of total recoverable zinc was transferred from the water column. Toxicity decreased from inflow to outflow based on 7-day *C. dubia* tests, and survival of *H. azteca* in hydrosol was >80%. Data illustrate the ability of integratively designed wetlands to transfer and sequester zinc from the water column while concomitantly decreasing associated toxicity. (Gillespie, et al.1999.)

The relationship between zinc and copper toxicity in *Hyalella azteca* and accumulation from metal-spiked sediments was determined for future use in identifying sites where these metals contribute to sediment toxicity. Both zinc and copper accumulation increased rapidly with increasing sediment concentrations, showing no evidence of saturation as seen in waterborne exposures. Zinc accumulation from zinc spiked sediments was substantial, even at concentrations below those causing chronic toxicity, and body concentrations can readily be used to infer chronic toxicity.

Chronic toxicity in spiked sediment exposures occurred at excess (above background) body concentrations twice as high as in waterborne exposures, possibly as a result of zinc in gut contents. Copper, but not zinc, caused a reduction in growth, even at concentrations below the 4-week LC25. Reduced growth can, therefore, be used to differentiate between chronic copper and zinc toxicity in sediments (Borgmann, & Norwood, 1997). Toxicity of sediments from Manitowadge Lake, Ontario, to *Hyalella* correlated closely with bioaccumulation of zinc but not copper. Bioaccumulation in 1-week exposures was sufficient to infer chronic zinc toxicity. Close similarity between toxicity-accumulation relationships from Manitowadge Lake and those obtained from zinc-spiked Hamilton Harbor sediments indicate that toxicity is due to zinc itself and not some other chemical that correlates with zinc in sediments. Sediment concentrations of zinc, on the other hand, are unreliable indicators of effects; toxicity was not highest in sediments from the most contaminated site. Body concentrations of zinc and copper in wild animals from contaminated lakes (in contrast to laboratory animals exposed to sediments from those lakes) are not reliable indicators of metal toxicity, either because these amphipods have adapted to contaminated conditions or because they survive in selected microhabitats with reduced metal availability (Borgmann & Norwood, 1997).

A study conducted to assess the extent of harm done by heavy metal zinc to *Penaeus indicus* through lethal toxicity tests and through histopathological observations on the hepatopancreas and gills of the animals following exposure to sub-lethal concentrations of the metal. The histopathological effects of starvation of the hepatopancreas were also worked out for a comparative study. Juvenile of *P. indicus* were exposed to various concentrations of zinc over a period of 96 hrs and the study reveals that juvenile *P. indicus* can tolerate zinc concentrations to an extent which is indicated by the high 96 hr LC₅₀ value of 1668.16 ppb. It is also evident from the results of histopathological studies that concentration as low as 100 ppb and 300 ppb are found to bring about destructive and deteriorating changes in the hepatopancreas and gills of the animal (Viswanathan & Manisseri, 1995).

Zinc (Zn) toxicity to a marine fouling dreissinid bivalve *Mytilopsis sallei* was investigated and the safe concentration of Zn was determined for this bivalve. In acute toxicity tests on *M. sallei* the 96 h LC₅₀ value for Zn is found to be 8.364 mg/L. The results suggest that Zn has the lowest toxic effect on *M. sallei* when compared to copper and mercury. Effect of Zn on the oxygen consumption of *M. sallei* was studied. Sub-lethal concentrations of Zn functioned as respiratory depressants. Accumulation of Zn was investigated in relation to exposure period and concentration in *M. sallei*. Zn accumulation was found to be sensitive to both concentration of Zn and exposure time. The effect of exposure time and concentration on body biochemical components was also studied in *M. sallei*. This accumulated Zn has resulted in impairment of carbohydrate metabolism and protein utilization. (Devi, 1995)

Amphibians: The nickel hazard was evaluated by means of a 7-day toxicity test with *Bufo arenarum* embryos. The LC₅₀ values for this metal from 24 to 168 h diminished from about 26 to 1.8 mg Ni²⁺/L, respectively, but from 96 h onward, the LC₅₀ varied very slightly. Although a noticeable difference among the LC₅₀ and LC₁₀ or LC₉₀ was observed at 24 h of exposure, these parameters tended to a similar value at 168 h of exposure while the confidence intervals of LC₅₀ overlapped all other confidence interval values. These results, plotted as toxicity profile curves, are useful for determining time and concentration thresholds for Ni. Nickel-zinc interactions on *B. arenarum* embryos were evaluated by means of simultaneous treatments with both cations (Ni: 5-35 mg Ni²⁺/L; Zn: 0.5-130 mg Zn²⁺/L). As a general pattern, low Zn concentrations (0.5 mg Zn²⁺/L) did not have a clear-cut effect on Ni toxicity, higher Zn concentrations (2-20 mg Zn²⁺/L) enhanced Ni toxicity, and concentrations of 30 mg Zn²⁺/L and higher had a beneficial effect in most cases. The metal interaction studies provide a scientific basis for the establishment of water quality criteria for wildlife protection purposes. (Herkovits, et al., 1999)

Fish: In dietary experiments with gilthead fish, zinc concentrations in the diet affected the weights and the tissue metal content. Fish fed a diet with an added 900 mg Zn/kg, showed the lowest weight gain and significantly higher values of Zn in gills, liver, and kidney, compared to fish fed a control diet with no added Zn. Muscle and brain had the lower mean values and showed a tight control of zinc levels. These results reinforce the hypothesis that zinc is strictly controlled in the CNS in order to maintain the functional role of the metal. (Serra et al., 1996)

Many studies conducted involving heavy metal pollution are limited to the estimation of pollutants in water, their accumulation in aquatic organisms and lethal effects. But it is important to understand the effects of pollutants on various vital systems of organisms at the cellular level. Dinesan & George, (1993) investigated the damages caused by zinc at the cellular level on various organs of milkfish.

The acute Zinc toxicity of adult freshwater teleosts *Notopterus notopterus* (Pallas) and *Puntius javanicus* (Blkr) for 96-hour were performed using static-bioassay test. In separate exposures of Zinc to *N. notopterus* and *P. javanicus*, the LC₅₀ (median lethal) values were estimated to be 59.79 mg/L and 29.62 mg/L respectively. The ET₅₀ values were 53 hours at 70 mg/L for *N. notopterus* and 53 hours at 40 mg/L for *P. javanicus*. The harmless concentration for *N. notopterus* and *P. javanicus* were 0.3109 mg/L and 0.1356 mg/L respectively. The comparative account of behavioral responses against Zinc toxicity exhibited that *P. javanicus* was more sensitive than *N. notopterus*. (Gupta & Chakrabarti, 1993)

Selected specimens of murrel (*Channa punctatus*) were exposed to a sublethal zinc concentration (2.4 mg L super(-1)), determined as a fraction of the 96-h LC₅₀ value, for 15, 30, 60, and 120 d. The zinc exposure produced marked changes in the chemical composition of liver and muscle tissues. These changes were more severe after 60 and 120 d, suggesting a cumulative action of zinc on the tissues. The metabolism of the fish decreased with the time of exposure and there was a decline in the calorific value of lipid, protein, and glycogen in muscle and liver. Glycogen and iron were the components most affected and the nutritive value of the fish was also adversely affected (Malik, et al., 1998).

Issue: Zinc
 Author: R. Tucker
 Version: 07/27/00

Stressor-specific impacts considered: Biological integrity Biodiversity Habitat/ecosystem health Ecosystem function	Zn may negatively impact individual organisms in populations, especially those in contact with contaminated benthic sediments. Soil organisms may also be affected at contaminated sites
Key impacts selected (critical ecological effects)	Because of the widespread contamination by zinc, impacts on fresh water and estuarine benthic organisms are probably key.
Exposure Assessment	
Exposure routes and pathways considered	Industrial activity contaminating specific sites, Atmospheric deposition to soil or to aquatic ecosystems, runoff from soil to aquatic ecosystems; uptake to plants and animals from soil, sediment, or water containing excess quantities of Zn.
Population(s)/ecosystem(s) exposed statewide Quantification of exposure levels statewide	<p>Freshwater and estuarine ecosystems, and associated populations of organisms</p> <p>Average background Zn soil concentrations in NJ: 38.5 mg/kg (suburban land use; Fields et al., 1993), 34.0 mg/kg (rural land use), 64.8 mg/kg (farm land use; Fields et al., 1993), 93.0 mg/kg (urban Piedmont regions; Wong & Sanders, 1998), and 162.3 mg/kg (urban areas; Fields et al., 1993).</p> <p>Newark Bay (217 samples from NOAA's Coastal Protection and Restoration Division Data for Newark Bay Watershed Database) Mean – 532 mg/kg (ppm) Range - 20.5 1900 mg/kg</p> <p>Tidal Delaware Zn concentrations above the ER-M occurred throughout much of the area from Trenton (River mile 133) to Artificial Island (River mile 50), with concentrations > 500 ppm at RM 125 (near Trenton) and between RM 88.5 and 101.0 (particularly Philadelphia-Camden area and associated industrial/petroleum complex). Zn exceeded ER-L in 17 of 24 samples and exceeded the ER-M in 10 of the 24 sediment samples taken in a study of the Delaware River.</p>
	Barnegat Bay samples from cores, which can be dated by radionuclide tracers: Recent bay sediment samples mean 127.7 ppm (n=17) Range: 62 - 239 ppm

	<p>From pre-1978 portion of cores mean 106.6 ppm (n=12) Range: 41 - 200 ppm Samples from marinas mean 369.5 ppm (n=4) Range: 328 – 395 ppm (Moser & Bopp, 1999)</p> <p>Freshwater sediment samples from throughout New Jersey USGS, statewide sampling, 1998 mean 119 mg/kg (ppm); range 1 – 520 ppm (36 Zn measurements/ 124 total stations) Concentrations in water ranged from <10 to 50 µg/L (12 samples from 10 to 50; 33 samples at < 10)</p> <p>Marine Coastal Waters: Surface water Zn concentrations ranged from 1.7-9.3 µg/L in 1988, and 1.0–2.0 µg/L in 1991 in the New York Bight Apex (Battelle, 1991).</p> <p>Zn concentrations in the New York Bight Apex sediments ranged from 660-1,020 mg/kg (n≈200; Krom et al., 1985), and from 7-230 mg/kg (n=27; NOAA, 1982). At the Mud Dump Site (HARS) sediment concentrations ranged from 27 mg/kg to 329 mg/kg (n=23; Battelle, 1996). Background Zn concentrations in New York Bight sands were 15-30 mg/kg (n=45; Battelle, 1996).</p>
<p>Specific population(s) at increased risk</p> <p>Quantification of exposure levels to population(s) at increased risk</p>	<p>Benthic organisms, both freshwater and estuarine; especially in Newark Bay Complex</p> <p>(see above for benthic sediment values) Benchmarks: Surface Water: Zinc Freshwater (chronic) =100 µg/l (@ hardness of 100 mg/L CaCO₃) Marine/Estuarine (chronic) = 81 µg/L (NJDEP, 2000)</p> <p>Sediment: Zinc Freshwater=120 mg/kg (LEL) and 820 mg/kg (SEL) (Persaud et al., 1993)</p> <p>Marine/Estuarine=150 mg/kg (ER-L; lower 10th percentile of apparent effects) and 410 mg/kg (ER-M; median effects level) (Long et al., 1995)</p>
	<p>Soil = 50 mg/kg (based on phytotoxicity; Efrogmson et al., 1997)</p> <p>Proposed soil cleanup guideline = 1500 ppm (NJDEP Site Remediation Program)</p>

	Zinc is a relatively common metal with an average concentration of 50 mg/kg soil and a range of 10-300 mg/kg soil (Barceloux, 1999).
Dose/Impact-Response Assessment	
Quantitative impact-assessment employed	Comparison with freshwater and estuarine screening values
Risk Characterization	
Risk estimate(s) by population at risk Risk Score = (Severity/Irreversibility) x (Frequency) x (Magnitude)	Score Mean sediment values in Newark Bay samples substantially exceeded the screening guidelines. Although mean concentrations in benthic sediments in other parts of the State were below guidelines, the ranges exhibited samples above the guideline value.
Assessment of severity/irreversibility 5 - Lifeless ecosystems or fundamental change; Irreversible 4 - Serious damage: • many species threatened/endangered • major community change • extensive loss of habitats/species Long time for recovery 3 - Adverse affect on structure and function of system: • all habitats intact and functioning • population abundance and distributions reduced Short time for recovery 2 - Ecosystem exposed but structure and function hardly affected 1 - No detectable exposure	2 - 3 (3 may apply to Newark Bay, but because of the multitude of other contaminants, it would be difficult to attribute specific effects just to Zn).
Assessment of frequency of effect(s) (list definition for each category, e.g., rare = 1/decade) 5 - Often and increasing 4 - Often and continuing 3 - Occasional 2 - Rare 1 - Possible in the future 0 - Unlikely (or 0.1)	3 - 4

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Size of population(s) and/or extent of the State/habitat affected (magnitude) 5- >50% of the State/population impacted 4- 25-50% of the State/population impacted 3- 10-25% of the State/population impacted 2- 5-10% of the State/population impacted 1- <5% of the State/population impacted	(2) - Based on available data, some limited effect in many benthic habitats Statewide, substantial effects in Newark Bay
	Total: 12-24 (18)
Assessment of uncertainties in this assessment (H,M,L) and brief description	M Limited data makes assessment uncertain, especially for terrestrial locations.
Potential for additional data to result in a significant future change in this risk estimate (H, M, L) and brief description. (Data Gaps; highlight significant data needs)	M - more data needed especially for terrestrial sites
Potential for future changes in the underlying risk from this stressor (+++, ++, +, 0, !, =, ≡; where + is improvement), and brief description.	0 - underlying risk fairly well characterized, change unlikely
Potential for catastrophic impacts (H,M,L) and brief description	L - large scale discharges unlikely
Link to other Work Groups (e.g., socioeconomic impacts)	
Extent to which threat is currently regulated or otherwise managed	Guidelines and criteria exist for water. Non-point sources, especially traffic (tire) related, unregulated
Barriers to restoration	

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Relative Contributions of Sources to Risk (H,M,L); include any information/details on sources	
Non-local air sources incl. deposition	M
Biota sinks	L

Summary Statement:

Zinc is known to have been present in ancient brass long before it was recognized as a separate element. The use of zinc in metal alloys and medicinal lotions dates back several millennia. Currently, most of the commercial production of zinc involves the galvanizing of iron and the manufacture of brass. Zinc oxide is used in large quantities as an accelerant in the vulcanizing process in making tires; wearing of tires in use is a major contributor to environmental contamination by zinc, particularly by road-runoff to aquatic ecosystems. Zinc is a relatively common metal with an average concentration of 50 mg/kg soil and a range of 10-300 mg/kg soil, but much more likely to occur in urban soils because of traffic flows and other human inputs in cities.

Zinc is an essential co-factor in a variety of cellular processes in plants and animals, and thus commonly included in fertilizer applications and in animal feeds. But toxic concentrations of zinc can limit plant growth and inhibit reproduction in animal populations. Soil properties are a major influence on the bioavailability and toxicity of metals and represent one of the important factors that complicate the extrapolation of results from laboratory tests to field situations.

Bioaccumulation may result in chronic zinc toxicity. Experiments with aquatic animals indicated close similarity between toxicity-accumulation relationships from sediments and found that toxicity is due to zinc itself and not some other chemical that correlates with zinc in sediments. Sediment concentrations of zinc, on the other hand, are unreliable indicators of effects; toxicity was not highest in sediments from the most contaminated site. In the bivalve *Mytilopsis sallei*, sub-lethal concentrations of zinc functioned as respiratory depressants.

In embryo-larval stages of *Mytilus galloprovincialis*, zinc was found to be markedly more toxic than cadmium, particularly within the period of early cleavage and blastula stages. In fish, zinc concentrations in the diet affect growth. Zn may negatively impact individual organisms in populations, especially those in contact with contaminated benthic sediments. Soil organisms may also be affected at contaminated sites

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Statewide Analysis of Threat Threat = Zinc

Ecosystem	Severity Irreversibility	Frequency	Magnitude	Score
Inland Waters	2-3	3	2	14
Marine Waters	3	4	3	36
Wetlands	2	4	3	24
Forests	1	2	1	2
Grasslands	1	2	1	2
Total Score				78
Average Score (Total ÷ 5)				15.6

Risk by Watershed Management Region

THREAT = Zinc	ECOSYSTEM				
Watershed Management Region	Inland Waters	Marine Waters	Wetlands	Forests	Grasslands
Upper Delaware	L	NA	L	L	L
Passaic	L-M	M	L-M	L	L
Raritan	L-M	M	L-M	L	L
Atlantic	L	L-M	L	L	L
Lower Delaware	L-M	M	L-M	L	L
Region/Watershed (secondary)					
Urban	L-M	M	L-M	L	L
Suburban	L-M	M	L-M	L	L
Rural	L	L-M	L	L	L

H=high, M=medium, L=low, NA = not applicable



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