Potential for Atrazine Use in the Chesapeake Bay Watershed to Affect Six Federally Listed Endangered Species: Shortnose Sturgeon (*Acipenser brevirostrum*); Dwarf Wedgemussel (*Alasmidonta heterodon*); Loggerhead Turtle (*Caretta caretta*); Kemp's Ridley Turtle (*Lepidochelys kempii*); Leatherback Turtle (*Dermochelys coriacea*); and Green Turtle (*Chelonia mydas*)

Pesticide Effects Determination

Environmental Fate and Effects Division
Office of Pesticide Programs
Washington, D.C. 20460

August 31, 2006
(March 14, 2007 – amended during informal consultation with U.S. Fish and Wildlife Service and National Marine Fisheries Service)

Table of Contents

1.0 Executive Summary ... 10
2.0 Problem Formulation ... 12
2.1 Purpose ... 12
2.2. Scope of Assessment ... 14
2.2.1. Assessed Uses ... 14
2.2.2. Chemicals Assessed ... 14
2.2.3. Species Assessed ... 16
2.3 Previous Assessments ... 23
2.4. Characterization of Waters Inhabited by the Six Assessed Species ... 24
2.5. Action Area ... 32
2.6. Stressor Source and Distribution ... 35
2.6.1. Environmental Fate and Transport Summary ... 35
2.6.2. Use Characterization ... 36
2.7. Mechanism of Herbicidal Action ... 40
2.8. Assessment Endpoints and Measures of Ecological Effect ... 40
2.9. Conceptual Model ... 43
2.9.1. Risk Hypotheses ... 43
2.9.2. Diagram ... 43
3.0 Exposure Assessment ... 45
3.1. Conceptual Model of Exposure ... 46
3.2. Use of Modeling to Characterize Potential Exposures to Atrazine in the Chesapeake Bay Watershed ... 47
3.2.1. Modeling Approach ... 47
3.2.2. Model Inputs ... 55
3.2.3. Model Results ... 61
3.3. Additional Modeling Exercises Used to Characterize Potential Exposures ... 63
3.3.1. Residential Uses ... 63
3.3.2. Additional Characterization of Agricultural Use EECs ... 67
3.3.3. Specific Characterization for Headwater Streams at Locations of the Dwarf Wedge- mussel ... 72
3.3.4. Summary of Alternative Modeling Exercises ... 79
3.4. Monitoring Data ... 79
3.4.1. National USGS NAWQA Data ... 80
3.4.2. USGS Watershed Regression of Pesticides (WARP) Data ... 84
3.4.3. Regional USGS NAWQA Data ... 84
3.4.4. Chesapeake Bay Program Data ... 86
3.4.5. Heidelberg College Data ... 92
3.4.6. U.S. EPA ORD Great Lakes Program – Lake Michigan Mass Balance Project ... 94
3.4.7. Summary of Open Literature Sources of Monitoring Data for Atrazine ... 95
3.5 Summary of Modeling vs. Monitoring Data ... 96
3.6. Oral Exposure to Sea Turtles ... 97
3.6.1. Dietary Exposure from Contaminated Food Items ... 97

3.6.2. Potential Exposure to Sea Turtles from Water Intake 97
4.0 Effects Assessment 98
4.1. Toxicity Data Used to Evaluate Assessment Endpoints 99
4.2. Toxicity Classification Scheme 101
4.3. Laboratory Effects Data 102
4.3.1. Toxicity to Fish 102
4.3.2. Toxicity to Aquatic Invertebrates 105
4.3.3. Toxicity to Sea Turtles 109
4.4. Terrestrial Plant Toxicity 112
4.5. Aquatic Plant Toxicity Data 114
4.5.1. Laboratory Data 114
4.5.2. Field Data 115
4.6. Community-Level Endpoints: Threshold Concentrations 115
4.7. Use of Probit Slope Response Relationship to Provide Information on the Endangered Species Levels of Concern 118
4.8. Incident Database Review 119
5.0 Risk Characterization 121
5.1. Direct Effects Assessment 121
5.1.1. Risk Estimation 121
5.1.2. Risk Description, Direct Effects 123
5.1.3. Summary of Direct Effects Conclusions 127
5.2. Indirect Effects 127
5.2.1. Summary of Biological and Ecological Information Used to Evaluate Potential Indirect Effects of Atrazine 129
5.2.2. Evaluation of the Potential for Atrazine to Induce Indirect Effects on the Shortnose Sturgeon, Dwarf Wedgemussel, and Sea Turtles from Reduction in Animal Food Items 132
5.2.3. Summary of Effects Determinations: Indirect Effects from Direct Effects to Aquatic Animals 140
5.2.4. Evaluation of Potential Indirect Effects to the Six Listed Species from Potential Effects to Aquatic Plants 140
5.2.5. Potential Indirect Effects to the Listed Species via Direct Effects to Terrestrial Plants 144
5.2.6. Potential for Atrazine to Affect the Assessed Species via Effects on Riparian Vegetation in the Chesapeake Bay Watershed 152
6.0 Uncertainties 163
7.0 Conclusions 172
8.0 References 174

List of Appendices

Appendix A	Ecological Effects Data
Appendix B	Supporting Information for the Aquatic Community Level of Concern (LOC) and Method to Apply the LOC to the Exposure Data (Use of Threshold Concentrations and CASM Model)
Appendix C-1	Summary of Exposure Modeling Approach and Supporting Information for Scenario Development
Appendix C-2	Residential Exposure Assumptions
Appendix D	Profiles of the Six Assessed Species
Appendix E	Incident Database Information
Appendix F	RQ Method and Description of LOCs
Appendix G	Map of Submerged Aquatic Vegetation in the Chesapeake Bay
Appendix H	Jellyfish Count and Biovolume Data in the Chesapeake Bay
Appendix I	Land use data Surrounding Dwarf Wedgemussel Populations and Analyses of Riparian Areas of Dwarf Wedgemussel Populations on Maryland's Eastern Shore
Appendix J	Bibliography of ECOTOX Open Literature Not Evaluated
Appendix J-2	Bibliography of Rejected ECOTOX Open Literature

LIST OF TABLES

Table 1.1. Summary of Effects Determinations For Six Listed Species 11
Table 1.2. Identification and Listing Status of Six Listed Species Included in this Assessment 12
Table 2.1. Summary of Formation Pathway of Atrazine Degradates 15
Table 2.2. Summary of Available Degradate Toxicity Data in Birds and Mammals 16
Table 2.3. Location of the Dwarf Wedgemussel in the Chesapeake Bay Watershed 18
Table 2.4. Summary of Location of The Six Assessed Listed Species Within the Chesapeake Bay 25
Table 2.5. Summary of Assessment Endpoints and Measures of Ecological Effect 42
Table 3.1. Types of Waters Inhabited by the Assessed Species and Data Used in Exposure Refinement 47
Table 3.2 Label Application Information for the Chesapeake Bay Endangered Species Assessment 57
Table 3.3. Comparison of Modeled Application Rates and Number of Applications with Typical Use Data Used in the Triazine Cumulative Risk Assessment 60
Table 3.4. Summary of PRZM/EZAMS Environmental Fate Data Used For Aquatic Exposure Inputs For Atrazine Endangered Species Assessment for the Chesapeake Bay 61
Table 3.5. Summary of PRZM/EXAMS Output for all Scenarios Modeled for the Atrazine Endangered Species Assessment for the Chesapeake Bay Watershed using the Standard Water Body 62
Table 3.6. Summary of the Impact of Variations in Pervious to Impervious Ratio on Predicted Exposures from the PRZM/EXAMS Residential Scenario (granular) with 1% Overspray 63
Table 3.7. Comparison of Residential Scenario with an Assumption of No Overspray on Impervious Surface to the Alternate Assumptions of 10% and 1% Overstay to Impervious Surfaces 64
Table 3.8. Comparison of Residential Scenario with an Assumption of 50% of the ¼ acre lot treated to the Residential Scenario (granular) with an Assumption of 75% and 10% of the ¼ acre lot treated 65
Table 3.9. Percentage of Runoff Resulting from Impervious Surfaces 66
Table 3.10. Summary of the Impact of Runoff Differential on the Predicted Exposures from the PRZM/EXAMS Residential Scenario (granular) Using the Analysis from Table 3.9 67
Table 3.11. Summary of PRZM/EXAMS Output for all Scenarios Modeled for the Atrazine Endangered Species Assessment for the Chesapeake Bay Watershed Using a Standard Water Bod 68
Table 3.12. Comparison of Alternative PRZM Modeling with EEC Generated Using a Static Water Body 71
Table 3.13. Estimated Flow Rate for Water Bodies Known to be Inhabited by the Dwarf Wedgemussel. 72
Table 3.14. Aquia Creek Flow Information (ft^3/sec) 74

Table 3.15. South Anna River Flow Information (ft^3/sec) ... 75
Table 3.16. Summary of Alternative PRZM Modeling Using the Index Reservoir and Site Specific Flow data from Aquia Creek and South Anna River Compared with PRZM EEC Modeling Using a Static Water Body ... 77
Table 3.17. Annualized Time Weighted Mean (TWM) Concentration (µg/L) for the Top Ten NAWQA Surface Water Sites Ranked by Maximum Concentration Detected ... 82
Table 3.18. Maximum Concentration (µg/L) for the Top Ten NAWQA Surface Water Sites Ranked by Maximum Concentration Detected ... 83
Table 3.19. Annual Time Weighted Mean and Maximum Concentration from the Top Three USGS NAWQA Surface Water Sites Located in the Potomac Study Unit ... 85
Table 3.20. Annual Time Weighted Mean and Maximum Concentration from the Top Three USGS NAWQA Surface Water Sites Located in the Lower Susquehanna River Study Unit ... 86
Table 3.21. Annual Time Weighted Mean and Annual Maximum Concentrations from Selected Sample Locations from the Surface Water Monitoring Data from the Chesapeake Bay Program ... 88
Table 3.22. Annual Time Weighted Mean and Maximum Concentrations (µg/L) for Atrazine in Two Ohio Watersheds from the Heidelberg College Data ... 92
Table 3.23. Magnitude and Duration Estimates from the 1997 Data from Sandusky Watershed 94
Table 3.24. Dietary Exposure Estimations for Sea Turtle ... 98
Table 4.1. Summary of Toxicity Data Used to Evaluate the Assessment Endpoints for the Six Assessed Listed Species. ... 101
Table 4.2. Summary of Fish Toxicity Studies Used In Risk Quotient Calculations. ... 103
Table 4.3. Acute and Chronic Aquatic Invertebrate Toxicity Values Used ... 108
Table 4.4. Summary of Available Acute Oral, Subacute Dietary, Reproduction Toxicity Studies in Birds, and Available Studies in Reptiles. ... 111
Table 4.5. Nontarget Terrestrial Plant Seedling Emergence Toxicity (Tier II). ... 113
Table 4.6. Nontarget Terrestrial Plant Vegetative Vigor Toxicity (Tier II). ... 113
Table 5.1. Summary of Aquatic RQs to Assess Potential Direct Effects to the Shortnose Sturgeon. ... 122
Table 5.2. Summary of RQs Used to Estimate Direct Effects to the Dwarf Wedgemussel. ... 122
Table 5.3. Summary of RQs Used to Assess Potential Direct Effects to Sea Turtles. ... 123
Table 5.4. Summary of Direct Effects Determinations to the Six Assessed Listed Species. ... 127
Table 5.5. Summary of Shortnose Sturgeon Dietary Items. ... 130
Table 5.6 Summary of Dietary Items of the Assessed Sea Turtles. ... 132
Table 5.7. Summary of Animal Prey Items of the Six Assessed Species. ... 133
Table 5.8. Summary of RQs Used to Estimated Indirect Effect to Shortnose Sturgeon and Sea Turtles via Potential Direct Effects on Dietary Items. ... 133
Table 5.9. Summary of RQs Used to Assess Potential Risk to Animal Food Items of the Shortnose Sturgeon, Dwarf Wedgemussel, and Four Sea Turtles. ... 135

Table 5.10. Summary of Indirect Effects Determinations to the Six Assessed Listed Species Resulting from Direct Effects to Aquatic Animals. 140
Table 5.11. Summary of RQs Used to Estimated Indirect Effect to Shortnose Sturgeon, Dwarf Wedgemussel, and Sea Turtles via Direct Effects on Primary Productivity. 141
Table 5.12. Summary of PRZM/EXAMS Output for Corn, Sorghum, and Fallow/Idle Land and Comparison of Estimated Atrazine Concentrations to Community Level Effects Thresholds. . 142
Table 5.13. Summary of Results of EPA Monitoring Data. 143
Table 5.14. Summary of Indirect Effects Determinations to the Six Assessed Listed Species Resulting from Direct Effects to Aquatic Plants. 144
Table 5.15. Semi-quantitative Criteria Related to Riparian Vegetation for Assessing the Health of Riparian Areas for Supporting Aquatic Habitats. 145
Table 5.16. Screening-Level Exposure Estimates for Terrestrial Plants to Atrazine. 147
Table 5.17. Nontarget Terrestrial Plant Seedling Emergence Toxicity. 148
Table 5.18. Nontarget Terrestrial Plant Vegetative Vigor Toxicity. 149
Table 5.19. Conclusions for the Potential of Atrazine to Affect Specific Dwarf Wedgemussel Populations. 159
Table 7.1. Summary of Effects Determinations For Six Listed Species 173

LIST OF FIGURES

Figure 2-1. Shortnose Sturgeon Captured Between 1994 and 2003. 20
Figure 2-2. Conceptual Model of an Exposure Scenario Representing Headwater Streams in the Chesapeake Bay Watershed. 26
Figure 2-3 Conceptual Model of an Exposure Scenario Representing Mid-Level River Reach in the Chesapeake Bay Watershed. 27
Figure 2-4. Conceptual Model of an Exposure Scenario Representing Major River Reach in the Chesapeake Bay Watershed. 28
Figure 2-5 Conceptual Model of an Exposure Scenario Representing a Mouth of Major River in the Chesapeake Bay Watershed. 29
Figure 2-6. Conceptual Model of an Exposure Scenario Representing a Minor Estuarine Inlet in the Chesapeake Bay Watershed. 30
Figure 2-7. Conceptual Model of an Exposure Scenario Representing the Main Stem of the Chesapeake Bay. 31
Figure 2-8. Action Area Defined by Chesapeake Bay Watershed for the Atrazine Endangered Species Assessment 34
Figure 2-9 National Distribution of Atrazine Use. 37
Figure 2-10. Atrazine Use (from Kaul et al, 2005) in the Immediate Vicinity of the Chesapeake Bay 39
Figure 2-11. Conceptual Model 44
Figure 3-1. Percentage of Impervious Surfaces in the Chesapeake Bay and its Immediate Tributaries 52
Figure 3-2 Density of Road, Railways, and Pipelines as Surrogate for Right of Way Density in Chesapeake Bay Watershed (Action Area) 54
Figure 3-3. Location of NAWQA Surface Water Sites on Aquia Creek Relative to Chesapeake Bay Program (CBP) Surface Water Sites. 78
Figure 3-4. Range of Atrazine Concentrations Detected in the Chesapeake Bay and its Immediate Tributaries. 89
Figure 3-5. Location of Surface Water Monitoring Sites in the Chesapeake Bay and its Immediate Tributaries. 90
Figure 3-6. Location of Maximum Atrazine Detection (ng/l) in the Chesapeake Bay Watershed in data from the Chesapeake Bay Program. 91
Figure 4-1. Range of Aquatic Invertebrate Acute Toxicity Values Reported for Atrazine. 106
Figure 4-2. Use of Threshold Concentrations in Endangered Species Assessment. .... **Error! Bookmark not defined.**
Figure 5-1. Range of Aquatic Invertebrate Acute Toxicity Values Reported for Atrazine. 137
Figure 5-2. Map of Land Cover Data in the Po River Watershed 155
Figure 5-3. Example of a Riparian Area of Norwich Creek. 157

Figure 5-4. Example of Riparian Area Upstream of the Norwich Creek Site. Presented in Figure 5-3. 158
Figure 5-5. Summary of the Potential of Atrazine to Affect the Six Assessed Listed Species via Riparian Habitat Effects.. 163
Figure 6-1. Jellyfish Monitoring Data in the Chesapeake Bay.. 168

1.0 Executive Summary

The purpose of this assessment is to make an "effects determination" for six listed species in the Chesapeake Bay watershed: shortnose sturgeon, dwarf wedgemussel, loggerhead sea turtle, leatherback sea turtle, Kemp's ridley sea turtle, and green sea turtle. The following assessment endpoints were evaluated: (1) direct toxic effects on the survival, reproduction, and growth of the assessed species; (2) indirect effects resulting from reduction of food supply; and (3) indirect effects resulting from habitat modification. This assessment was completed in accordance with the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998), the August 5, 2004 Joint Counterpart Endangered Species Act Section 7 Consultation Regulations specified in 50 CFR Part 402 (USFWS/NMFS, 2004a; FR 69 47732-47762), and procedures outlined in the Agency's Overview Document (U.S. EPA, 2004).

Environmental fate and transport models were used to estimate high-end exposure values as a result of agricultural and non-agricultural atrazine use in accordance with label directions. Modeling was initially performed using the standard static water body. However, the environments in which the assessed species are located include primarily flowing water bodies such as streams and rivers and the main stem of the Chesapeake Bay. Except for short-term exposures in small, flowing streams and small estuarine inlets, estimated exposures from the available standard models are not likely to be representative of the types of waters inhabited by the assessed species. Therefore, additional modeling was used together with available monitoring data for the purpose of characterizing atrazine exposures in flowing waters. This analysis shows that peak atrazine concentrations are expected to be approximately 50 µg/L or higher, but longer-term (days to weeks) exposures are expected to be in the low µg/L range.

The assessment endpoints include direct toxic effects on the survival, reproduction, and growth of the assessed species, as well as indirect effects, such as reduction of the prey base and/or modification of its habitat. Direct effects are based on toxicity information for surrogate species (U.S. EPA, 2004). Given that food items and habitat requirements of the assessed species are dependant on the availability of aquatic invertebrates, aquatic plants, and terrestrial plants (i.e., riparian habitat), toxicity information for these taxonomic groups is also discussed. In addition, indirect effects, via impacts to aquatic plant community structure and function, are also evaluated based on time-weighted threshold concentrations that correspond to potential aquatic plant community-level effects.

Risk quotients (RQs) are derived as quantitative estimates of potential high-end risk. Acute and chronic RQs are compared to the Agency's levels of concern (LOCs) to identify instances where atrazine use within the action area has the potential to adversely affect the six assessed species via direct toxicity or indirectly based on direct effects to their food supply (i.e., freshwater invertebrates) or habitat (i.e., aquatic plants and terrestrial riparian vegetation). When RQs for a particular type of effect are below LOCs, the potential for adverse effects to the assessed species is expected to be negligible, leading to a conclusion of "no effect". Where RQs exceed LOCs, a potential to cause adverse effects is identified, leading to a conclusion of "may affect". If a determination is made that use of atrazine within the action area "may affect" the assessed species, additional information is considered to refine the potential for exposure and effects, and the best available information is used to distinguish those actions that “may affect, but are not likely to adversely affect” from those actions that are “likely to adversely affect” the assessed species.

A summary of the risk conclusions and effects determinations for the six listed species is presented in Table 1-1. Further information on the results of the effects determination is included as part of the Risk Description in Section 5.

Table 1.1. Summary of Effects Determinations For Six Listed Species

Assessment Endpoint	Species	Effects Determination	Basis for Determination
Direct effects to listed species (Section 5.1)	All six assessed species	No Effect	No acute or chronic LOCs for endangered species are exceeded.
Indirect effects to listed species via reduction of aquatic animals as food supply (Section 5.2.2.)	Shortnose sturgeon, loggerhead turtle, Kemp’s ridley turtle, green turtle, leatherback turtle	Not likely to adversely affect	Acute LOCs are exceeded for some animals that are food items of the assessed species. However, the low magnitude of potential effects on any one species, the low number of dietary species potentially affected (indicated by LOC exceedances) relative to the number potentially consumed by the assessed species, and the conservative nature of the EECs used to derive RQs for organisms in flowing water systems suggests that the potential effects to the food supply of the assessed species constitutes an insignificant effect.[a]
	Dwarf wedgemussel	No effect	No acute or chronic LOCs are exceeded.
Indirect effects to listed species via reduction of aquatic plants as food items or primary productivity (Section 5.2.4.)	All six assessed species	Not likely to adversely affect	No known obligate relationship between the assessed species and any single aquatic plant species exists, and short-term and long-term atrazine concentrations were estimated to be lower than established thresholds for community-level effects to aquatic vegetation.
Indirect effects to listed species via direct effects on riparian areas required to maintain acceptable water quality and spawning habitat (Section 5.2.5.)	Shortnose sturgeon and each of the four assessed sea turtles	Not likely to adversely affect	Acreage of riparian habitat expected to be sensitive to atrazine is sufficiently low in the Chesapeake Bay watershed such that potential impacts of atrazine to sensitive riparian buffers are not expected to result in a measurable effect to the assessed species that reside in the main stem of the Chesapeake Bay and the Major river systems. Therefore, potential effects to riparian areas from use of atrazine are expected to constitute an insignificant effect [a].
	Dwarf wedgemussel	Not likely to adversely affect	Land cover data surrounding watersheds of dwarf wedgemussel habitats suggest that riparian area exposure to atrazine is expected to be minimal and/or that the predominant riparian area adjacent to waters of dwarf wedgemussel habitats is not expected to be sensitive to atrazine. Therefore, potential effects to the dwarf wedgemussel from effects to riparian areas are expected to constitute an insignificant effect.[a]

[a] Significance of Effect: Insignificant effects are those that cannot be meaningfully measured, detected, or evaluated in the context of a level of effect where take occurs for even a single individual

2.0 Problem Formulation

Problem formulation provides a strategic framework for the risk assessment. By identifying the important components of the assessed species and ecological stressor, it focuses the assessment on the most relevant life history stages, habitat components, chemical properties, exposure routes, and endpoints. This assessment was completed in accordance with the August 5, 2004 Joint Counterpart Endangered Species Act (ESA) Section 7 Consultation Regulations specified in 50 CFR Part 402 (U.S. FWS/NMFS, 2004a; FR 69 47732-47762). The structure of this risk assessment is based on guidance contained in U.S. EPA's *Guidance for Ecological Risk Assessment* (U.S. EPA, 1998), the Services' *Endangered Species Consultation Handbook* (USFWS/NMFS, 1998) and procedures outlined in the Overview Document (U.S. EPA, 2004).

2.1 Purpose

The purpose of this ecological risk assessment is to evaluate the potential direct and indirect effects resulting from use of the herbicide atrazine (6-chloro-N-ethyl-N-isopropyl-1, 3, 5-triazine-2, 4-diamine) in the Chesapeake Bay watershed on the survival, growth, and/or reproduction of the following six federally listed species: (1) shortnose sturgeon (*Acipenser brevirostrum)*; (2) dwarf wedgemussel (*Alasmidonta heterodon)*; (3) loggerhead turtle (*Caretta caretta)*; (4) Kemp's ridley turtle (*Lepidochelys kempii)*; (5) green turtle (*Chelonia mydas)*; and (6) leatherback turtle (*Dermochelys coriacea)*. A summary of the listing status for these species is provided in Table 1.2, and a brief summary of key biological and ecological components related to an assessment of these species is provided in Section 2.2.3. This ecological risk assessment is a component of the settlement for the *Natural Resources Defense Council, Civ. No: 03-CV-02444 RDB (filed March 28, 2006)*. No critical habitat has been designated within the Chesapeake Bay watershed for the assessed species.

Table 1.2. Identification and Listing Status of Six Listed Species Included in This Assessment

Species	Status	Date Listed	Listing Agency
Shortnose sturgeon (*Acipenser brevirostrum*)	Endangered 32 FR 4001; 38FR 41370	March 11, 1967	National Marine Fisheries Service (NMFS)
Dwarf wedgemussel (*Alasmidonta heterodon*)	Endangered 55FR 9447	March 4, 1990	U.S. Fish and Wildlife Services (USFWS)
Loggerhead turtle (*Caretta caretta*)	Threatened 32FR 4001; 43 FR 32800	July 28, 1978	USFWS and NMFS
Kemp's ridley turtle *Lepidochelys kempii*	Endangered 35 FR 18319-18322	December 2, 1970	USFWS and NMFS
Green turtle *Chelonia mydas*	Endangered 43 FR 32808	July 28, 1978	USFWS and NMFS
Leatherback turtle *Dermochelys coriacea*	Endangered 35 FR 8491-8498	June 2, 1970	USFWS and NMFS

In this endangered species assessment, direct and indirect effects to the six assessed species are evaluated in accordance with the screening-level methodology described in the Agency's Overview document (U.S. EPA, 2004). The indirect effects analysis in this assessment utilizes more refined data than is generally available for ecological risk assessment. Specifically, a robust set of microcosm and mesocosm data and aquatic ecosystem models are available for atrazine that allowed for a refinement of the indirect effects associated with potential aquatic community-level effects (via aquatic plant community structural change and subsequent habitat modification). Use of such information is consistent with the guidance provided in the Overview document (U.S. EPA, 2004), which specifies that "the assessment process may, on a case-by-case basis, incorporate additional methods, models, and lines of evidence that EPA finds technically appropriate for risk management objectives" (Section V, page 31 of U.S. EPA, 2004).

As part of the "effects determination", one of the following three conclusions is reached regarding the potential for atrazine to adversely affect the assessed species:

- "No effect"
- "May affect, but not likely to adversely affect"
- "Likely to adversely affect"

If during the screening-level assessment it is determined that there are no indirect effects, and LOCs for listed species are not exceeded for direct effects, a "no effect" determination is made based on atrazine's use within the designated action area. A description of the action area for the assessed species is provided in Section 2.5.

If a determination is made that use of atrazine within the action area may affect the listed species, additional information is considered to allow for further refinement and characterization of exposure and effects. Based on the additional characterization, the best available information is used to distinguish those actions that may affect, but are "not likely to adversely affect" from those actions that are "likely to adversely affect" a particular listed species.

The criteria used to make determinations that the effects of an action are not likely to adversely affect listed resources include the following:

- <u>Significance of Effect</u>: Insignificant effects are those that cannot be meaningfully measured, detected, or evaluated in the context of a level of effect where take occurs for even a single individual
    - "Take" in this context means to harass or harm
        - Harm includes significant habitat modification or degradation that results in death or injury to listed species by significantly impairing behavioral patterns such as breeding, feeding, or sheltering.
        - Harass is defined as actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.

- Likelihood of the Effect Occurring: Discountable effects are those that are extremely unlikely to occur. Dose-response information is used to estimate the likelihood of effects.

- Adverse Nature of Effect: Effects that are wholly beneficial without any adverse effects are not considered adverse. See Assessment Endpoints in Table 2.5.

2.2. Scope of Assessment

2.2.1. Assessed Uses

This risk assessment is for currently registered uses of atrazine. Atrazine is currently registered as an herbicide in the U.S. to control annual broadleaf and grass weeds in corn, sorghum, sugarcane, and other crops. In addition to food crops, atrazine is also used on a variety of non-food crops, forests, residential/industrial uses, golf course turf, recreational areas, and rights-of-way. Although atrazine is used on a number of commodities and non-agricultural areas, this assessment addresses atrazine use in the Chesapeake Bay Watershed. Predominant uses in the Chesapeake Bay include corn, sorghum, residential uses, turf, rights-of-ways, and fallow/idle land.

Application rates and use patterns for these uses are described in Section 3. Use data considered in this assessment were obtained from the available labels and from U.S. EPA's Biological and Economic Analysis Division (BEAD) as discussed in Section 2.6.2.

2.2.2. Chemicals Assessed

This ecological risk assessment includes all potential ecological stressors resulting from the use of atrazine within the Chesapeake Bay watershed, including atrazine and its potential degradates of concern. Degradates of concern may include those that are found at significant (>10% by weight relative to parent) concentrations in available degradation studies and those that are of toxicological concern. Atrazine degradates and their routes of formation are summarized in Table 2.1 below.

Table 2.1. Summary of Formation Pathway of Atrazine Degradates					
Degradate	Formation Pathway				
	Photolysis in Water	Photolysis in Soil	Aerobic Metabolism in Soil	Anaerobic Metabolism in Soil	Anaerobic Metabolism in Water
Deethylatrazine (DEA)	X (18%)[a]	X (18%)[a]	X	X	X
Deisopropylatrazine (DIA)	X	X	X	X	X
Diaminochlorotriazine (DACT)	X (15%)[a]	X	X	X	X
Hydroxyatrazine (HA)		X	X	X	X
Deethylhydroxyatrazine (DEHA)		X	X		
Deisopropylhydroxyatrazine (DIHA)		X	X		

[a] Values in parentheses are percentage of parent formed; only values for major (>10%) degradates are shown. See U.S. EPA, 2003a for additional discussion on these degradates.

Degradates of atrazine include hydroxyatrazine (HA), deethylatrazine (DEA), deisopropylatrazine (DIA), and diaminochloroatrazine (DACT). Comparison of available toxicity information for the degradates of atrazine indicates lesser aquatic toxicity than the parent for fish, aquatic invertebrates, and aquatic plants. Specifically, the available degradate toxicity data for HA indicate that it is not toxic to freshwater fish and invertebrates at the limit of its solubility in water. In addition, no adverse effects were observed in fish or daphnids at DACT concentrations up to 100 mg/L. Acute toxicity values for DIA are 3- and 36-fold less sensitive than acute toxicity values for atrazine in fish and daphnids, respectively. In addition, available aquatic plant degradate toxicity data for HA, DEA, DIA, and DACT report non-definitive EC_{50} values (i.e., 50% effect was not observed at the highest test concentrations) at concentrations that are at least 700 times higher than the lowest reported aquatic plant EC_{50} value for parent atrazine. Although degradate toxicity data are not available for terrestrial plants, lesser or equivalent toxicity is assumed, given the available ecotoxicological information for other taxonomic groups including aquatic plants and the likelihood that the degradates of atrazine may lose efficacy as an herbicide. Therefore, given the lesser toxicity of the degradates as compared to the parent, and the relatively small proportion of the degradates expected to be in the environment and available for exposure relative to atrazine, the focus of this assessment is parent atrazine. Additional details on available toxicity data for the degradates are provided in Section 4 and Appendix A.

The results of available toxicity data for mixtures of atrazine with other pesticides are presented in Section A.6 of Appendix A. According to the available data, other pesticides may combine with atrazine to produce synergistic, additive, and/or antagonistic toxic effects. Synergistic effects with atrazine have been demonstrated for a number of organophosphate insecticides including diazanon, chlorpyrifos, and methyl parathion, as well as herbicides including alachlor. If chemicals that show synergistic effects with atrazine are present in the environment in combination with atrazine, the toxicity of atrazine may be increased, offset by other environmental factors, or even reduced by the presence of antagonistic contaminants if they are also present in the mixture. The variety of chemical interactions presented in the available data set suggest that the toxic effect of atrazine, in combination with other pesticides used in the environment, can be a function of many factors including but not necessarily limited to: (1) the exposed species, (2) the co-contaminants in the mixture, (3) the ratio of atrazine and co-contaminant concentrations, (4) differences in the pattern and duration of exposure among contaminants, and (5) the differential effects of other physical/chemical characteristics of the receiving waters (e.g. organic matter present in sediment and suspended water). Quantitatively predicting the combined effects of all these variables on mixture toxicity to any given taxa with confidence is beyond the capabilities of the available data. However, a qualitative discussion of implications of the available pesticide mixture effects data involving atrazine on the confidence of risk assessment conclusions for the freshwater mussels is addressed as part of the uncertainty analysis for this effects determination.

However, DEA has been shown to be of similar toxicity to birds on an acute oral basis compared with atrazine. Other dealkylatrazine degradates have been shown to be more acutely toxic to female rats and more developmentally toxic to gestating rat pups than the parent atrazine (Table 2.2 below). Acute avian studies suggest that DIA is less toxic than atrazine to birds on an acute oral basis. No avian toxicity data for DACT are available; therefore, based on the equivalent toxicity in mammals, DACT may also be of toxicological concern in birds, which are used as surrogate species for turtles. For this reason, DACT and DEA were both considered qualitatively in the risk assessment for sea turtles. Because preliminary analyses described in Section 5 indicate that the degradates would be expected to have negligible impact for sea turtles, aquatic concentrations of the degradates were not quantified and were instead discussed qualitatively in the Risk Characterization (Section 5).

Table 2.2. Summary of Available Degradate Toxicity Data in Birds and Mammals

Chemical	Acute Bird LD50 (mg/kg-bw)	Acute Mammal LD50 (female rats; mg/kg-bw)	Mammal Developmental NOAEC (mg/kg-bw)
Atrazine	940 (MRID 00024721)	1200 (U.S. EPA, 2003a)	200 (U.S. EPA, 2003a)
HA	>2000 (MRID 46500008)	Not available	500 (U.S. EPA, 2003a)
DEA	768 (MRID 46500009)	670 (U.S. EPA, 2003a)	25 (U.S. EPA, 2003a)
DIA	>2000 (MRID 46500007)	810 (U.S. EPA, 2003a)	5 (U.S. EPA, 2003a)
DACT	Not available	Not available	50 (U.S. EPA, 2003a)

2.2.3. Species Assessed

A brief introduction to the six listed species assessed, including a summary of habitat, diet, and reproduction data relevant to ecological risk assessment for the Chesapeake Bay and its source waters is presented below. A more comprehensive discussion of the biology and ecology of the six assessed species is provided in Appendix D and in the Risk Characterization (Section 5).

2.2.3.1. Dwarf Wedgemussel

The dwarf wedgemussel is an Atlantic Coast freshwater mussel usually found in sand, firm muddy sand, and gravel bottoms in rivers of varying sizes with slow to moderate current. To survive, the dwarf wedgemussel needs silt-free, stable, stream beds and well oxygenated water (U.S. EPA, 2003b). Host fish (see Appendix D for information on the life cycle of the dwarf wedgemussel) for populations in the Chesapeake Bay are unknown for this species. The dwarf wedgemussel filter feeds on suspended detritus, phytoplankton, and zooplankton. Known locations of the dwarf wedgemussel within the Chesapeake Bay watershed are summarized in Table 2.3 below (U.S. FWS, 1993; Maryland Department of Natural Resources (MD DNR), 2006; Virginia Department of Game and Inland Fisheries (VA DGIF), 2006 [DWM_locations_dist1783. Vector digital data. Acquired August 01, 2006].

Table 2.3. Known Locations of Dwarf Wedgemussels in the Chesapeake Bay Watershed			
Location	**County, State**	**Description[a]**	**Status of Population and Major Threats[b]**
Tuckahoe Creek Drainage			
Norwich Creek	Queen Anne's and Talbot Counties, Maryland	Headwater streams	**Status:** Poor, not reproducing **Threats:** Non-point chemical pollution; sedimentation from agriculture; population density too low to allow successful reproduction; residential, highway, or industrial development
Long Marsh Ditch; Mason Branch	Queen Anne's/Caroline Counties, MD	Headwater streams	**Status:** Poor, not reproducing **Threats:** Non-point chemical pollution, sedimentation from forestry operations; sedimentation from agriculture; population density too low to allow successful reproduction; headwater channelization and "stream improvement" projects Mason Branch and Long Marsh Ditch records likely represent a single population
Potomac River Drainage			
McIntosh Run	Saint Mary's County, Maryland	Headwater streams	**Status:** Fair, reproducing **Threats:** Residential, highway, or industrial development
Nanjemoy Creek	Charles County, Maryland	Headwater streams	**Status:** Fair, reproducing **Threats:** Not listed
Aquia Creek	Stafford County, Virginia	Headwater streams	**Status:** Fair to good **Threats:** Non-point chemical pollution; Sedimentation from forestry operations; Sedimentation from agriculture; Residential, highway, or industrial development
York River Drainage			
South Anna River	Louisa and Hanover Counties, VA	Headwater streams, mid-level reach	**Status:** Poor **Threats:** Sedimentation from forestry operations; sedimentation from agriculture; population density too low to allow successful reproduction; residential, highway, or industrial development
Po River	Spotsylvania County, VA	Headwater streams, mid-level reach	**Status:** Not listed **Threats:** Not listed
Rappahannock River Drainage			
Rappahannock River	Spotsylvania County	Headwater streams, mid-level reach	Location data from Virginia Department of Game and Inland Fisheries, 2006 (DWM_locations_dist1783. Vector digital data. Acquired August 01, 2006.)
Carter Run	Fauquier County	Headwater streams	
Southeast Creek or Corsica River Drainage			
Browns branch, Granny Finley, Southeast Creek tributary	Queen Anne's County, MD	Headwater streams	Location data provided by Maryland Department of Natural Resources, 2006 Brown's branch, Granny Finley branch, and Southeast creek tributary records may represent a single metapopulation
Corsica River tributary			

[a] See Section 2.4 for description of stream classification
[b] Status and threats information from U.S. FWS (1993), status and threats were not available for populations not included in U.S. FWS (1993).

2.2.3.2. Shortnose Sturgeon

The most recent documentation of shortnose sturgeon locations are from incidental capture via the U.S. Fish and Wildlife Service Reward program for Atlantic Sturgeon, which began in 1996 in the Chesapeake Bay and its tributaries. Shortnose sturgeon were primarily captured in the upper Chesapeake Bay north of Hart-Miller Island (Figure 2-1). However, sturgeon have also been captured in the lower Susquehanna River, Bohemia River, Potomac River, and Elk River, south of the Bay Bridge near Kent Island, near Howell Point, near Hoopers Island, and in Fishing Bay (U.S. EPA, 2003b). Historical records indicate that shortnose sturgeon have also been documented in the Chesapeake Bay, the Potomac River, near the mouth of the Susquehanna river, and near the mouth of the James and Rappahannock rivers (U.S. EPA, 2003b; NMFS, 1998).

In many river systems, shortnose sturgeon appear to spend most of their life in their natal river systems and only occasionally enter higher salinity environments. They are benthic omnivores and continuously feed on benthic and epibenthic invertebrates, including mollusks, crustaceans and oligochaete worms (NMFS, 1998; U.S. EPA, 2003b).

Shortnose sturgeon depend on free-flowing rivers and seasonal floods to provide suitable spawning habitat. For shortnose sturgeon, spawning grounds have been found to consist mainly of gravel or rubble substrate in regions of fast flow. Flowing water provides oxygen, allows for the dispersal of eggs, and assists in excluding predators. Seasonal floods scour substrates free of sand and silt, which might suffocate eggs (U.S. EPA, 2003b).

Shortnose sturgeon spawn in upper, freshwater sections of rivers and feed and overwinter in both freshwater and saline habitats. In populations that have free access to the total length of a river (absence of dams), spawning areas are located at the farthest accessible upstream reach of the river, often just below the fall line (U.S. EPA, 2003b). Tributaries of the Chesapeake Bay that appear to have suitable spawning habitat for the shortnose sturgeon include the Potomac, Rappahannock, James, York, Susquehanna, Gunpowder, and Patuxent Rivers (U.S. EPA, 2003b). Other scientists believe that very little if any suitable spawning habitat remains for shortnose sturgeon because of past sedimentation in tidal freshwater spawning reaches (U.S. EPA, 2003b).

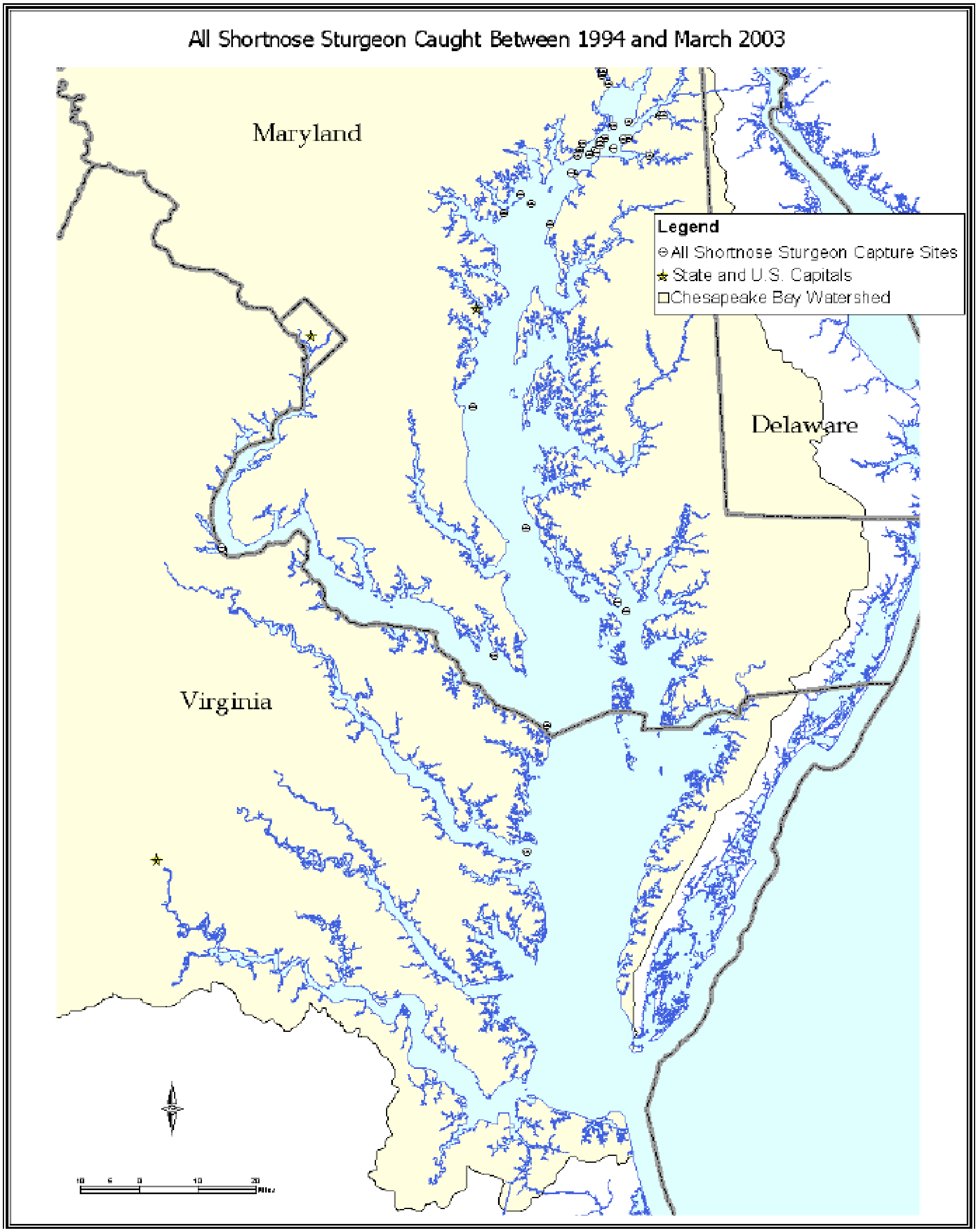


Figure 2-1. Shortnose Sturgeon Captured Between 1994 and 2003
(Source: U.S. EPA, 2003b)

2.2.3.3. Loggerhead Turtle

Between 2,000 and 10,000 loggerhead sea turtles enter the Chesapeake Bay each spring/summer from May to early November (Kimmel, *et al.*, 2006) when the sea temperatures rise to 18-20^{o} C. The majority are juveniles that utilize the Bay seasonally as a feeding ground (http://www.chesapeakebay.net/info/seaturtle.cfm; http://www.2fla.com/loggerhead.htm, accessed May 16, 2006; http://www.fisheries.vims.edu/turtletracking/stsp.html, accessed March 10, 2006). They live along the channel edges (17 to 43 feet), forage on the bottom, and appear to have foraging site fidelity (Byles 1988; Keinath *et al.* 1987; Kimmel, Driscoll and Brush 2006). Many loggerheads remain in the Virginia portion of the Bay, where salinities are higher (http://www.chesapeakebay.net/info/seaturtle.cfm, accessed May 16, 2006). Loggerheads concentrate their feeding around river mouths and areas of the Bay deeper than 13 feet. The loggerhead's range within the Bay includes primarily the main body of the Bay, river mouths, estuarine inlets, and river main stems.

This species is carnivorous throughout its life, and its diet varies by region. Hatchlings eat small animals living in sea grass mats, which are often distributed along drift lines and eddies. Juveniles and adults eat a wide variety of prey such as conchs, clams, crabs, horseshoe crabs, shrimp, sea urchins, sponges, fish, squid, and octopus (http://www.tpwd.state.tx.us/huntwild/wild/species/endang/animals/reptiles_amphibians/logghead.phtml).

All of the loggerhead nesting beaches are located outside of the Chesapeake Bay action area. Of all the sea turtles, loggerheads are known to nest the furthest north on the eastern coast of the United States. Some nest as far north as Virginia, just outside of the Chesapeake Bay; however, no available reports have been located to indicate that loggerheads nest within the Chesapeake Bay action area.

2.2.3.4. Kemp's Ridley Turtle

Kemp's ridley sea turtles are found within the Gulf of Mexico, and up through the Atlantic Coast of the United States, including the Chesapeake Bay. Developmental habitat for Kemp's ridley sea turtles, especially for foraging, has been identified within the Chesapeake Bay (Lutz and Musick 1997). Habitat for the Kemp's ridley turtle, outside of nesting beaches that are located outside of the Chesapeake Bay action area, includes mostly near shore and inshore waters, usually less than 50 m deep. Within the Chesapeake Bay, Kemp's ridleys tend to stay in coastal areas, foraging in beds of eelgrass. The primary range for the Kemp's ridley turtle within the Bay includes main body of the Bay, river mouths, estuarine inlets, and river main stems.

Kemp's ridleys are primarily carnivorous, with a preference for crab species, especially blue crab (*Callinectes sapidus*); however, they also feed on other crustaceans, mollusks, jellyfish, fish, and marine plants and algae. While hatchlings and juveniles normally feed at the surface, adult Kemp's ridleys feed on the bottom of coastal habitats as well.

2.2.3.5. Leatherback

Oceanic jellyfish are the preferred prey of the leatherback sea turtle throughout its life stages. They also may incidentally ingest algae, vertebrates, and other invertebrate species. The leatherback will often follow schools of jellyfish floating at the ocean's surface for its food source; however, prey is also found in benthic habitats, especially near coastal habitats (NMFS 1992a). It has been estimated that hatchling leatherbacks eat approximately their weight in jellyfish per day for growth and maintenance (Ernst, *et al.*, 1994). Jellyfish that are likely to be food items for the leatherback in the Chesapeake Bay include pink comb (*Beroe ovata*) and sea walnut (*Mnemiopsis leidyi*), among other species (www.chesapeakebay.net/baybio.htm, accessed May 17, 2006). There is no specific tracking information on leatherback turtles within the Bay. Their range may include the main body of the Bay, river mouths, and possibly main stems of rivers, and estuarine inlets.

Similar to the other sea turtles in the Bay, the preferred nesting beaches for female leatherbacks are outside the action area of the Chesapeake Bay and are generally high-energy beaches with proximity to deep water, generally rough seas, and sufficiently-sloped sandy beaches backed with vegetation (Ernst, *et al.*, 1994; http://www.fws.gov/northflorida/SeaTurtles/Turtle Factsheets/leatherback-sea-turtle.htm, Accessed April 13, 2006).

2.2.3.6. Green Turtle

Green turtles are distributed worldwide in tropical and subtropical waters; however, a very small number enter the Chesapeake Bay each summer. Occasional juveniles and adults have been identified in the Bay (Mansfield and Kimmel, personal communication 2006; http://www.chesapeakebay.net/info/seaturtle.cfm, accessed May 16, 2006; http://www.fisheries.vims.edu/turtletracking/stsp.html, accessed May 16, 2006). With its rich food supply and extensive shoals, the Chesapeake Bay provides ideal habitat for juvenile green turtle development. Many of the turtles remain in the Virginia portion of the Bay where salinities are higher. In general, green turtles are found in waters inside reefs, bays, and inlets (except when migrating). The turtles are attracted to lagoons and shoals with an abundance of marine grass and algae (NMFS 1991a). Like the leatherback, there is no specific tracking information on greens within the Bay. Their primary range may include the main body of the Bay, river mouths, and possibly main stems of rivers and estuarine inlets.

Hatchling green turtles eat a variety of plants and animals, but adults feed almost exclusively on sea grasses (especially *Sargassum* spp.) and marine algae with small amounts of animal foods such as sponges, crustaceans, sea urchins, and mollusks. Within the Chesapeake Bay, green turtle diet items likely include eelgrass, widgeon grass and algae, though there is no available data on specific gut contents. (www.fws.gov/northflorida/SeaTurtles/Turtle Factsheets/green-sea-turtle.htm, Accessed April 13, 2006; www.tpwd.state.tx.us/huntwild/wild/species/endang/animals/reptiles_amphibians/greentur.phtml, accessed May 16, 2006).

As with the other sea turtle species in the Bay, all nesting beaches for the green turtle are located outside of the Chesapeake Bay action area.

2.3 Previous Assessments

In January 2003, EPA completed a refined risk assessment that evaluated the potential impacts of atrazine on the environment (USEPA, 2003a). This assessment was based on toxicity data from laboratories as well as microcosm and mesocosm field studies coupled with exposure data including model-estimated environmental concentrations and a substantial amount of monitoring data from freshwater streams, lakes, reservoirs, and estuarine areas. Additionally, incident reports of adverse effects on aquatic and terrestrial organisms associated with the use of atrazine were also evaluated. In the refined assessment, risk is described in terms of the likelihood that concentrations in water bodies (i.e., monitoring sites in lakes/reservoirs, streams, and estuarine areas) may equal or exceed concentrations shown to cause adverse effects in laboratory and field-based toxicity studies. The results of the refined aquatic ecological assessment indicated that exposure to atrazine is likely to result in community-level and population-level effects to aquatic communities at concentrations greater than or equal to 10-20 µg/L on a recurrent basis or over a prolonged period of time.

The results of the ecological assessments for atrazine are fully discussed in the January 31, 2003, Interim Reregistration Eligibility Decision (IRED)[1]. The assessment identified the need for the following information related to potential ecological risks: 1) a monitoring program to identify and evaluate potentially vulnerable water bodies in corn, sorghum, and sugarcane use areas; and 2) further information on potential amphibian gonadal developmental responses to atrazine. On October 31, 2003, EPA issued an addendum that updated the IRED issued on January 31, 2003. This addendum described new scientific developments pertaining to monitoring of watersheds and potential effects of atrazine on endocrine-mediated pathways of amphibian gonadal development.

As discussed in the October 2003 IRED, an evaluation of the submitted studies regarding the potential effects of atrazine on amphibian gonadal development was conducted and presented in the form of a white paper for external peer review to a FIFRA Scientific Advisory Panel (SAP) in June 2003[2]. In the white paper dated May 29, 2003, seventeen studies consisting of both open literature and registrant-submitted laboratory and field studies involving both native and non-native species of frogs were summarized. It was concluded that none of the studies fully accounted for environmental and animal husbandry factors capable of influencing endpoints that the studies were attempting to measure. It was also concluded that the current lines-of-evidence did not show that atrazine produced consistent effects across a range of exposure concentrations and amphibian species tested.

Based on this assessment (U.S. EPA, 2003a), it was concluded that there was sufficient evidence to formulate a hypothesis that atrazine exposure may impact gonadal development in

[1] The 2003 Interim Reregistration Eligibility Decision for atrazine is available via the internet at http://www.epa.gov/oppsrrd1/REDs/0001.pdf
[2] The Agency's May 2003 White Paper on Potential Developmental Effects of Atrazine on Amphibians is available via the internet at http://www.epa.gov/oscpmont/sap/meetings/2003/june/finaljune2002telconfreport.pdf.

amphibians, but there were insufficient data to confirm or refute the hypothesis (http://www.epa.gov/oscpmont/sap/meetings/2003/june/finaljune2002telconfreport.pdf). Because of the inconsistency and lack of reproducibility across studies and an absence of a dose-response relationship in the currently available data, it was determined that the data did not alter the conclusions reached in the January 2003 IRED regarding uncertainties related to atrazine's potential effects on amphibians. The SAP supported EPA in seeking additional data to reduce uncertainties regarding potential risk to amphibians. Subsequent data collection has followed the multi-tiered process outlined in the white paper to the SAP (U.S. EPA, 2003d). In addition to addressing uncertainty regarding the potential use of atrazine to cause these effects, these studies are expected to characterize the nature of any potential dose-response relationship. A data call-in for the first tier of amphibian studies was issued in 2005 and studies are on-going; however, as of this writing, the results are not available.

2.4. Characterization of Waters Inhabited by the Six Assessed Species

Because all six species assessed are aquatic, the conceptual model of exposure is based on the nature of surface water body types within the watershed. Surface water within the watershed consists of surface streams ranging from headwater streams (first and second order by the Strahler classification; Allan 1995) to mid-size reaches and rivers to main stem navigable rivers such as the Potomac and Susquehanna rivers. In addition, the watershed is defined by the main stem of the Chesapeake Bay, as well as the estuarine mouths of main rivers and a multitude of minor stream/estuary inlets, which rim the Bay but are not connected to any major river systems.

For the purposes of this assessment, the surface water network of the Chesapeake Bay watershed was divided into six broad classifications for comparison with monitoring data and modeled estimated exposure concentrations (EECs). These broad classes include headwater streams, mid range streams, major rivers, estuarine inlets of rivers (river mouths), minor estuarine inlets, and the main stem (or open water portion of the bay) of the Chesapeake Bay. Representative examples of this classification scheme are provided in Figures 2-2 through 2-8. These classifications are for characterization purposes only (e.g. comparing exposure estimates with regions) and do not define distinct regions within the watershed. In other words, the classification is qualitative in nature and does not define "bright lines" between regions but is based on a comparative analysis of the stream network. These classifications are used more fully in comparison with monitoring data collected specifically for the Chesapeake Bay watershed and this analysis is described in **Section 3.4**.

The principal significance of this type of scheme is to allow for a simplified comparison of species location information with monitoring data and modeled EECs. This type of generalization is particularly significant for modeled exposure estimates. Surface water modeling was conducted using the Pesticide Root Zone Mode and Exposure Analysis Modeling System (PRZM/EXAMS, described in more detail below) using existing scenarios linked to the standard water body for ecological assessments. The standard water body is a static water body and was developed to represent high-end estimates of atrazine that might be found in ecologically sensitive environments (i.e., headwater streams) near agricultural fields.

Characterization of the water bodies of the Chesapeake Bay watershed is used to compare PRZM/EXAMS exposure estimates to monitoring data from the assessed species location. In the Chesapeake Bay, exposure estimates generated using the static water body most closely represent short-term exposures in low-order streams, such as those found on the Eastern Shore where agriculture is dominant, and the minor type of estuary, where relevant land use abuts the water body. Exposure concentrations (particularly longer-term concentrations) in other water body types in the watershed (mid-size and main-stem rivers, estuarine mouths of rivers, and the main stem of the Bay) are not expected to be as well represented by the standard water body, thus, EECs from modeling likely over-estimate exposure in these settings. Analysis of the impact of flow on modeled predictions of peak and long-term exposures (see Section 3) suggests that model predictions are over-estimating longer-term exposures in water bodies with moderate to high flow, or with larger volumes, while providing a reasonable approximation of exposure in slowly flowing water bodies. Overall, the classification of water body types and the qualitative comparison of modeled EECs with the classes of water body types is useful for characterizing where exceedances may, or may not, be likely to occur.

A summary of the assessed species expected to be located in the six classes of water bodies assessed is in Table 2.4 below. Attempts to better characterize potential exposures in these settings are described in Section 3.3.

Table 2.4. Summary of Location of The Six Assessed Listed Species Within the Chesapeake Bay

Water Body Classification	**Species[a]**					
	Dwarf Wedge-mussel[b]	**Shortnose Sturgeon[c]**	**Kemp's Ridley Turtle[d]**	**Leatherback Turtle[d,e]**	**Loggerhead Turtle[d]**	**Green Turtle[d]**
Headwater streams	X					
Mid-range streams	X					
Major rivers		X	X	X	X	X
River mouths		X	X	X	X	X
Minor estuarine inlets			X	X	X	X
Main body of the Chesapeake Bay		X	X	X	X	X

[a] See Appendix D for additional information on species habitat.
[b] Dwarf wedgemussels are confined to small geographic locations and are found in streams with low to moderate flow.
[c] Juvenile sturgeons are found in freshwater rivers; adult sturgeons are found mostly at river mouths, but spawn in freshwater rivers.
[d] All of the turtles are transitory within the Bay and located in various habitats within the Bay.
[e] Leatherback turtles are highly pelagic and are typically found in open waters.

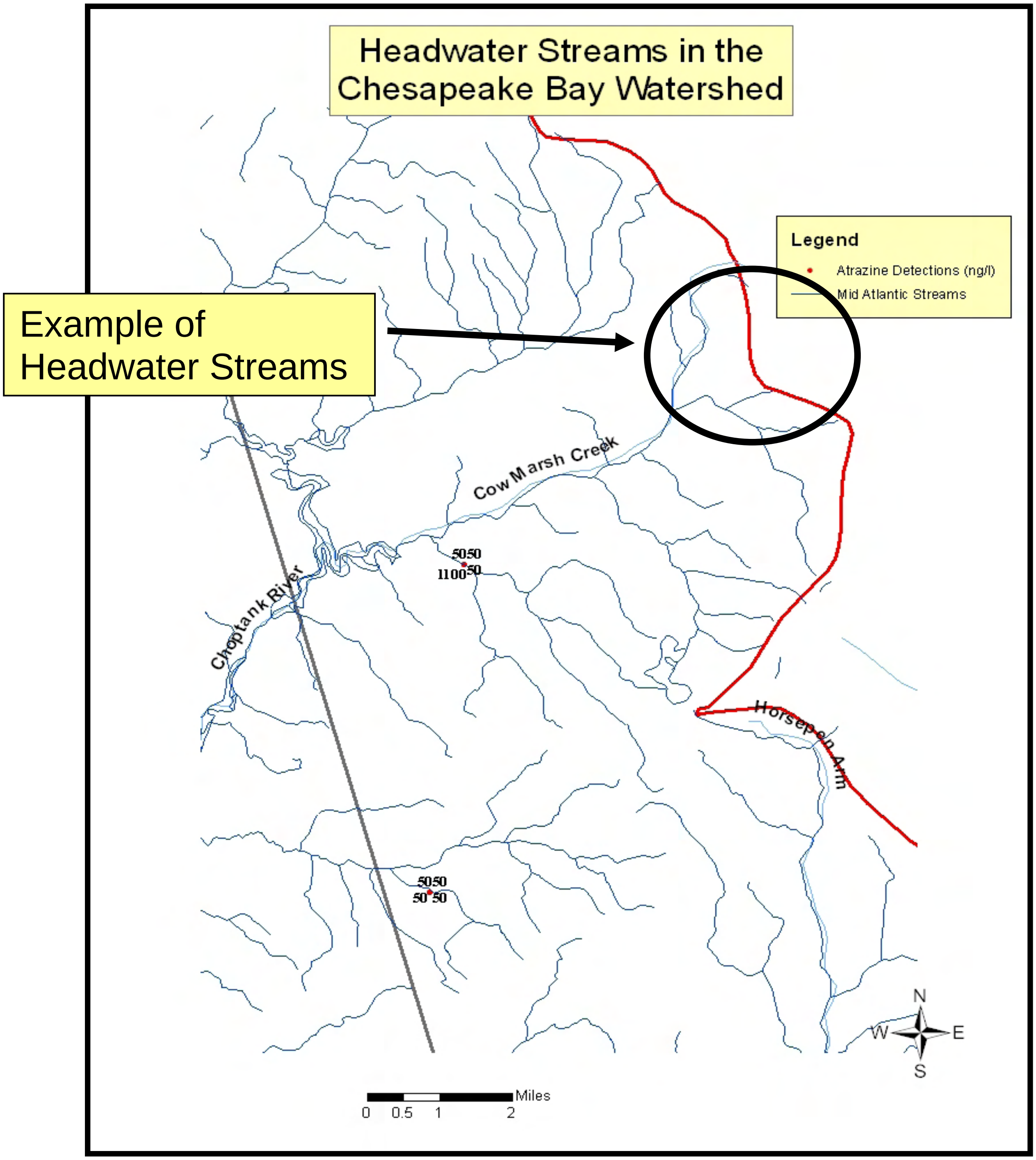


Figure 2-2. Conceptual Model of an Exposure Scenario Representing Headwater Streams in the Chesapeake Bay Watershed

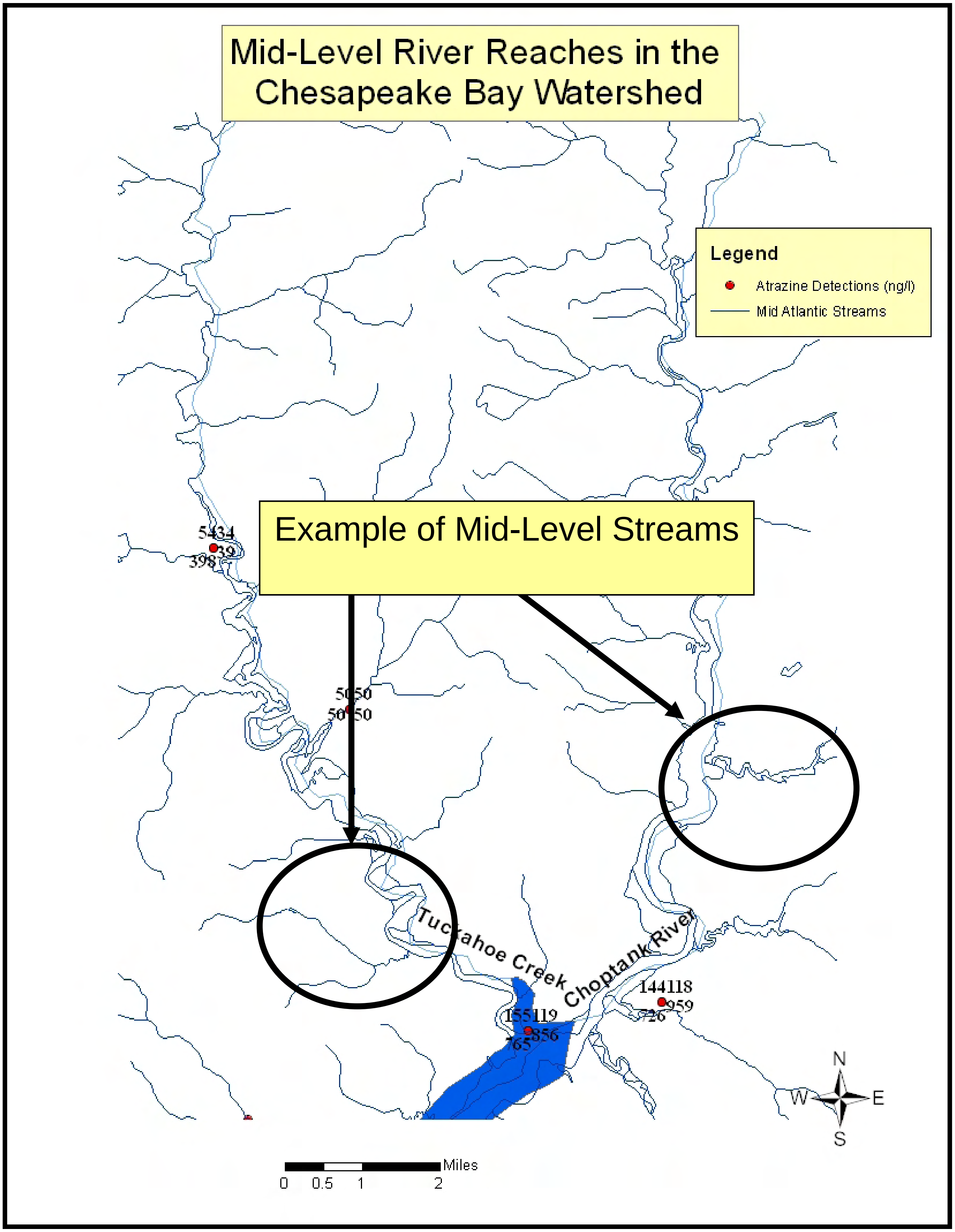


Figure 2-3. Conceptual Model of an Exposure Scenario Representing Mid-Level River Reach in the Chesapeake Bay Watershed

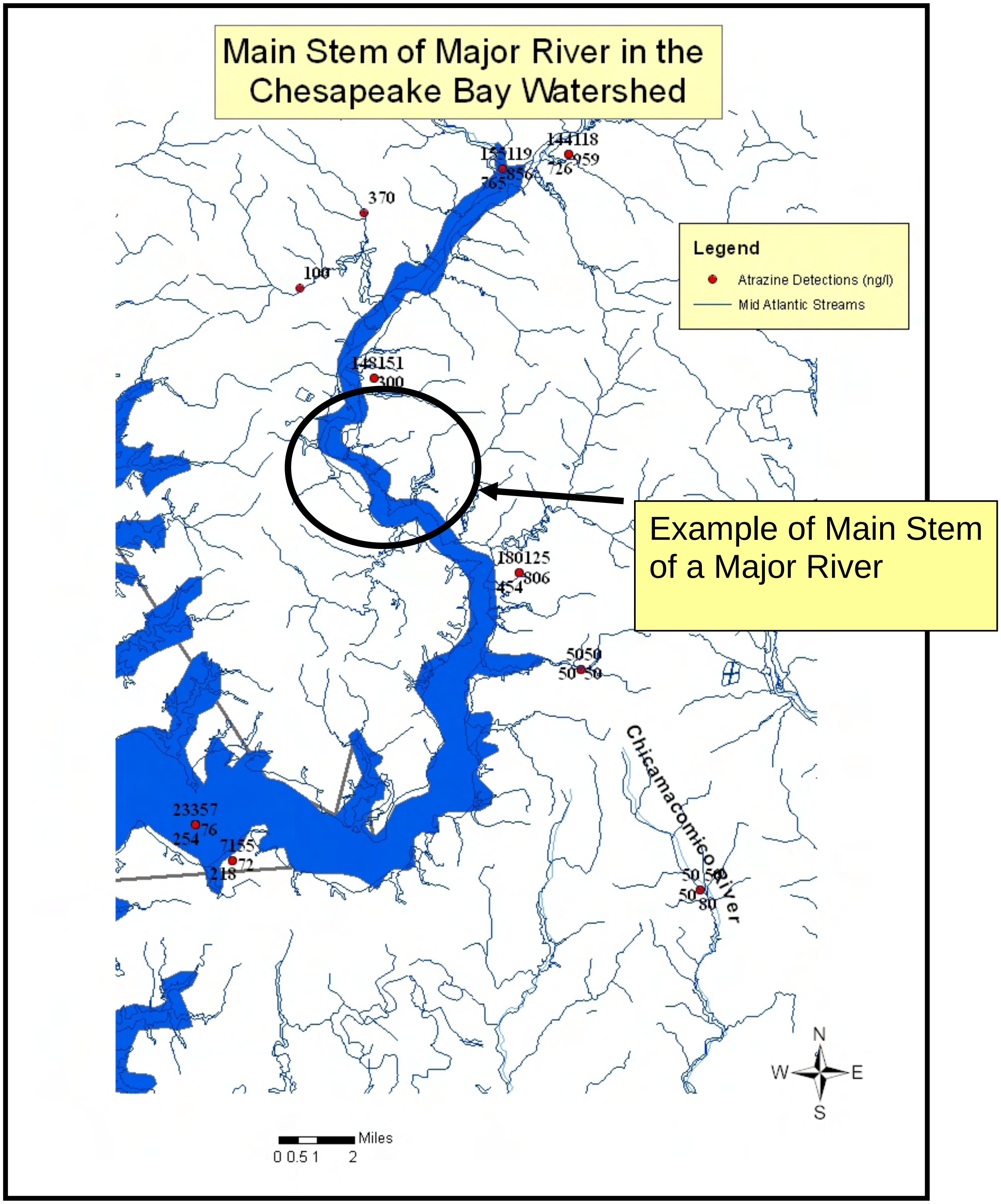


Figure 2-4. Conceptual Model of an Exposure Scenario Representing Major River Reach in the Chesapeake Bay Watershed

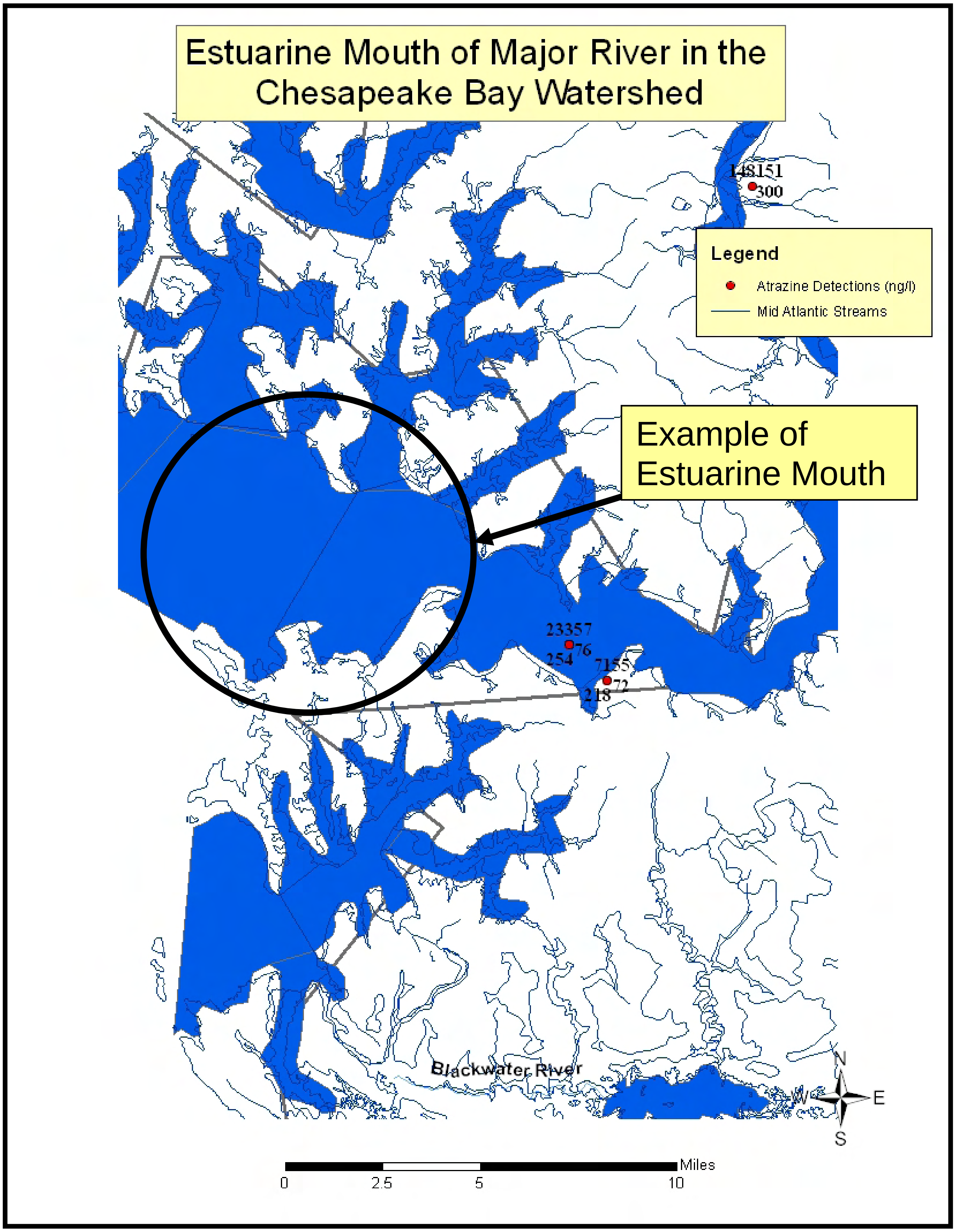


Figure 2-5. Conceptual Model of an Exposure Scenario Representing a Mouth of Major River in the Chesapeake Bay Watershed

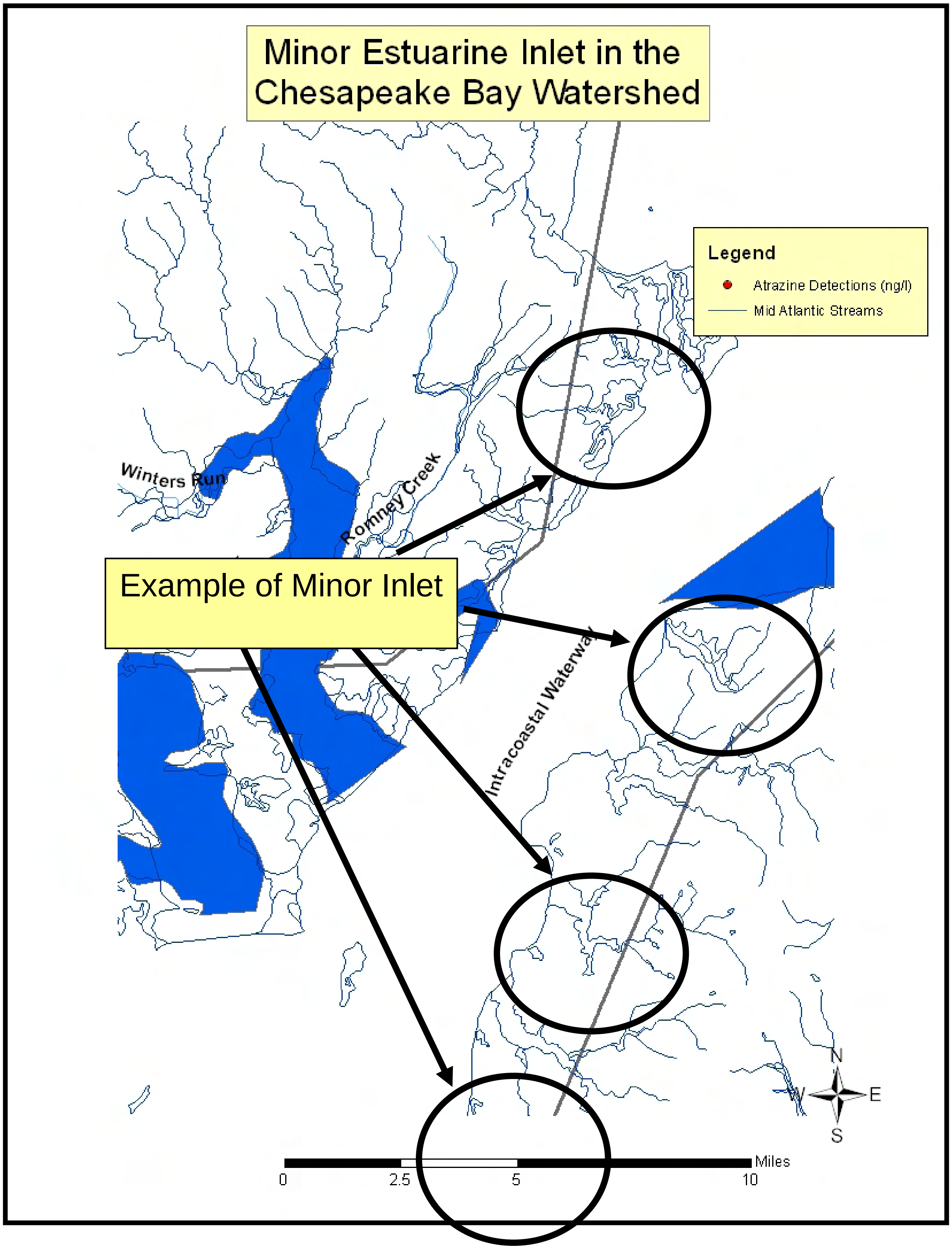


Figure 2-6. Conceptual Model of an Exposure Scenario Representing a Minor Estuarine Inlet in the Chesapeake Bay Watershed

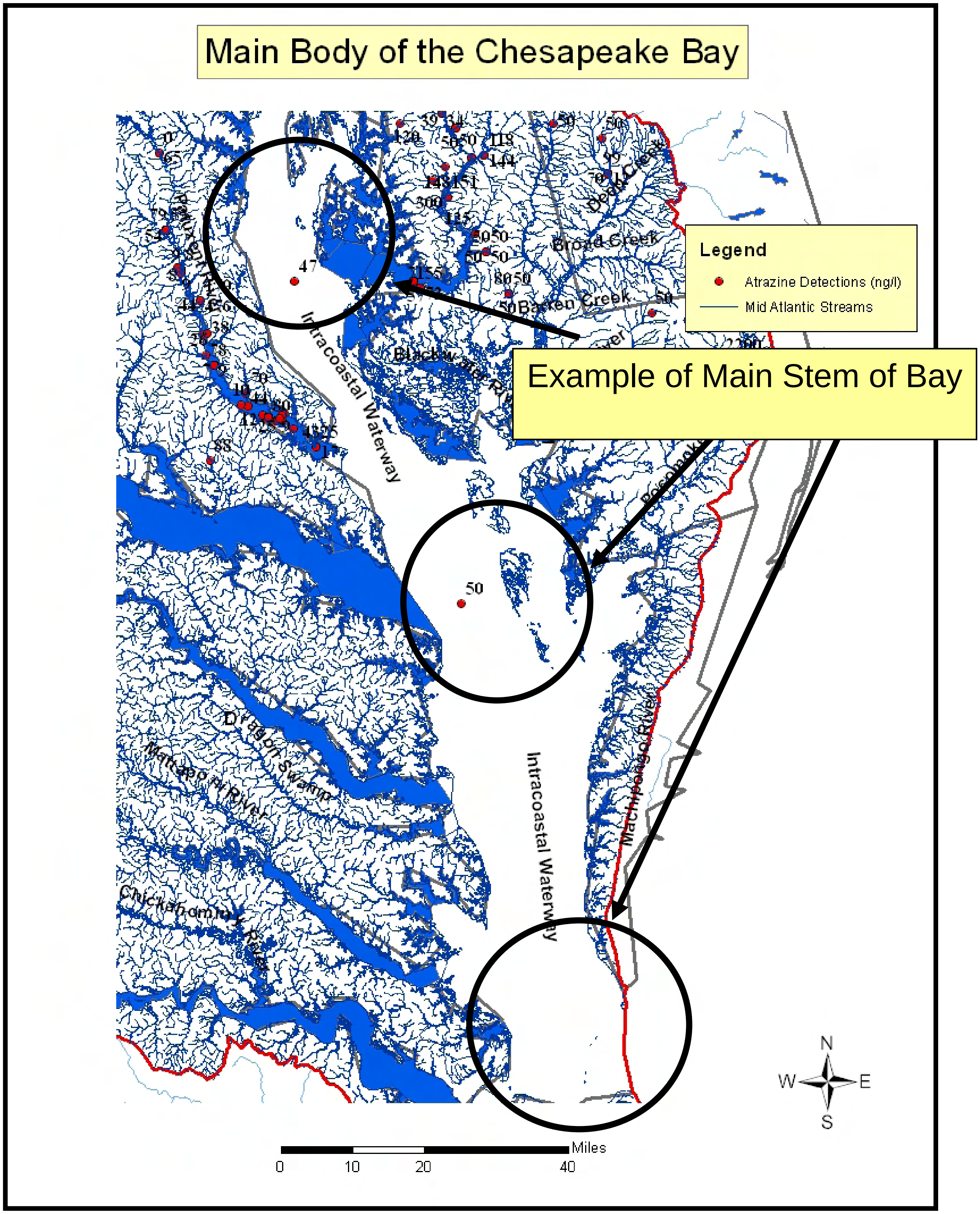


Figure 2-7. Conceptual Model of an Exposure Scenario Representing the Main Stem of the Chesapeake Bay

2.5. Action Area

It is recognized that the overall action area for the national registration of atrazine uses is likely to encompass considerable portions of the United States based on the large array of both agricultural and non-agricultural uses. However, the scope of this assessment limits consideration of the overall action area to those portions that may be applicable to the portion of the United States that includes the following federally listed species as they occur within the watershed of the Chesapeake Bay: shortnose sturgeon, dwarf wedgemussel, and loggerhead, green, leatherback, and Kemp's ridley sea turtles. Deriving the geographical extent of this portion of the action area is the product of consideration of the types of effects atrazine may be expected to have on the environment, the exposure levels to atrazine that are associated with those effects, and the best available information concerning the use of atrazine and its fate and transport within the Chesapeake Bay Watershed.

Modeled concentrations of atrazine for labeled uses expected to occur within the Chesapeake Bay watershed exceed established ecological risk levels of concern for aquatic plants and for some food items of the assessed species suggesting adverse effects on components of the environment is possible. The results of the screening-level assessment suggest that effects on components of the environment are possible anywhere in the Chesapeake Bay watershed up to and including the shallow water fringe of the Bay itself. Estimated atrazine concentrations are deemed most appropriate for headwater streams and minor inlets surrounding the Bay as described in Section 2.4; however, the potential for exceedances in other water body types cannot be precluded. An exception to this is the open waters of the Chesapeake Bay itself. Model predictions are not representative of the open waters of the Chesapeake Bay due to the large volume of water present, the influence of tidal fluxes, and differences in water chemistry relative to the EXAMS water body (freshwater), which are not accounted for in PRZM/EXAMS. Exposure in the open waters of the Chesapeake Bay is best represented by monitoring data which suggest that exposure levels in the open water are below levels of concern. Thus, the open water portion of the Bay is not included in the action area. It is likely that exposure concentrations predicted with modeling are not uniform throughout the watershed and portions of the action area may be below levels of concern. However, these areas cannot be definitively drawn on a map; therefore, the entire area described above includes all land draining to the Bay. Since the action area is defined as an area where effects may occur, and lack of LOC exceedance indicates a "no effect" conclusion, the LOC was used to define the action area. In addition, the action area was limited to the Cheseapeake Bay watershed by the terms of the settlement for the *Natural Resources Defense Council, Civ. No: 03-CV-02444 RDB (filed March 28, 2006).* Therefore, species populations that occur outside of the Chesapeake Bay watershed were not included in this assessment. More detail on the definition of the action area follows.

The named species being assessed as part of this endangered species assessment for the Chesapeake Bay are generally known to inhabit the main stem of the Bay as well as its main tributaries and headwaters of the main tributaries. Because this assessment is for multiple species, the action area is defined using the aggregated greatest extent of the various named species in conjunction with analysis of land use data, the registered use pattern for atrazine, and available use information for atrazine. In general, because of the varied use pattern that includes non-agricultural uses on residential, turf, forestry, and rights-of-way sites, the action area is initially defined as the entire Chesapeake Bay Watershed. Use of atrazine anywhere within the Chesapeake Bay watershed may lead to exposure within the Bay and its associated tributaries.

An evaluation of usage information was conducted to determine whether any or all of the area defined by the Chesapeake Bay Watershed should be included in the action area. Current labels were reviewed to determine which atrazine uses may be present within the defined area. A more detailed review of the local use information was completed. These data suggest that limited agricultural uses are present within the defined area and that non-agricultural uses cannot be precluded. Finally, local land cover data available from the Chesapeake Bay Program (http://www.chesapeakebay.net/index.cfm) were analyzed to refine the understanding of potential atrazine use in the areas immediately surrounding the Bay. The overall conclusion of this analysis is that certain agricultural uses in the Chesapeake Bay watershed can likely be excluded, and some non-agricultural uses of atrazine are unlikely to occur; however, no areas are excluded from the final action area based on usage and land cover data.

Finally, environmental fate properties of atrazine were evaluated to determine which routes of transport are likely to have an impact on the named species. Review of the environmental fate data as well as physico-chemical properties suggest that transport via overland flow and spray drift are likely to be the dominant routes of exposure (U.S. EPA, 2003a). Long-range atmospheric transport of pesticides could potentially contribute to concentrations in the aquatic habitat used by the listed species in the Bay. Given the physico-chemical profile for atrazine and the fact that atrazine has been detected in both air and rainfall samples, the potential for long range transport from outside the area defined by the Chesapeake Bay Watershed cannot be precluded, but is not expected to result in exposure concentrations that approach those predicted by modeling using the agricultural and residential scenarios (see Section 3.2).

Transportation of atrazine away from the site of application by both spray drift and volatilization is well documented. Spray drift is a localized route of transport off of the application site in the exposure assessments. Currently, quantitative models to address the longer-range transport of pesticides from application sites are not available. The environmental fate profile of atrazine, coupled with the available monitoring data, suggest that long-range transport of volatilized atrazine is a possible route of exposure to non-target organisms. Therefore, the full extent of the action area could be influenced by this route of exposure. However, given the extent of atrazine use within the Chesapeake Bay watershed, the magnitude of documented exposures in rainfall at or below available surface water monitoring data (as well as modeled estimates for surface water), the extent of the action area is defined by the transport processes of runoff and spray drift for the purposes of this assessment.

Based on this analysis, the action area for atrazine as it relates to the named species in this assessment is defined by the full extent of the Chesapeake Bay Watershed, with the exception of the open waters of the Bay because monitoring data suggest that exposure levels in the open water are below levels of concern. Figure 2-8 presents the action area graphically. Note that although the Chesapeake Bay is depicted in Figure 2-8, the open waters of the main trunk of the Chesapeake Bay are not included in the action area.

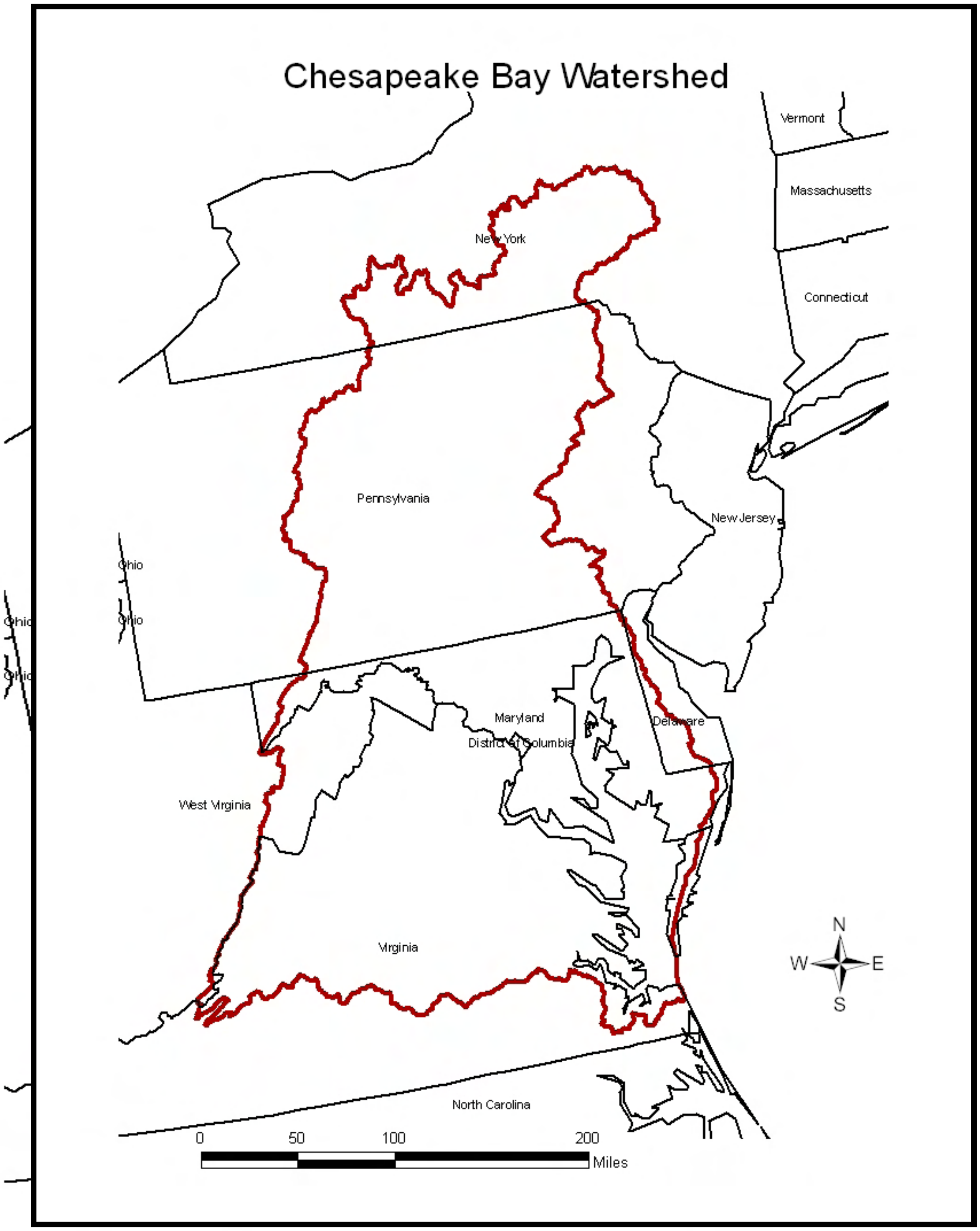


Figure 2-8. Action Area Defined by Chesapeake Bay Watershed for the Atrazine Endangered Species Assessment

As Figure 2-8 indicates, the Chesapeake Bay Watershed encompasses a vast drainage area. The watershed is defined by a diverse mixture of land covers, soils, and surface stream types, as well as a varied climate. Because of the limited geographic extent of the species being assessed relative to the overall watershed extent, the assessment focuses on those areas in closest proximity to the Chesapeake Bay. The underlying assumption is that use of atrazine in the area immediately surrounding the Bay is likely to be the most significant contributor to loading of atrazine to the Bay. This was considered a reasonable assumption considering that the areas of highest atrazine use in the Chesapeake Bay watershed are in the Eastern Shore and southeast Pennsylvania. Therefore, modeling was conducted for these areas and compared to local monitoring data. Scenarios modeled for these high use areas are expected to be representative of the highest exposures in the entire Chesapeake Bay watershed. It is possible that isolated areas outside of the area immediately surrounding the Bay could have higher exposures, however, it is expected that these locations are few and the impact of these exposures would be diluted prior to reaching the Bay. The available monitoring data tend to support this in that the highest atrazine concentrations detected in regional USGS NAWQA data are from sites closest to the Bay while sites further removed from the Bay tend to have lower concentrations.

2.6. Stressor Source and Distribution

2.6.1. Environmental Fate and Transport Summary

The following fate and transport description for atrazine was summarized based on information presented in the 2003 IRED (U.S. EPA, 2003a). In general, atrazine is expected to be mobile and persistent in the environment. The main route of dissipation is microbial degradation under aerobic conditions. Because of its persistence and mobility, atrazine is expected to reach surface and ground water. This is confirmed by the widespread detections of atrazine in surface water and ground water. Atrazine is persistent in soil, with a half-life (time until 50% of the parent atrazine remains) exceeding 1 year under some conditions (Armstrong *et al.*, 1967). Atrazine can contaminate nearby non-target plants, soil and surface water via spray drift during application or via runoff after application. Atrazine is applied directly to target plants during foliar application, but pre-plant and pre-emergent applications are generally far more prevalent.

The resistance of atrazine to abiotic hydrolysis (stable at pH 5, 7, and 9) and to direct aqueous photolysis (stable under sunlight at pH 7), and its moderate susceptibility to degradation in soil (aerobic laboratory half-lives of 3-4 months) indicates that atrazine is unlikely to undergo rapid degradation on foliage. Likewise, a relatively low Henry's Law constant (2.6×10^{-9} atm-m^3/mol) indicates that atrazine will probably not undergo rapid volatilization from foliage. However, its relatively low octanol/water partition coefficient (Log K_{ow} = 2 .7), and its relatively low soil/water partitioning (Freundlich K_{ads} values < 3 and often < 1) may somewhat offset the low Henry's Law constant value thereby possibly resulting in some volatilization from foliage. In addition, its relatively low adsorption characteristics indicate that atrazine may undergo substantial washoff from foliage.

In terrestrial field dissipation studies performed in Georgia, California, and Minnesota, atrazine dissipated with half lives of 13, 58, and 261 days, respectively. The inconsistency in these reported half-lives could be attributed to the temperature variation between the studies in which atrazine was seen to be more persistent in colder climate. Long-term field dissipation studies also indicated that atrazine could persist over a year in such climatic conditions. A forestry field dissipation study in Oregon (aerial application of 4 lb ai/A) estimated an 87-day half-life for atrazine on exposed soil, a 13-day half-life in foliage, and a 66-day half-life on leaf litter.

Atrazine is applied during pre-planting and/or pre-emergence applications or directly to turf. Atrazine is transported indirectly to soil due to incomplete interception during foliar application, and due to washoff subsequent to foliar application. The available laboratory and field data are reported above. For aquatic environments reported half-lives were much longer. In an anaerobic aquatic study, atrazine overall (total system), water, and sediment half-lives were given as 608, 578, and 330 days, respectively.

A number of degradates of atrazine were detected in laboratory and field environmental fate studies. Deethyl-atrazine (DEA) and deisopropyl-atrazine (DIA) were detected in all studies, and hydroxy-atrazine (HA) and diaminochloro-atrazine (DACT) were detected in all but one of the listed studies. Deethylhydroxy-atrazine (DEHA) and deisopropylhydroxy-atrazine (DIHA) were also detected in one of the aerobic studies.

All of the chloro-triazine and hydroxy-triazine degradates detected in the laboratory metabolism studies were present at less than the 10% of applied that is used to classify degradates as "major degradates", however, several of these degradates were detected at percentages greater than 10% in soil and aqueous photolysis studies. Insufficient data were available to estimate half-lives for these degradates from the available data. The dealkylated degradates are more mobile than parent atrazine, while HA is less mobile than atrazine and the dealkylated degradates.

2.6.2. Use Characterization

2.6.2.1. National Use Information

Atrazine has the second largest poundage of any herbicide in the U.S. and is widely used to control broadleaf and many other weeds, primarily in corn, sorghum and sugarcane (U.S. EPA, 2003a). As a selective herbicide, atrazine is applied pre-emergence and post-emergence. Figure 2.1 presents the national distribution of use of atrazine (Kaul et al., 2005).[3]

[3] Kaul et al. from U.S. EPA, Office of Pesticide Programs, Biological and Economic Analysis Division (BEAD)

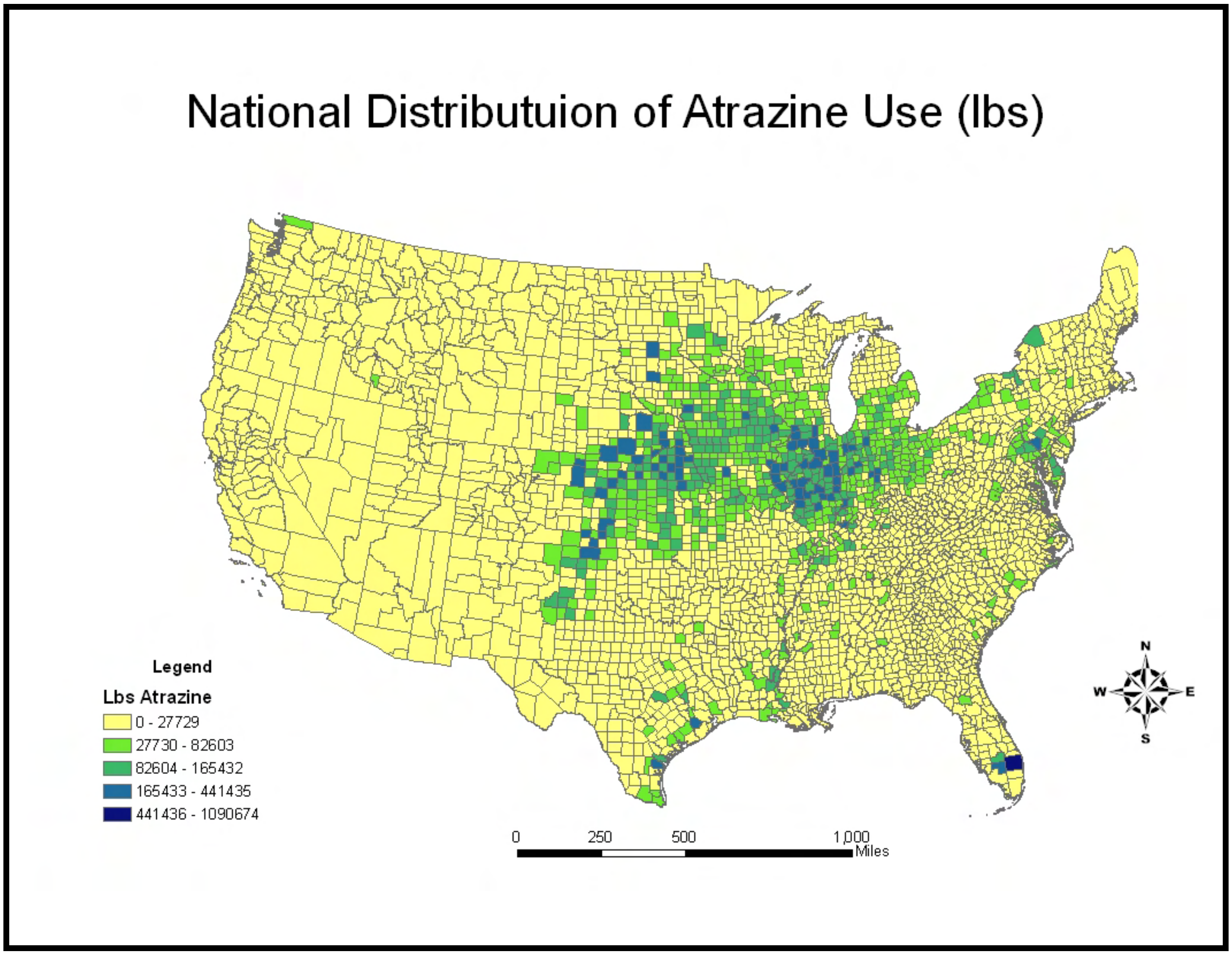


Figure 2-9. National Distribution of Atrazine Use

Atrazine is used on a variety of terrestrial food crops, non-food crops, forests, residential/industrial uses, golf course turf, recreational areas and rights-of-way. Atrazine yields season-long weed control in corn, sorghum and certain other crops. The major atrazine uses include: corn (83 percent of total ai produced per year - primarily applied pre-emergence), sorghum (11 percent of total ai produced), sugarcane (4 percent of total ai produced) and others (2 percent ai produced). Atrazine formulations include dry flowable, flowable liquid, liquid, water dispersible granule, wettable powder and coated fertilizer granule. The maximum registered use rate for atrazine is 4 lbs ai/acre; and 4 lbs ai/acre is the maximum, single application rate for the following uses: sugarcane, forest trees (softwoods, conifers), forest plantings, guava, macadamia nuts, ornamental sod (turf farms), ornamental and/or shade trees, and Christmas trees.

2.6.2.2. Use Information in the Chesapeake Bay Watershed

An analysis of available usage and land cover information was performed to determine which atrazine uses are likely to be present in the action area. The evaluation also is intended to place priority on those uses likely to be in closest proximity to the Chesapeake Bay. The analysis indicates that of all registered uses; the agricultural uses are likely to result in the highest exposures to the listed species. This is due to the preponderance of potential agricultural use sites (corn and sorghum) in the immediate vicinity of Chesapeake Bay.

Critical to the development of appropriate modeling scenarios and to Office of Pesticide Program's evaluation of the appropriate model inputs is an assessment of usage information. The Biological and Economic Analysis Division (BEAD) provided an analysis of both national and local use information (BEAD: Kaul, et al, 2005, Zinn, et al, 2006, Kaul, et al, 2005a, Kaul, et al, 2006b). State level usage data for Maryland, Virginia, and Pennsylvania obtained from USDA-NASS[4] and Doane (www.doane.com; the full dataset is not provided due to its proprietary nature), which were averaged together over the years 2000 to 2004 to calculate average annual usage statistics by state and crop for atrazine, including pounds of active ingredient applied, percent of crop treated, number of applications per acre, application rate per acre, and base acres treated. State level data from 1998 to 2004 were averaged together and extrapolated down to the county level based on apportioned to county level crop acreage from the 2002 USDA Agriculture of Census (AgCensus) data. In general, this information suggests that atrazine use on corn and Sorghum was approximately 500,000 lbs per year in Maryland, 600,000 lbs per year in Virginia, and 1,500,000 lbs per year in Pennsylvania on corn and sorghum. Other agricultural commodities on which atrazine is used (macadamia nut, guava, and sugarcane) are not present in the action area. Only information on agricultural uses were available.

In addition, general use information that indicates where the main uses of atrazine on agricultural crops are located is in Figure 2-10. A more complete summary of the use information used in this assessment may be found in Section 3.2.

[4] United States Depart of Agriculture (USDA), National Agricultural Statistics Service (NASS) Chemical Use Reports provide summary pesticide usage statistics for select agricultural use sites by chemical, crop and state. See http://www.usda.gov/nass/pubs/estindx1.htm#agchem.

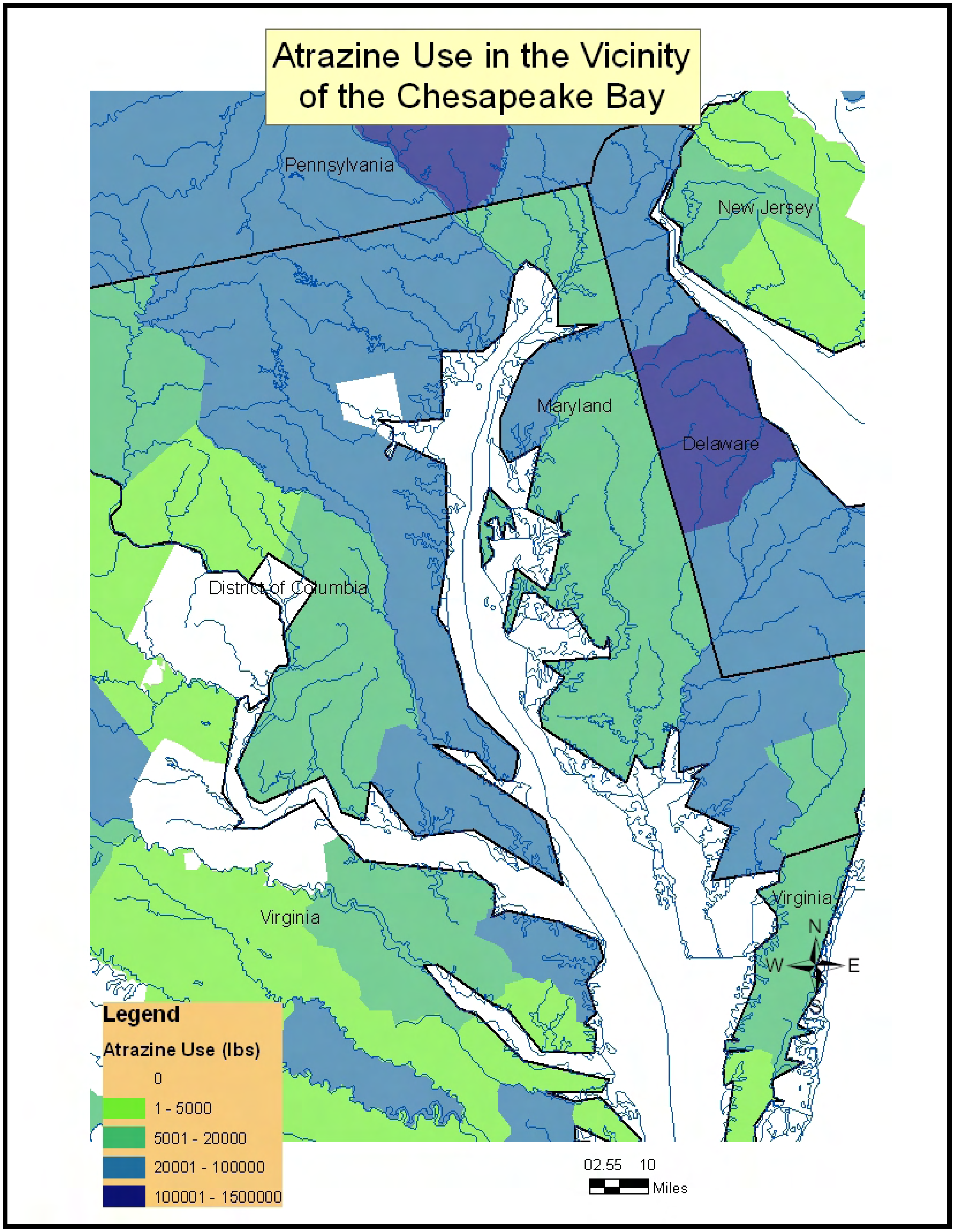


Figure 2-10. Atrazine Use (from BEAD; Kaul et al, 2005) in the Immediate Vicinity of the Chesapeake Bay

2.7. Mechanism of Herbicidal Action

Atrazine inhibits photosynthesis by stopping electron flow in Photosystem II. Triazine herbicides associate with a protein complex of the photosystem II in chloroplast photosynthetic membranes (Schulz *et al.*, 1990). The result is an inhibition in the transfer of electrons that in turn inhibits the formation and release of oxygen.

2.8. Assessment Endpoints and Measures of Ecological Effect

Assessment endpoints are defined as "explicit expressions of the actual environmental value that is to be protected."[5] Selection of the assessment endpoints is based on valued entities (i.e., shortnose sturgeon, dwarf wedgemussel, Kemp's ridley sea turtle, loggerhead sea turtle, leatherback sea turtle, and green sea turtle), the ecosystems potentially at risk (i.e., Chesapeake Bay and tributaries), the migration pathways of atrazine (i.e., runoff and spray drift), and the routes by which ecological receptors are exposed to atrazine-related contamination (i.e., direct contact and dietary exposure).

Assessment endpoints for the six species included in this assessment are direct toxic effects on the survival, reproduction, and growth of the species, as well as indirect effects, such as reduction of food supply and/or modification of their habitat. Each assessment endpoint requires one or more "measures of ecological effect," which are defined as changes in the attributes of an assessment endpoint itself or changes in a surrogate entity or attribute in response to exposure to a pesticide. Specific measures of ecological effect are evaluated based on acute and chronic toxicity information from the best available data. The reptile effects database is limited; therefore, birds are used as surrogate species for reptiles as outlined in U.S. EPA (2004). Also, effects data in sturgeon or dwarf wedgemussels are not available. Therefore, available toxicity data in surrogate freshwater and estuarine/marine fish species are used to assess potential direct effects to the shortnose sturgeon, and surrogate oyster and other invertebrate species are used to assess potential direct effects to the dwarf wedgemussel. Additional ecological effects data from the open literature, including effects data in reptiles and freshwater and saltwater microcosm and mesocosm data were also considered.

Measures of effect from microcosm and mesocosm data provide an expanded view of potential indirect effects of atrazine on aquatic organisms, their populations and communities in the laboratory, in simulated field situations, and in actual field situations. With respect to the microcosm and mesocosm data, threshold concentrations were determined from realistic and complex time variable atrazine exposure profiles (chemographs) for modeled aquatic community structure changes. Methods were developed to estimate ecological community responses for monitoring data sets of interest based on their relationship to micro- and mesocosm study results, and thus to determine whether a certain exposure profile within a particular use site and/or action area may have exceeded community-level threshold concentrations. Ecological modeling with the Comprehensive Aquatic Systems Model (CASM) (Bartell et al., 2000; Bartell et al., 1999; and DeAngelis et al., 1989) was used to integrate direct and indirect effects of atrazine to indicate changes to aquatic community structure and function.

---

[5] From U.S. EPA (1992). *Framework for Ecological Risk Assessment.* EPA/630/R-92/001.

A complete discussion of all the toxicity data available for this risk assessment, including use of the CASM model and associated threshold concentrations, and the resulting measures of ecological effect selected for each taxonomic group of concern are included in Section 4 of this document. A summary of the assessment endpoints and measures of ecological effect selected to characterize potential risks to the assessed species associated with exposure to atrazine is provided in Table 2.5.

Table 2.5. Summary of Assessment Endpoints and Measures of Ecological Effect to Six Listed Species		
Species	**Assessment Endpoint**	**Measures of Ecological Effect**[a]
Loggerhead, Kemp's ridley, green, and leatherback sea turtles	1. Survival, growth, and reproduction via direct effects	1a. Avian acute oral gavage (LD_{50}) and subacute dietary (LC50) toxicity studies 1b. Avian reproduction NOAEC
	2. Survival, growth, and reproduction via indirect effects on food supply	2a. Acute toxicity studies in most sensitive surrogate potential food items; LC_{50} and EC_{50} 2b. Most sensitive chronic and reproduction toxicity studies in potential food items; NOAEC 2c. Collective sensitivity distribution of toxicity values of surrogate food items with effects data 2d. Microcosm/mesocosm threshold concentrations showing aquatic community-level effects
	3. Survival, growth, and reproduction via indirect effects on habitat and/or primary productivity (i.e., aquatic plant community)	3a. Most sensitive aquatic plant EC_{50} 3b. Microcosm/mesocosm threshold concentrations showing aquatic primary productivity community-level effects
	4. Survival, growth, and reproduction via indirect effects on terrestrial vegetation (riparian habitat) required to maintain acceptable water quality.	4a. Monocot and dicot seedling emergence EC_{25} 4b. Monocot and dicot vegetative vigor EC_{25}
Shortnose Sturgeon	1. Survival, growth, and reproduction via direct effects	1a. Most sensitive fish acute LC_{50} 1b. Most sensitive fish chronic NOAEC
	2. Survival, growth, and reproduction via indirect effects on food supply	2a. Most sensitive aquatic invertebrate acute EC_{50} 2b. Most sensitive aquatic invertebrate chronic NOAEC 2c. Collective sensitivity distribution of toxicity values of all surrogate food items with effects data
	3. Survival, growth, and reproduction via indirect effects on habitat and/or primary productivity (i.e., aquatic plant community)	3a. Most sensitive aquatic vascular plant EC_{50} 3b. Non-vascular plant (algae) acute EC_{50} and NOAEC 3c. Microcosm/mesocosm threshold concentrations showing aquatic primary productivity community-level effects
	4. Survival, growth, and reproduction of shortnose sturgeon individuals via indirect effects on terrestrial vegetation (riparian habitat) required to maintain acceptable water quality and spawning habitat	4a. Monocot and dicot seedling emergence EC_{25} 4b. Monocot and dicot vegetative vigor EC_{25}
Dwarf Wedgemussel	1. Survival, growth, and reproduction via direct effects	1a. Eastern Oyster EC_{50}
	2. Survival, growth, and reproduction of individuals via indirect effects on food source or host (i.e., fish)	2a. Aquatic plant, invertebrate, and fish EC_{50} or LC_{50} and NOAEC
	3. Survival, growth, and reproduction via indirect effects on habitat and/or primary productivity (i.e., aquatic plant community)	3a. Most sensitive vascular aquatic plant EC_{50} 3b. Non-vascular plant (algae) EC_{50} 3c. Microcosm/mesocosm threshold concentrations showing aquatic primary productivity community-level effects
	4. Survival, growth, and reproduction Dwarf Wedgemussel via indirect effects on terrestrial vegetation (riparian habitat) required to maintain acceptable water quality and habitat.	4a. Monocot and dicot seedling emergence EC_{25} 4b. Monocot and dicot vegetative vigor EC_{25}

[a] Other data used in the characterization of potential risks to the assessed species are described in Section 4.

2.9. Conceptual Model

2.9.1. Risk Hypotheses

Risk hypotheses are specific assumptions about potential adverse effects (i.e., changes in assessment endpoints) and may be based on theory and logic, empirical data, mathematical models, or probability models (U.S. EPA, 1998). For this assessment, the risk is stressor-linked, where the stressor is the release of atrazine to the environment. The following risk hypotheses are presumed for this endangered species assessment:

- Atrazine in surface water and/or runoff/drift from treated areas into the Chesapeake Bay and its source waters may directly affect one or more of the assessed species by causing mortality or adversely affecting growth or reproduction;
- Atrazine in surface water and/or runoff/drift from treated areas may indirectly affect one or more of the assessed species by reducing or changing the composition of food supply in the Chesapeake Bay and its source waters;
- Atrazine in surface water and/or runoff/drift from treated areas may indirectly affect one or more of the assessed species by reducing or changing the composition of the plant community in the Chesapeake Bay and its source waters, thus affecting primary productivity and/or cover; and
- Atrazine in or runoff/drift from treated areas may indirectly affect one or more of the assessed species by reducing or changing the composition of the terrestrial plant community (i.e., riparian habitat) required to maintain acceptable water quality and stream characteristics in the Chesapeake Bay and its source waters. Runoff or drift into the terrestrial riparian buffer could damage or destroy the riparian vegetation, which provides important ecosystem services such as temperature regulation, energy input, and stream bank stabilization.

2.9.2. Diagram

The conceptual model is a graphic representation of the structure of the risk assessment. It specifies the stressor (atrazine for all assessed species, and two degradates (DEA and DACT) only for effects to turtles), release mechanisms, abiotic receiving media, biological receptor types, and effects endpoints of potential concern. The conceptual model for the atrazine endangered species assessment for the Chesapeake Bay is shown in Figure 2-11. Exposure routes shown in dashed lines are not quantitatively considered because the resulting exposures are expected to be sufficiently low as not to cause adverse effects to the species considered in this assessment.

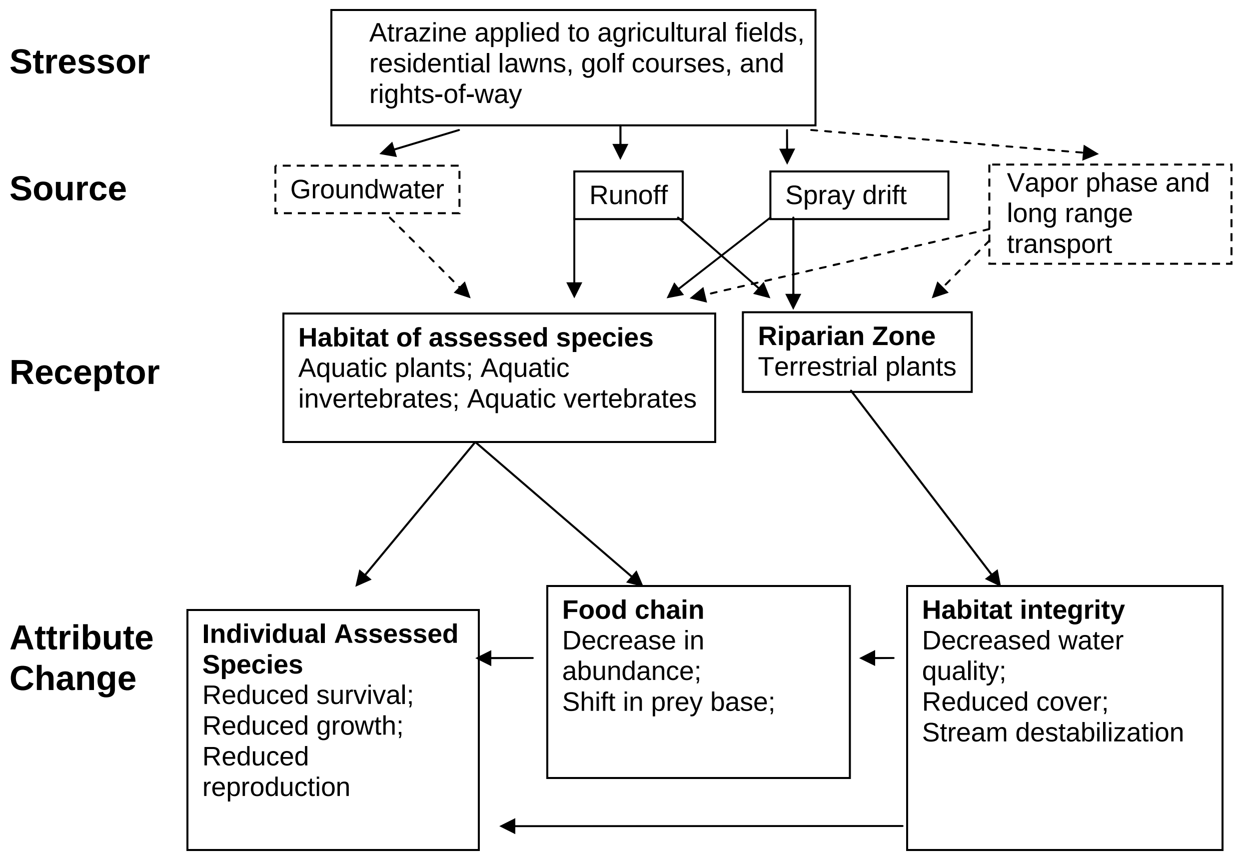


Figure 2-11. Conceptual Model

The conceptual model provides an overview of the expected exposure routes within the action area previously described in Section 2.5. In addition to the species included in this assessment, other aquatic receptors that may be potentially exposed to atrazine include freshwater and marine/estuarine fish, invertebrates, and plants. For turtles, the major exposure routes are expected to be ingestion of contaminated water and food items. For the sturgeon and mussel, the major routes of exposure are considered to be via the respiratory surface (gills) or the integument. Direct uptake and adsorption are the major routes of exposure for aquatic plants.

The source and mechanism of release of atrazine into surface water are ground and aerial application via foliar spray and coated fertilizer granules to agricultural (e.g., corn and sorghum) and non-agricultural areas (i.e., golf courses, residential lawns, rights-of-way, etc). Surface water runoff from the areas of atrazine application is assumed to follow topography, resulting in direct runoff to the Chesapeake Bay and its source waters. Spray drift and runoff of atrazine may also affect the foliage and seedlings of terrestrial plants that comprise the riparian habitat surrounding the Chesapeake Bay and its source waters. Additional release mechanisms include spray drift and atmospheric transport via volatilization, which may potentially transport site-related contaminants to the surrounding air. Atmospheric transport is not considered as a significant route of exposure for this assessment because the magnitude of documented exposures in rainfall are at or below available surface water and monitoring data, as well as modeled estimates of exposure. In addition, modeling tools are not available to predict the potential impact of long range atmospheric transport of atrazine.

Direct effects to freshwater or marine/estuarine organisms other than the six endangered species included in this assessment may occur from exposure to atrazine. Effects to these species may indirectly affect the sturgeon, mussel, and turtle species via reduction in food or habitat availability or quality.

In addition to aquatic receptors, terrestrial plants may also be exposed to spray drift and runoff from atrazine. Detrimental changes in the riparian vegetation adjacent to spawning areas of the sturgeon or the habitat of the dwarf wedgemussel may adversely affect the assessed species Indirect effects from riparian habitat alteration may include effects on water temperature, stream bank stability, and sediment loading. Additional information on riparian habitat is included in Section 5 (Risk Characterization).

Although the highest fish bioconcentration factor (BCF) of 8.5 (U.S. EPA, 2003c) suggests that bioconcentration is not a primary concern for atrazine, the principle exposure route for turtles is expected to be ingestion of contaminated food items. For this reason, bioconcentration is considered in this assessment as it relates to dietary exposure of contaminated food items by turtles as described in Section 3.

3.0 Exposure Assessment

An assessment of the potential for the listed species in the Chesapeake Bay watershed to be exposed to atrazine has been conducted. This exposure assessment represents a modification of the standard approach outlined in the Overview Document (U.S. EPA, 2004). The PRZM/EXAMS model has been used to provide estimates of exposure in the standard water body. Existing and new PRZM scenarios representing both agricultural and non-agricultural use sites were utilized. A modified approach for assessing residential uses included modeling a pervious (¼ acre lot) and impervious surface and weighting the output based on local data on the percentage of impervious surfaces in the region. Finally, non-standard durations of exposure to match available community-level ecotoxicity threshold concentrations were calculated. The highest overall exposures were predicted to occur from the agricultural uses of atrazine (corn, sorghum, and fallow/idle land) that are also likely to be in closest proximity to the species locations. Modeling indicates that peak exposure estimates from application of atrazine to forestry is similar to agricultural uses. However, model uncertainties and available information on atrazine use in forestry operations suggests these are over-estimates and should not be used for risk estimation. In general, the exposure assessment yields modeled peak exposure estimates that are consistent with local and national monitoring data, while the modeled annual average concentrations are two to ten times higher (depending on use site) than those seen in monitoring. The intermediate duration exposures (14-day, 21-day, 30-day, 60-day, and 90-day averages) cannot be estimated from the monitoring data due to insufficient sample frequency. However, additional modeling exercises that simulate flowing water bodies suggest that modeled longer-term exposures predicted using PRZM/EXAMS are likely overestimated for the environments inhabited by the assessed species. Taken together, the PRZM/EXAMS modeling and existing monitoring data suggest that longer-term exposures (days to weeks) are expected to be in the lower µg/L range. The general approach used for the exposure modeling and discussion of assumptions in the residential exposure modeling are outlined in Appendices C-1 and C-2.

3.1. Conceptual Model of Exposure

The general conceptual model of expected exposure in this assessment is that the highest exposures will occur in the headwater streams of the tributaries surrounding the Bay. Exposure models for deriving EECs within an estuarine water body such as the Chesapeake Bay are not available. However, available monitoring data obtained on May 3, 2006 from the U.S. EPA Chesapeake Bay Program (http://www.chesapeakebay.net/index.cfm) was used to estimate exposure to the assessed species within the main stem of the Chesapeake Bay. It is expected that, given the likelihood that significant amounts of atrazine are being used in watersheds of southern Pennsylvania and the Eastern Shore of Maryland and Virginia which drain to the Bay, the available monitoring data are insufficient to predict all possible exposure in these areas. Therefore, the best available monitoring data from multiple sources together with modeling estimates were used to characterize potential exposures to the assessed species.

Two general types of estimates were used to characterize potential exposures to the six assessed listed species: (1) modeling described in Sections 3.2 and 3.3; and (2) monitoring data from several sources described in Section 3.4. Initial screening-level exposure estimates, derived using PRZM/EXAMS and the standard water body scenario, were used in risk estimation for risk quotient calculation. For reasons discussed in Section 3.3, EECs based on the standard water body pond scenario are likely to be representative of short-term exposure concentrations in headwater streams and minor estuarine inlets, but may overestimate exposure in larger water bodies and/or flowing systems. Therefore, if the standard water body EECs resulted in LOC exceedance, exposure was further characterized using additional modeling exercises and available monitoring data. Table 3.1 below summarizes the data used to further characterize exposures when screening-level EECs exceed the LOC for each of the six assessed species. All methods and data are described in detail in below.

Table 3.1. Types of Waters Inhabited by the Assessed Species and Data Used in Exposure Refinement		
Water Body	**Data Source for Characterization of Standard Water Body EECs (if necessary)**	**Assessed Species Located In Water Body**
Headwater streams; mid-range streams	**Peak EEC**: PRZM/EXAMS standard water body. No refinement because peak EEC is considered representative of these waters. **Longer-term EECs[a]:** (1) Modified PRZM/EXAMS scenarios including incorporation of site specific flow data into reservoir scenario and variable volume water model (2) All available monitoring data.	Dwarf wedgemussel
Major rivers	**Peak EEC:** All available monitoring data in major rivers. **Longer-term EECs[a]:** (1) Modified PRZM/EXAMS scenarios including incorporation of flow data. (2) All available monitoring data.	Shortnose sturgeon, All four sea turtle species assessed
River mouths (estuarine inlets); main body of the Chesapeake Bay	**Peak and longer-term EEC:** Chesapeake Bay monitoring data.	Shortnose sturgeon, All four sea turtle species assessed
Minor estuarine inlets	**Peak**: PRZM/EXAMS standard water body. **Longer-term averages[a]:** (1) Qualitative analysis. (2) Chesapeake Bay monitoring data.	All four sea turtle species assessed

[a] Longer term exposure averages include durations of approximately several days and longer.

3.2. Use of Modeling to Characterize Potential Exposures to Atrazine in the Chesapeake Bay Watershed

3.2.1. Modeling Approach

The analysis of both available monitoring data and usage information indicates that the exposure assessment cannot rely exclusively on monitoring data. Although of high quality and generally located where higher concentrations are expected to occur (Eastern Shore corn belt), the timing, frequency of sampling, and location of the sample stations are unlikely to capture peak exposure to atrazine in the high use areas and are unlikely to have sufficient sample frequency to accurately estimate longer-term exposures. Therefore, an approach was implemented which blends the standard assessment approach using standard PRZM/EXAMS scenarios for corn, sorghum and turf with the non-agricultural scenarios (residential, impervious, rights-of- way, and fallow/idle land) recently developed for use in the Barton Springs endangered species assessment (U.S. EPA, 2006a). Available usage data (BEAD: Kaul et al, 2005, Kaul et al, 2006a, Kaul et al, 2006b) suggests that the heaviest usage of atrazine is likely to be on corn in the Eastern Shore; therefore, all selected modeling scenarios were run using the weather data from the Wilmington, Delaware meteorological station that is closest to the high use area.

A total of seven scenarios were utilized for the Chesapeake Bay endangered species assessment. Of these, three were developed as part of an endangered species assessment of atrazine for the Barton Springs salamander (U.S. EPA, 2006a). Two of the Barton Springs scenarios were used in tandem (residential and rights-of-way) with an impervious scenario (described below) while a third (fallow/idle land) was used by itself as a standard PRZM/EXAMS scenario. The remaining four scenarios (corn, sorghum, forestry, and turf) were taken from existing scenarios developed for other regions of the United States and modeled using weather data from the Eastern Shore. No additional scenarios were developed for this assessment. To address the potential use of atrazine on the labelled use sites, all of the scenarios have been modeled; however, the results were characterized to place emphasis on those actually expected to be present. Although not specifically developed for the Eastern Shore, using the Pennsylvania (corn and turf), Kansas (sorghum), Oregon Christmas tree (forestry), and non-agricultural scenarios described below (impervious, residential, rights-of-way, and fallow/idle land) is expected to provide reasonable high-end estimates of exposure. In addition, the Oregon Christmas tree scenario (developed for the OP cumulative assessment) was used as a surrogate for forestry use in this area. Further description and copies of the existing PRZM scenarios may be found at the following website.

http://www.epa.gov/oppefed1/models/water/przmenvironmentdisclaim.htm

One outcome of the 2003 IRED process was a modification to all existing atrazine labels that stipulated setback distances around intermittent/perennial streams and lakes/reservoirs. The label changes specify setback distances of 66 feet and 200 feet for atrazine applications surrounding intermittent/perennial streams and lakes/reservoirs, respectively. These distances were incorporated into this assessment and, the standard spray drift assumptions were modified accordingly using AgDrift to estimate the impact of a setback distance of 66 feet on the fraction of drift reaching a surface water body. The revised spray drift percentages were 0.6% for ground applications and 6.5% for aerial applications and were incorporated into PRZM/EXAMS modeling.

Models to estimate the effect of setbacks on load reduction for runoff are not currently available. It is well documented that vegetated setbacks can result in a substantial reduction in pesticide load to surface water (USDA, NRCS, 2000). Specifically for atrazine, data reported in the USDA study indicate that well vegetated setbacks have been documented to reduce atrazine loading to surface water by as little as 11% and as much as 100% of total runoff without a buffer. It is expected that the presence of a well vegetated setback between the site of application of atrazine and receiving water bodies could result in reduction in loading. Therefore, the aquatic EECs presented in this assessment are likely to over-estimate exposure in areas with well-vegetated setbacks. While the extent of load reduction can not be accurately predicted through each relevant stream reach in the action area data from USDA (USDA, 2000) suggests reductions could range from 11 to 100%.

3.2.1.1. Modeling Agricultural Uses

The non-agricultural scenarios were used within the standard framework of PRZM/EXAMS modeling using the standard graphical user interface (GUI) shell, PE4v01.pl which may be found at;

http://www.epa.gov/oppefed1/models/water/index.htm#przmexamsshell

EEC were calculated for the following exposure durations; single day, 14-day, 21-day, 30-day, 60-day, and 90-day. Durations of exposure for 14, 30 and 90 days were post-processed manually using Microsoft Excel to provide standard one in ten year return frequency exposures.

A complete discussion of the standard modeling approach including further details on PRZM/EXAMS may be found at the following website:

http://www.epa.gov/oppefed1/models/water/index.htm

3.2.1.2. Modeling Non-Agricultural Uses (Residential and Rights-of-Ways)

A modified approach for assessing residential uses that includes modeling of a pervious (¼ acre lot) and impervious surface scenario was developed. Model output was weighted based on local data on the percentage of impervious surfaces in the action area region.

The residential scenario was used in tandem with the impervious scenario. It is likely that some overspray does reach the impervious surfaces in the residential setting. In order to account for this, the impervious surface was modeled using three separate assumptions. For the purposes of risk assessment it was assumed that 1% of the application rate could reach the impervious surfaces surrounding each residential lot. This amount of overspray is not based on empirical data (no publicized studies on this occurrence were found in the open literature); however, the assumption is consistent with the standard assumption of 1% spray drift with ground applications in ecological risk assessments. It should be remembered that this scenario represents general impervious surfaces within a watershed not part of the ¼ acre lot and includes roads, parking lots, and buildings where overspray from residential lots is expected to be minimal. The ¼ acre lot, by comparison, was developed with a curve number reflective of the fact that the lot is covered with both pervious surfaces (grass and landscaped gardens) and impervious surfaces (driveways, sidewalks, and buildings). To test the assumption and address the uncertainty with the lack of data for overspray, two alternate scenarios were modeled in order to characterize the effect the 1% assumption. Modeling was completed for the impervious surface with 0% and 10% over spray to provide a lower bound and an upper bound. The results of these alternate modeling exercises are discussed more fully in Section 3.3 of this assessment.
Two additional assumptions are critical to modeling the residential use. First, the scenario assumes that the ¼ acre lot is typical for this use pattern. In order to justify the assumption of ¼ acre lot as a typical exposure scenario, publicly available data from the United States Census (Census) 2003 American Housing Survey (AHS) was reviewed on July 10, 2006 and is available at the following website.

http://www.census.gov/hhes/www/housing/ahs

Data for all suburban homes available nationally was considered. It is assumed that most pesticide applications, particularly herbicide applications, will occur in suburban settings. In order to test the assumption of the ¼ acre lot as the best representation, the AHS data for suburban homes that list total number of houses by lot size and by square footage of house (see Table 1C-3 at the AHS website above) was evaluated. With a total of 45,552,000 total units reported nationally for all suburban areas, 12,368,000 units (the largest class at 27%) were on lots between 1/8 acre and ¼ acre, while 9,339,000 units (the second largest class at 21%) were on lots between ¼ acre and ½ acre. Overall, the median lot size was 0.37 acre. This analysis suggests that the ¼ acre lot is a reasonable approximation of suburban pesticide use.

The second critical assumption is that 50% of a ¼ acre lot will be treated with atrazine. This assumption was based partially on data from the AHS website and partially from professional judgment about typical features and the percentage of a typical lot those features might require. For example, the AHS survey data reports that of a total of 43,328,000 reported single detached homes in suburban areas, 10,124,000 (the largest group at 23%) were between 1,500 and 2,000 square feet, while 7,255,000 (the third largest group at 17%) were between 2,000 and 2,500 square feet, and 9,513,000 (the second largest group at 22%) were between 1,000 and 1,500 square feet. From these data, it was assumed that a typical home is 2,000 square feet with a 1,000 square foot footprint. The lower sized houses less than 1,500 square feet are more likely to represent single floor structures; thus, the 1,000 square foot estimate for a house footprint is reasonable.

In addition to the footprint of the typical house, it was assumed that a typical house would have a driveway of approximately 25 by 30 feet or 750 square feet and roughly 250 square feet of sidewalk. A typical suburban home was also assumed to have roughly 300 square feet of deck space and 900 square feet of garage. Finally, it was assumed that a substantial portion of the typical home would be planted in landscaping with an estimate of 2,000 square feet. All of the previous estimates are based on professional judgment and are not derived from the AHS data. All of these areas are assumed to not be treated with a turf herbicide, resulting in a total area not treated with atrazine of 5,200 square feet. Taking a total ¼ acre lot size of 10,890 square feet and subtracting the untreated square footage yields a total remaining area of 5,690, or roughly 50% of the total lot that could be potentially treated.

Currently, two categories of formulations are registered for atrazine use on residential sites. These are granular and liquid formulations (wettable powder dry flowables). The formulations have been modeled separately because application rates are different (2 lbs/acre for granular and 1 lb/acre for liquid), and the standard assumption for modeling granular formulations is different from liquid formulations. Granular formulations are typically modeled as soil applied (in PRZM the application method, or CAM, must be set to 8 for soil application with a minimized incorporation depth of 1 cm) and 0% spray drift compared with a foliar application (application method (CAM) set to 2 for foliar application with a 4-cm depth of incorporation) and standard spray drift assumption of 1% for ground applications.

For the residential scenarios, it was assumed a percentage of the watershed was represented by the ¼ acre lot and some percentage was represented by impervious surfaces. To account for this effect in the modeling of the residential scenario, the relative contribution of the impervious and residential scenarios for different portions of the region surrounding the Chesapeake Bay was evaluated. Land cover data (http://www.chesapeakebay.net/data/index.cfm) suggests that in the northern Bay, near Baltimore, Maryland, impervious surface area near the Bay can approach

70% of the total area. Alternatively, on the Eastern Shore, the percentage of impervious surface rarely exceeds 30% and is generally less than 10%. In the southern Bay, near the Tidewater region of Virginia, the percentage of impervious is roughly 50% of total. Figure 3-1 presents the analysis of impervious coverage in the area surrounding the Chesapeake Bay relative to available atrazine use data. For this screening level assessment, it was assumed that 30% of the watershed is impervious.

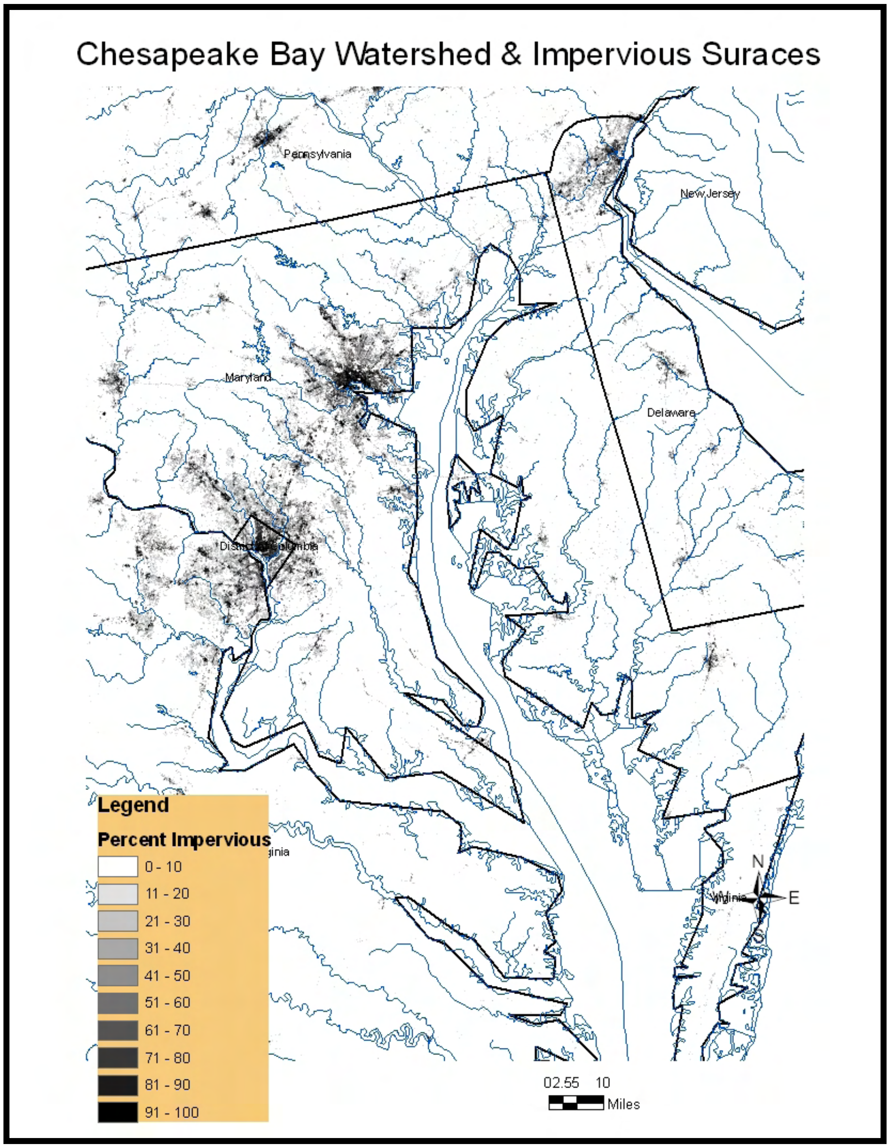


Figure 3-1. Percentage of Impervious Surfaces in the Chesapeake Bay and its Immediate Tributaries

For the rights-of-way scenario, it was assumed that rights-of-way consist of 50% impervious and 50% pervious cover. In addition, it was assumed that no single watershed will be completely covered by a rights-of-way use. This assumption seems reasonable given that rights-of-way (roads, rail and utility lines) are typically long linear features that traverse a watershed. For the screening level assessment, it was assumed that no more than 10% of the watershed is covered in rights-of-way.

An analysis was completed for the Chesapeake Bay endangered species assessment for atrazine to assess the amount of rights-of-way likely to be present in the action area. In this analysis, spatial data specific to the Chesapeake Bay watershed compiled by the Chesapeake Bay Program (CBP) for roads and railways and obtained on July 12, 2006 that may be found at;

(http://www.chesapeakebay.net/data/data_desc.cfm?DB=CBP_GIS)

and internal Agency data for pipelines was obtained (spatial data for utility easements was unavailable). The road, rail, and pipeline land cover data were added to a GIS map of the action area (Figure 3-2) and a comparison of the density of the total network of potential use sites was made. Each land cover feature in the GIS map is presented as a line with no width associated. EFED applied a buffer using the Arc Toolbox within Arc Map in order to account for the potential width of the each linear feature. This assignment of area to each feature was done in order to compare the total area of each feature type (e.g. railways) with the total area of the action area.

For each feature, an assumption was made about the typical width of the feature (e.g. width of the road surface plus shoulders) plus the right of way area adjacent to the feature that could potentially be treated. In each case, a conservative assumption for the width of the feature zone plus the potentially treated area surrounding each was made. These assumed widths were based on professional judgment but skewed the total feature estimate to the largest feature in the class. For example, it was assumed that a national highway would yield that largest width which was then applied to all primary and secondary highways within the action area. This approach is also assumed to be conservative because it is unlikely that all features within the available data will be treated with atrazine because many of the areas are likely to be maintained using mechanical methods (e.g. mowing) or not treated at all. Using the CBP data different road types were able to be distinguished using the USGS' classification scheme for digital line graph (DLG) data. In these data set distinguishing between highways (primary and secondary roads), state and county highways, state and local streets, unimproved trails, as well as minor features such as interchanges and traffic circles was possible. Different buffer widths were applied to each category of roadway depending upon the general use of the feature. For example, it is generally assumed that primary and secondary roads are wider than state/county roads, which in turn are generally wider than local streets. Based on this approach the following assumptions were made for the width of each feature.

- Primary/Secondary Roads – 200 feet
- Class 3 Roads (State/County) – 100 feet
- Class 4 Roads (Streets in built up areas) – 50 feet
- Unimproved Trails – 25 feet
- Rail – 200 feet
- Pipeline – 100 feet
- Utility Line – 200 feet

Given these assumptions the percentage of rights-of-way land cover types plus associated buffers for roads, railways, and pipelines within the action area for the Chesapeake Bay is 0.2 % of the total area for rail, 3.6 % for all primary and secondary roads (interstates, national and state highways), and 0.5% for pipelines. Including all classes of roads in the data set yields a road density of 9.5%. This is believed to be an over estimation because it includes a high number of

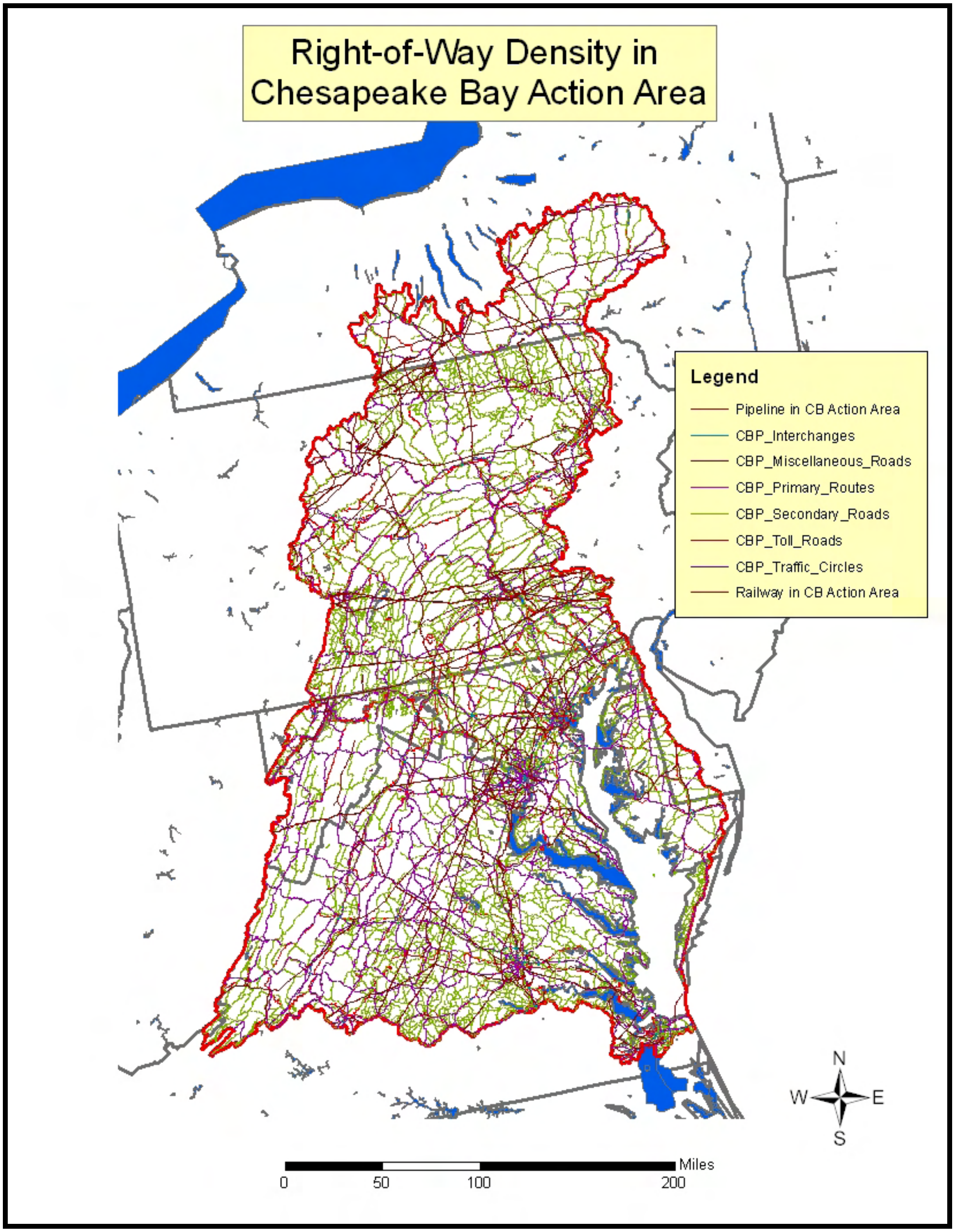


Figure 3-2 Density of Road, Railways, and Pipelines as Surrogate for Right of Way Density in Chesapeake Bay Watershed (Action Area)

roads in urban and suburban areas unlikely to be treated with pesticides for right of way control. Locally, it appears that higher percentages occur near more urbanized areas, however, it was assumed that less right of way application of pesticides occur in urbanized areas. Additional roads may be present in the action area not captured by the available spatial data and the analysis above does not include utilities for which no spatial data are available. Therefore, the10 % assumption of rights-of-way in the action area used in this assessment is an over-estimation but given the uncertainties is reasonable while still being conservative and protective. The impact of this assumption on overall EECs is addressed in Section 3.3.

Use of atrazine on commercial forestry operations cannot be precluded as a potential use; however, the available information suggests that atrazine is rarely used on commercial forestry operations in the Chesapeake Bay watershed (Powers, 2006; VA DOF, 2004; Muir, 2006; USDA, 2004; Wagner *et al.*, 2004; Pannill, 2006). However, forestry is a predominant land cover in some areas of the Chesapeake Bay watershed; therefore, this potential use has been addressed using the Oregon Christmas tree scenario. This scenario was developed specifically for the OP cumulative assessment recently completed by the Agency (USEPA, 2006b) and represents a vulnerable site based on OP use information intended to represent a commercial nursery operation. Information on the OP cumulative and scenarios used in modeling may be found at:

http://www.epa.gov/pesticides/cumulative/2006-op/index.htm

The Oregon Christmas tree scenario is expected to approximate commercial forestry operations where herbicides are typically applied during the seedling emergence and juvenile growth stages to prevent competition with newly planted trees. The scenario was not modified to represent local conditions but was modeled using local weather data. Several factors suggest that modeling of forestry uses of atrazine are likely to result in an over-estimation of exposure. As previously mentioned, atrazine use in forestry operations in the Chesapeake Bay watershed is considered to be rare. Secondly, modeled estimates represent a one in ten year return frequency using 30 years of modeled output; however, if atrazine were used at all, it would likely be applied for only one or two years during early growth stages. Taken together, the best available data suggest that the modeled exposures for atrazine forestry use are likely to over-estimate exposure. In addition, the highest EECs were from the sorghum scenario and these were used for risk estimation while the EECs for forestry are discussed qualitatively in the risk description.

3.2.2. Model Inputs

3.2.2.1. Label Application Rates and Intervals

Labels may be categorized into two types: labels for technical products and labels for formulated, or end use, products. Technical products contain atrazine of high purity and are used only to make formulated products. Formulated products can be applied in specific areas to control weeds while technical products are not used directly in the environment but rather to make formulated products. The formulated product labels limit atrazine's potential use to only those sites where the labels specify.

In the January and October IREDs, EPA stipulated numerous changes to the use of atrazine including label restrictions and other mitigation measures designed to reduce risk to human health and the environment. Specifically pertinent to this assessment, the Agency entered into a Memorandum of Agreement (MOA) with the atrazine registrants. In the MOA, the Agency stipulated certain label changes must be implemented on all atrazine labels prior to the 2005 growing season including cancellation of some uses, reduction in application rates, and requirements for harmonization across labels including setbacks from waterways. Specifically, the label changes stipulate no use of atrazine within 50 feet of sinkholes, within 66 feet of intermittent and perennial streams, and within 200 feet of lakes and reservoirs. It is expected that a setback distance will result in a reduction in loading due to runoff across the setback zone; however, current models are not capable of estimating these reductions quantitatively. A qualitative discussion of the potential impact of these setbacks on estimated environmental concentrations of atrazine for the assessed species is discussed further in Section 3.2. Table 3.2 provides a summary of label application rates for atrazine uses evaluated in this assessment.

Although currently registered uses of atrazine are numerous, only residential uses, turf, corn, sorghum, fallow/idle land, and rights-of-ways were modeled because these uses are expected to predominate in the Chesapeake Bay watershed. Atrazine use in forestry was also modeled; however, for reasons previously described, atrazine use on forestry is expected to be minimal.

Atrazine is formulated as liquid, wettable powder, dry flowable, and granular formulations. Application equipment for the agricultural uses includes ground application (the most common application method), aerial application, band treatment, incorporated treatment, various sprayers (low-volume, hand held, directed), and spreaders for granular applications. Risks from ground boom and aerial applications are quantified in this assessment because they are expected to result in the highest off-target levels of atrazine due to generally higher spray drift levels. Due to the high mobility of atrazine, runoff associated with large rain events is expected to be responsible for the greatest off-target movement of atrazine. Smaller runoff events resulting from over irrigation would result in lower levels of off-target movement.

Table 3.2 Label Application Information for the Chesapeake Bay Endangered Species Assessment						
Scenario	**Maximum Application Rate (lbs/acre)**	**Maximum Number of Applications**	**Date of First Application**	**Formulation**	**Method of Application**	**Interval Between Applications**
Residential	2.0	2	April 1	Granular	Ground	30 days
Residential	1.0	2	April 1	Liquid	Ground	30 days
Right-of-Way	1.0	1	June 1	Liquid	Ground	NA
Fallow/idle land	2.25	1	November 1	Liquid	Ground	NA
Turf	2.0	2	April 1	Granular	Ground	30 days
Turf	1.0	2	April 1	Liquid	Ground	30 days
Corn	2	1[1]	April 1	Liquid	Ground and Aerial	NA
Sorghum	2	1[1]	April 1	Liquid	Ground and Aerial	NA
Forestry	4.0	1	June 1	Liquid	Ground and Aerial	NA
Fallow/idle land	2.25	1	November 1	Liquid	Ground and Aerial	NA

[1] Actual labeled maximum rates are 2.0 lb/acre for a single application with no more than 2.5 lbs/acre per year. The rate and number of applications reported in this table are an approximation of the label maximum given the current limitation in PRZM/EXAMS graphical user interface PE4v01.pl. Currently, PE4v01.pl allows multiple applications but the rate cannot be varied from one application to the next.

3.2.2.2. Typical Use Rates and Application Intervals

Application rates, number of applications, and application intervals were estimated at the state level for Maryland, Pennsylvania, and Virginia (BEAD: Kaul et al, 2006a, Kaul et al, 2006b, Zinn et al, 2006). The information from BEAD was developed from a combination of USDA-NASS[6], and data obtained from Doane (www.doane.com; the full dataset is not provided due to its proprietary nature). Data from both sources were averaged together over the years 2000 to 2004 to calculate average annual usage statistics by state and crop for atrazine, including pounds of active ingredient applied, percent of crop treated, number of applications per acre, application rate per acre, and base acres treated. Application rates are provided at the state level for only crops grown in the immediate vicinity of the Chesapeake Bay including corn, fallow/idle land (as a surrogate for rangeland), and sorghum on which atrazine is registered. No other labeled agricultural uses (sugarcane, guava, and macadamia nuts) are present in the action area.

For atrazine use aggregated for the three states identified above, typical application rates for corn and sorghum range from 1.0 to 1.2 lbs/acre, while typical rates on fallow/idle land range from 1.0 to 2.0 lbs/acre. Typically the 90th percentile of reported application rates is used as an upper bound on actual use (U.S. EPA, 2000); however, no data on the 90th percentile is available. Typical application rates and number of intervals should be evaluated with caution because these values represent an average , which implies that atrazine is actually applied at rates higher than those reported as typical a significant percentage of the time.

BEAD also provided additional estimates on the typical number of applications for atrazine in the Chesapeake Bay area. This information indicates that the typical number of applications for corn and sorghum is roughly half of the label maximum, while the typical rate on rangeland is approximately equivalent to the labeled directions. Typical application rates used to characterize exposure estimates are in Table 3.3.

To refine the risk assessment for the atrazine endangered species assessment, the minimum and typical application intervals when more than one application is made per year on a site were evaluated. Intervals were estimated by first determining the registered herbicide/site combinations within the Chesapeake Bay watershed. The sites chosen were those with the average number of applications greater than one (BEAD: Kaul, Grube, and Kiely, 2005). If the average number of applications equals one, it was assumed that only one application is made, and, therefore, that the typical interval is not needed. Only sites with greater than one application of a pesticide are discussed below.

For corn, most growers apply atrazine only once per season. However, approximately 12 percent of growers apply atrazine more than once, following a pre-emergence application with a post-emergence application (Assessment of Potential Mitigation Measures for Atrazine, 2003). According to atrazine label information for corn, the minimum application interval is either 14 days or not specified on the label (BEAD: Kaul and Carter, 2005). BEAD contacted experts in

[6] United States Depart of Agriculture (USDA), National Agricultural Statistics Service (NASS) Chemical Use Reports provide summary pesticide usage statistics for select agricultural use sites by chemical, crop and state. See http://www.usda.gov/nass/pubs/estindx1.htm#agchem.

the States and this interval was described as the absolute minimum interval. Usually application intervals would be longer.

In Maryland, atrazine intervals are likely to be at least 14 days at a minimum. Whether atrazine is applied as an early pre-plant application followed by an at-plant application, or atrazine is applied post-emergence after a pre-emergence treatment, the recommended interval is 14 days or more (Glenn, 2006). For Virginia, the BEAD estimate for the number of atrazine applications on corn may be high, with the number approximately 1.1. A typical interval for a post-emergence treatment after an earlier treatment is approximately 35 days (ranging between 30 to 40 days).

In Pennsylvania, atrazine is usually applied either pre-emergence or post-emergence. For those situations with both applications of atrazine, a pre-emergence application would be followed by a lower amount of atrazine as a post-emergence application, with the interval likely to be at least 21 days. Typically, the interval is 28 days.

For sorghum, atrazine may be applied at various timings. “Atrazine is effective at many application timings including: winter weed control, and pre-plant for control of weeds prior to planting through post-plant as long as weeds are no more than one and one-half inches and sorghum is six to 12 inches tall” (Assessment of Potential Mitigation Measures for Atrazine, 2003). According to atrazine label information, for sorghum, the minimum application interval is either 21 days or not specified on the label (BEAD: Kaul and Carter, 2005). The interval between applications is likely to be similar to that for corn for Maryland (Glenn, 2006). In Virginia, atrazine is only applied to sorghum once (Hagood, 2006).

For fallow use, according the Aatrex® 4L label and some other atrazine labels, only one application of atrazine may be made in fallow period (CDMS search). In addition, the Atrazine Interim Reregistration Decision (IRED) states that only one application per year may be made for chemical fallow applications (U.S. EPA, 2003a).

Typical application rates and number of intervals must be evaluated with caution in that these represent an average that implies that a percentage of the time atrazine is actually being applied at rates higher than those reported as typical. Table 3.3 summarizes the typical application rates and number of applications relative to those used in this assessment.

Table 3.3. Comparison of Modeled Application Rates and Number of Applications with Typical Use Data Used in the Triazine Cumulative Risk Assessment				
Scenario	**Maximum Application Rate (lbs/acre)**	**Maximum Number of Applications**	**Typical Application Rate[1] (lbs/acre)**	**Typical Number of Applications[1]**
Corn	2.0	1	1.0 – 1.2	1 – 1.3
Sorghum	2.0	1	1.0 – 1.2	1.0 – 1.6
Rangeland[2]	1.0	1	1.0 - 2.0	0.6 – 0.7[3]

[1] Reported as range of values from states of Delaware, Maryland, Pennsylvania, and Virginia as prepared for the triazine cumulative risk assessment
[2] Rangeland compared with reported use rates on Fallow; rangeland is no longer a labeled use (U.S. EPA, 2006c)
[3] Rates reported as less than 1.0 considered equivalent to 1.0

The appropriate PRZM input parameters were selected from the environmental fate data submitted by the registrant and in accordance with model parameter selection guidelines (Guidance for Selecting Input Parameters in Modeling the Environmental Fate and Transport of Pesticides, Version 2.3, February 28, 2002). These parameters are consistent with those used in both the 2003 IRED and cumulative triazine risk assessment. The date of first application was developed reviewing several sources of information including data provided by BEAD, Crop Profiles maintained by the USDA, and conversations with local experts. More detail on the crop profiles and the previous assessments may be found at:

http://pestdata.ncsu.edu/cropprofiles/cropprofiles.cfm

http://www.epa.gov/oppsrrd1/REDs/atrazine_ired.pdf

http://www.epa.gov/pesticides/cumulative/common_mech_groups.htm#chloro

A summary of the model inputs used in this assessment are provided in Table 3.4.

Table 3.4. Summary of PRZM/EZAMS Environmental Fate Data Used For Aquatic Exposure Inputs For Atrazine Endangered Species Assessment for the Chesapeake Bay		
Fate Property	**Value**	**MRID (or source)**
Molecular Weight	215.7 g/mol	Product Chemistry
Henry's constant	2.58 x10^{-9} atm-m^3/mol @ 20^o C	Product Chemistry
Vapor Pressure	3 x 10^{-7} mm Hg @ 20^o C	Product Chemistry
Solubility in Water	33 mg/l	Product Chemistry
Photolysis in Water	335 days	MRID 42089904
Aerobic Soil Metabolism Half-lives	152 days	MRID 40431301 MRID 40629303 MRID 42089906
Hydrolysis	stable	MRID 40431319
Aerobic Aquatic Metabolism (water column)	304 days	2x aerobic soil metabolism rate constant
Anaerobic Aquatic Metabolism (benthic)	608 days	MRID 40431323
Koc	88.78 ml/g	MRID 40431324 MRID 41257901 MRID 41257902 MRID 41257904 MRID 41257905 MRID 41257906
Application Efficiency	95 percent for aerial 99 percent for ground	default value
Spray Drift Fraction	6.5 % for aerial 0.6 % for ground	AgDrift value based on setback distance of 66 ft

3.2.3. Model Results

Model estimated surface water concentrations are summarized in Table 3.5. These EECs represent the screening level EEC which are used in risk estimation. Discussion and additional characterization of these EEC is in Sections 3.3.

Table 3.5. Summary of PRZM/EXAMS Output for all Scenarios Modeled for the Atrazine Endangered Species Assessment for the Chesapeake Bay Watershed using the Standard Water Body									
				90th Percentile					
Use Site	Application Rate (lbs/acre)	Number of Applications (interval)	First Application Date	Peak EEC (µg/L)	14-day EEC (µg/L)	21-day EEC (µg/L)	30-day EEC (µg/L)	60-day EEC (µg/L)	90-day EEC (µg/L)
Residential – Granular[1]	2.0	2 (30 days)	April 1	11.5	11.4	11.3	11.3	11.1	10.8
Residential – Liquid[1]	1.0	2 (30 days)	April 1	7.7	7.6	7.6	7.5	7.3	7.2
Right-of-Way [1]	1	1	June 1	2.9	2.9	2.8	2.8	2.8	2.7
Corn	2	1[2]	April 1	47.2	46.9	46.8	46.5	45.6	44.4
Sorghum	2	1[2]	April 1	55.4	54.8	54.6	54.3	53.7	52.5
Fallow/idle land	2.25	1	November 1	45.1	45.0	45.0	45.0	45.0	44.9
Forestry[3]	4.0	1	June 1	50.5	49.7	49.7	49.2	47.5	45.9
Turf – Granular	2.0	2 (30 days)	April 1	9.9	9.9	9.9	9.9	9.8	9.8
Turf - Liquid	1.0	2 (30 days)	April 1	5.3	5.2	5.2	5.1	5.0	4.9

[1] Assumes 1% overspray of atrazine to the impervious surfaces. Alternate assumptions of 0% and 10% overspray to impervious surfaces are tested in Section 3.3.
[2] Actual labeled maximum rates are 2.0 lb/acre for a single application with no more than 2.5 lbs/acre per year. The rate and number of applications reported in this table are an approximation of the label maximum given the current limitation in PRZM/EXAMS graphical user interface PE4v01.pl. Currently, PE4v01.pl allows multiple applications but the rate cannot be varied from one application to the next.
3 – Forestry EEC not recommended for risk estimation due to uncertainty in actual use pattern and overestimation of application frequency

3.3. Additional Modeling Exercises Used to Characterize Potential Exposures

3.3.1. Residential Uses

As noted previously, it was assumed that that a reasonable exposure scenario for the residential use would be that 30% of the residential scenario is impervious surface. To evaluate this assumption, and to get a perspective of how atrazine EECs might vary throughout the Chesapeake Bay Watershed, the impact of alternate percentages of impervious surface coverage on overall EECs were evaluated. Table 3.6 below presents the results of this analysis. The analysis indicates that the overall EEC decreases as the percentage of impervious surface increases. This is likely due to the overall increase in runoff volume which dilutes the total pesticide mass loading.

Table 3.6. Summary of the Impact of Variations in Pervious to Impervious Ratio on Predicted Exposures from the PRZM/EXAMS Residential Scenario (granular) with 1% Overspray[1]

Use Site	Percent Impervious	Application Rate (lbs/acre)	Number of Applications (interval)	90th Percentile					
				Peak EEC (µg/L)	14-day EEC (µg/L)	21-day EEC (µg/L)	30-day EEC (µg/L)	60-day EEC (µg/L)	90-day EEC (µg/L)
Residential in Eastern Shore	30%	2.0	2 (30 days)	11.5	11.4	11.3	11.3	11.1	10.8
Residential in Northern Bay	70%	2.0	2 (30 days)	9.0	8.9	8.9	8.9	8.8	8.7
Residential in Southern Bay	50%	2.0	2 (30 days)	10.3	10.2	10.2	10.2	10.0	9.9

[1]In this case overspray represents an amount of granules landing on impervious (non-target) surfaces

To evaluate the assumption of 1% overspray, alternatives varying percentage of overspray that could occur on the impervious surface were modeled. For the residential and rights-of-way scenarios, it was assumed 1% overspray onto the impervious scenario. An alternative modeling exercise was conducted to evaluate the significance of overspray. To account for potential overspray, the impervious scenario (assuming 30% of watershed is impervious and 50% of the ¼ acre lot is treated as above) with a percentage of the application rate as being applied to the non-target surface was modeled. It was assumed that no more than 10% of the intended application rate would end up on the impervious surface. Given that the impervious scenario is intended to represent non-target surfaces such as roads, parking lots and buildings, it seems reasonable to assume that 10% overspray is an over-estimation of what would likely occur. To model the overspray, the binding coefficient was set to zero and the aerobic soil metabolism half life set to stable in lieu of actual data. Thus, it is assumed that non-binding would occur on these surfaces and that limited degradation would occur. The total application rate was then multiplied by the percentage overspray. For the residential scenario this yielded an application rate on the impervious surface of 0.2 lbs/acre. In addition, the same analysis using an assumption of 0% over spray was modeled.

Comparison of the resulting EEC indicates that with 10% overspray the overall EECs for the residential use pattern are increased by nearly a factor of three, while assuming 0% overspray only slightly decreases the EEC compared to 1% overspray. This is not unexpected given the increased runoff, lack of binding, and lack of degradation being assumed. Without actual data for these processes, it is impossible to determine whether these exposures reflect reality; although, it is expected that these assumptions are likely to be conservative (some binding and degradation could occur). The analysis does suggest that overspray onto impervious surfaces can, and possibly is a significant issue when the percentage of overspray is high. The comparison is presented in Table 3.7.

Table 3.7. Comparison of Residential Scenario with an Assumption of No Overspray on Impervious Surface to the Alternate Assumptions of 10% and 1% Overstay to Impervious Surfaces[1]

90^{th} Percentile of 30 Years of Output

Use Site	Application Rate (lbs/acre)	Number of Applications (interval)	First Application Date	Peak EEC (µg/L)	14-day EEC (µg/L)	21-day EEC (µg/L)	30-day EEC (µg/L)	60-day EEC (µg/L)	90-day EEC (µg/L)
Residential with 1% Overspray to Impervious	2.0	2 (30 days)	April 1	11.5	11.4	11.3	11.3	11.1	10.8
Residential with no Overspray to Impervious	2.0	2 (30 days)	April 1	9.3	9.2	9.2	9.1	8.8	8.6
Residential with 10% Overspray to Impervious	2.0	2 (30 days)	April 1	34.5	34.3	34.2	34.1	33.9	33.6

[1] assumes 30% impervious surface in watershed

For this assessment it was assumed that 50% of the ¼ acre lot is treated. To test the significance of this assumption, the exposure scenario was re-run using different assumptions of 75% and 10% treatment of the ¼ acre lot. Modeling an increasing percentage of the ¼ acre lot that is treated to 75% of the total area increases the EECs by roughly 50%, while decreasing the percentage treated to 10% of the total area decreases the EECs by a factor of three. The results of both analyses are presented in Table 3.8.

Table 3.8. Comparison of Residential Scenario with an Assumption of 50% of the ¼ acre lot treated to the Residential Scenario (granular) with an Assumption of 75% and 10% of the ¼ acre lot treated

90th Percentile of 30 Years of Output

Use Site	Application Rate (lbs/acre)	Number of Applications (interval)	First Application Date	Peak EEC (µg/L)	14-day EEC (µg/L)	21-day EEC (µg/L)	30-day EEC (µg/L)	60-day EEC (µg/L)	90-day EEC (µg/L)
Residential with 50% of lot treated	2.0	2 (30 days)	April 1	11.5	11.4	11.3	11.3	11.1	10.8
Residential with 75% of lot treated	2.0	2 (30 days)	April 1	16.1	16.0	15.9	15.8	15.5	15.1
Residential with 10% of lot treated	2.0	2 (30 days)	April 1	4.0	3.9	3.9	3.9	3.9	3.8

In the initial screening level assessment, it is assumed that the ratio of pervious to impervious surface (70/30) accounts for the difference in exposure and runoff. This ratio is best characterized as a conservative estimate of runoff potential. In fact, it is expected that the differential will be more highly skewed for runoff from the impervious scenario than is reflected in the ratio.

To test this differential and the potential effect it has on runoff and ultimately exposure, both scenarios were modeled using local weather data. The analysis indicates that, when run with the Wilmington, Delaware weather station data, the impervious surface scenario yields greater than five times more runoff than does the ¼ acre lot scenario. In areas where impervious cover approaches 100% of the total, the impervious scenario best represents the amount of runoff. Alternatively, in more rural areas where impervious cover is less than 5% of the total, the pervious scenario bests represent the runoff amount. The screening level EECs presented in Table 3.5 through Table 3.8 were generated by weighting the EECs based on the percentage of impervious and not the differential in runoff potential. To test the impact of the differential in runoff potential, an analysis of weighting the runoff yields from the two scenarios relative to the percentage of cover in the Northern Bay, Eastern Shore, and Southern Bay was conducted. The analysis indicates that where impervious cover is present, the straight weighting based on percentage of impervious under-estimates potential runoff and loading. This suggests that for the no-overspray scenario above, the effect is to over-estimate exposure. Alternatively, if the assumptions of runoff differential were applied to the overspray evaluation, the EEC would increase. Table 3.9 presents a summary of the impact of these varying impervious percentages on total runoff amounts.

Table 3.9. Percentage of Runoff Resulting from Impervious Surfaces within Different Parts of the Chesapeake Bay Watershed[1]

Location in Chesapeake Bay Watershed	Max % Impervious	Max % Pervious	Residential Runoff from PRZM (cm)	Residential Runoff Weighted by % Pervious (cm)	% Residential of Total	Impervious Runoff from PRZM (cm)	Residential Runoff Weighted by % Impervious (cm)	% of Impervious of Total	Total Weighted Runoff (cm)
North Bay	0.7	0.3	325.6	97.68	0.06	2126	1488.2	0.94	1585.88
Eastern Shore	0.3	0.7	325.6	227.92	0.26	2126	637.8	0.74	865.72
South Bay	0.5	0.5	325.6	162.8	0.13	2126	1063	0.87	1225.8

[1]Runoff reported as centimeter represents the depth of water running off of the entire watershed. For example, a runoff depth of 1 cm from a 10 hectare watershed equals 1,000,000,000 cm^3, or 1,000,000 liters

The impact of this effect on the residential scenario was evaluated by adjusting the assumed percentage of pervious to impervious from the 70 to 30 ratio used in the screening level assessment (Table 3.5.) and replacing it with the ratios in Table 3.9 for different portions of the Bay using the same assumptions of 50% treated lot and 1% overspray. The results of this analysis are presented in Table 3.10 and indicate that the estimates above are likely over-predicting exposure when no overspray is modeled.

Table 3.10. Summary of the Impact of Runoff Differential on the Predicted Exposures from the PRZM/EXAMS Residential Scenario (granular) Using the Analysis from Table 3.9[1]									
Use Site	Ratio of Pervious to Impervious	Application Rate (lbs/acre)	Number of Applications (interval)	90th Percentile					
				Peak EEC (µg/L)	14-day EEC (µg/L)	21-day EEC (µg/L)	30-day EEC (µg/L)	60-day EEC (µg/L)	90-day EEC (µg/L)
Residential	70/30	2.0	2 (30 days)	11.5	11.4	11.3	11.3	11.1	10.8
Residential in Northern Bay	06/94	2.0	2 (30 days)	8.0	7.9	7.9	7.8	7.8	7.7
Residential in Eastern Shore	26/74	2.0	2 (30 days)	8.9	8.8	8.8	8.8	8.7	8.6
Residential in Southern Bay	13/87	2.0	2 (30 days)	8.3	8.2	8.2	8.1	8.0	8.0

[1] Assuming 1% overspray to the impervious scenario

3.3.1.1. Summary of Non-Agricultural EECs

The above analysis suggests that EECs in the standard water body scenarios are likely in the low µg/L range (<10 µg/L). However, if substantial overspray occurs, resulting in higher atrazine levels on impervious surfaces, then EECs could be higher. However, all EECs for the residential uses were lower than those estimated for agricultural uses.

3.3.2. Additional Characterization of Agricultural Use EECs

3.3.2.1. Modeling Using the Standard Water Body and Typical Use Rates Within the Chesapeake Bay Watershed

In addition to the analysis of modeling data discussed above, alternative modeling of the corn and sorghum scenarios were conducted using the typical application rates information because these were the two scenarios yielding the highest exposures in the assessment. The rates and number of applications are similar for both uses with a typical application rate of 1 lb/acre with number of applications between 1 and 1.5 (1.5 applications represent an average of multiple applications applied at lower than maximum rates). To simplify this part of the assessment the refined application rate was modeled at 1 lb/acre with one application. Comparison of typical applications rates (essentially equivalent to the average of all available reported data) with monitoring data and modeling with labeled maximum rates should only be used for characterization because by its nature a typical, or average, rate implies that roughly 50% of the applications are occurring above this value. Given the site-specific nature of an endangered species assessment it is impossible to rule out that some percentage of actual applications are occurring in proximity to listed species. However, overall the results of this analysis are not unexpected and indicate that use at the typical application rates results in a reduction on EEC across the board by a factor of two. The results of this analysis are summarized in Table 3.11.

Table 3.11. Summary of PRZM/EXAMS Output for all Scenarios Modeled for the Atrazine Endangered Species Assessment for the Chesapeake Bay Watershed Using a Standard Water Body

Use Site	Application Rate (lbs/acre)	Number of Applications (interval)	First Application Date	Peak EEC (µg/L)	90th Percentile 14-day EEC (µg/L)	21-day EEC (µg/L)	30-day EEC (µg/L)	60-day EEC (µg/L)	90-day EEC (µg/L)
Corn	2	1	April 1	47.2	46.9	46.8	46.5	45.6	44.4
Corn	1	1	April 1	23.6	23.5	23.4	23.3	22.8	22.2
Sorghum	2	1	April 1	55.4	54.8	54.6	54.3	53.7	52.5
Sorghum	1	1	April 1	27.6	27.4	27.3	27.1	26.9	26.2

3.3.2.2. Characterization of Potential Exposures in Flowing Waters

The standard assessment for aquatic organisms relies on estimates of exposure derived from PRZM/EXAMS using the standard water body. The standard water body is a 1 hectare water body that is 2 meters deep with a total volume of 20,000,000 liters and is modeled without flow. The standard water body was developed to provide an approximation of high-end exposures expected in water bodies, lakes, and perennial/intermittent streams adjacent to treated agricultural fields. Typically, this has been interpreted as a stream with little or low flow. For non-persistent pesticides, the standard water body provides a reasonably high-end estimate of exposure in headwater streams and other low flow water bodies for both acute and longer-term exposures. For more persistent compounds, the non-flowing nature of the standard water body still provides a reasonable high-end estimate of peak exposure for many streams found in agricultural areas; however, it appears to over-estimate exposure for longer-term time periods in all but the most static water bodies.

The hydrologic landscape of the Chesapeake Bay watershed was characterized by generalizing the stream network into six general classifications (Section 2.4). In the case of the modeled concentrations (presented in Table 3.5) that were derived with a non-flowing standard water body, it is expected that the peak exposures are generally representative of the headwater streams in areas of low topography such as the Eastern Shore and parts of the Coastal Plain province, as well as minor inlets surrounding the Bay. It is also expected that the standard water body scenarios is over-estimating exposure in water bodies with flowing water, including the lower and main channels of the Bay tributaries, the mouths of the tributaries, and the Bay itself.

To characterize the potential impact of flowing water on the longer-term exposures (14-day, 21-day, 30-day, 60-day, 90-day, and annual average), additional modeling and analysis of available monitoring data was conducted. Alternate approaches to modeling with the standard water body were conducted to provide a general sense of the relative reduction in long term exposure which might be occurring in water bodies where flow is higher than small headwater streams in low topographic regions (interior of the Eastern Shore).

The sorghum scenario was selected and modeled for the Chesapeake Bay watershed using the same input parameters presented in Table 3.4 with assumptions for flow (described below). This scenario provided the highest EEC of any scenario modeled. In fact, usage data from BEAD suggests (Kaul et al, 2005) that relatively little sorghum is grown in the watershed when compared to corn. However, given that sorghum is present in the watershed and the fact that the predicted EEC for corn was similar the selection is reasonable and representative of both corn and sorghum.

The standard EXAMS standard water body for ecological risk assessment was used as the receiving body for runoff from a 10 hectare field and is a static water body. The standard water body is intended to represent a water body or an ecologically sensitive stream adjacent to an agricultural field. Typically, this is conceptualized as a headwater stream, however; there are examples of higher order streams with very low flow rates (e.g. small tidal inlets, oxbow lakes only occasionally fed by stream flow, etc.). In order to test the effect of flow on these predicted concentrations, the standard water body was modeled as above but allowed the model to route runoff water from the 10 hectare field through the 1 hectare water body. The results of all the modeling are presented in Table 3.12.

Further analysis was conducted by pairing PRZM output from the sorghum scenario with the variable volume water model (VVWM) that was developed for the Probabilistic Risk Assessment (PRA) process. The VVWM was developed based on the recommendation of the Scientific Advisory Panel (SAP) to account for the influence of input and output (flow) on model predictions. The VVWM was used to evaluate the impact of varying volume on the overall EECs. In general, the VVWM yielded EECs below the EXAMS water body, but still above the annual averages from the available monitoring data (see discussion below). Two alternate model runs with the VVWM were conducted. The first was done using standard assumptions and environmental fate parameters generally consistent with the non-flowing standard water body discussed previously. The assumptions in this model run included a 2 meter depth water body which can drop to 0.02 meter and rise to 3 meters before flow occurs. The second assumption was designed to represent a larger volume water body that maximizes flow into the water body. This was accomplished by increasing the overall maximum depth of the water body to 10 meters. The net effect of this change is to reduce the original estimates with the VVWM by roughly 50%. The results are summarized in Table 3.12. Documentation and rationale for the assumptions used in the VVWM may be found at:

http://www.epa.gov/scipoly/sap/2004/index.htm#march

To further characterize the impact of larger water bodies with flow, the sorghum scenario was run using the Index Reservoir as the receiving water body. The Index Reservoir represents a 5.3 hectare water body draining a 172 hectare watershed and is used for drinking water assessments for human health risk assessment. In the case of the Index Reservoir, the standard approach is to take the total runoff from the 172 hectare watershed calculated by PRZM through the Index Reservoir in EXAMS and route that volume of water as flow through the reservoir while assuming no change in reservoir volume. The predicted EECs and flow rates from these alternate approaches that assume flow are slightly below the original non-flowing EEC and are summarized in Table 3.12. More information on the Index Reservoir may be found at

http://www.epa.gov/oppfead1/trac/science/reservoir.pdf

The modeled output relative to actual flowing streams was evaluated to provide context to these estimates. The USGS collected flow rates from 734 streams, creeks and rivers from across Virginia representing the range of physiographic provinces in Virginia that are typical of stream types found in the Chesapeake Bay watershed. The flow data was subsequently separated into regions representing the Eastern Shore, Coastal Plain, Piedmont, and Mountain regions of Virginia. The modeled flow rates from PRZM were then compared with the regional dataset of flow developed by the USGS for the Virginia Department of Environmental Quality (VADEQ) and obtained on May 30, 2006 which can be found at:

http://va.water.usgs.gov/vadeq_data/number_scroll.htm

As shown in Table 3.12., the 7Q10 (7 day average with a return frequency of 10 years that is indicative of base-flow values) and Q50 (50th percentile of reported values) values indicate that flow varies dramatically from the low topography Eastern Shore to the mountainous regions of western Virginia. Neither flow estimate is a perfect representation of flow conditions as modeled, but is intended to provide a range of possible flow rates. These flow values range by a factor of two orders of magnitude across the state. Comparison with the modeled flow rates suggests that the PRZM modeling is yielding significantly lower flow rates than the Virginia data particularly when comparing the Q50 data.

To test the influence of these flow data on modeled EECs, a final analysis with the Index Reservoir was conducted that consisted of modifying the GUI (PE4v01.pl) for running PRZM/EXAMS. The modification consisted of altering the stream flow (STFLO) parameter in PRZM responsible for reporting flow through the receiving water body by using the VADEQ data as opposed to a runoff volume as described previously. Three alternate Index Reservoir scenarios were modeled using the 7Q10 flow rate for the Eastern Shore, the 7Q10 flow rate for the Coastal Plain, and the Q50 value for the Coastal Plain (no Q50 value was reported for the Eastern Shore). This was intended to provide a bracket on possible flow rates and modeled EECs within the regions most representative of the tributaries (streams, creeks and rivers) in the immediate vicinity of the Chesapeake Bay (Eastern Shore and Coastal Plain). The results of this analysis are presented in Table 3.12 and indicate that using 7Q10 values yield EECs comparable to the standard water body modeling, while using the Q50 values yield long-term EEC appreciably below those predicted using the static water body. This is consistent with the expectation that modeling is likely to be conservative relative to actual long-term average concentrations in flowing water.

Table 3.12. Comparison of Alternative PRZM Modeling with EEC Generated Using a Static Water Body

Scenario	Flow (ft^3/sec)	Peak EEC (µg/L)	96 hour EEC (µg/L)	21 day EEC (µg/L)	60 day EEC (µg/L)	90 day EEC (µg/L)	Yearly EEC (µg/L)
CB sorghum with static water body[1]	0	55.4	55.6	54.9	53.7	52.9	42.9
CB sorghum with flow thru standard water body	0.022	30.2	29.9	28.6	26.7	25.3	16.5
CB sorghum with VVWM with 3 meter depth	0.023	22.4	22.1	21.7	21.3	21.3	13.9
CB sorghum with VVWM with 10 meter depth	0.020	14.2	14.2	14.2	13.9	13.9	10.9
CB sorghum with Index Reservoir[2]	0.380	49.9	49.1	46.6	41.1	36.8	17.9
CB sorghum (IR) with 7Q10 flow from VA Eastern Shore	0.138	53.0	52.5	50.8	47.1	44.6	30.2
CB sorghum (IR) with 7Q10 flow from VA Coastal Plain	1.730	44.2	41.6	35.3	23.1	17.3	4.6
CB sorghum (IR) with Q50 flow from VA Coastal Plain	105.3	32.7	7.3	1.7	0.6	0.4	0.1
Flow Data From VADEQ Data[3]	**Q50**	**7Q10**					
Eastern Shore	No Data	0.138					
Coastal Plain	105.3	1.730					

[1]EEC generated using PE4v01.pl in this table are slightly different from those presented in Table 3.5 due to different duration of exposure and slight differences in the manual estimation technique used in Table 3.5.
[2] Sorghum IR scenario EEC reported using percent cropped area (PCA) of 87%
[3] VADEQ flow data reported as 7Q10 values

3.3.3. Specific Characterization for Headwater Streams at Locations of the Dwarf Wedge-mussel

One final piece of characterization of the PRZM/EXAMS modeling was performed relative to the headwater streams of known locations of the dwarf wedgemussel. The dwarf wedgemussel is unique with respect to habitat compared with the other assessed organisms in that it is immobile and resides in headwater and low-order streams with low to moderate flow. Therefore, the dwarf wedgemussel is located for most of its life cycle in types of waters estimated to have the highest atrazine concentrations using PRZM/EXAMS. However, because the standard water body likely overestimates long-term concentrations in flowing waters, potential effects of flow on atrazine concentrations were evaluated for known locations of the dwarf wedgemussel to allow for characterization of longer-term EECs in the types of waters listed for this mussel. Flow rates of streams known to be inhabited by the dwarf wedgemussel are in Table 3.13. Data were available for only three of the known locations of the dwarf wedgemussel. Therefore, the stream flow information for the three locations with data were used as surrogates for all locations for the purpose of estimating flow conditions where the dwarf wedgemussel is located.

Table 3.13. Estimated Flow Rate for Water Bodies Known to be Inhabited by the Dwarf Wedgemussel.

Location of Dwarf Wedge Mussel	Estimated Flow Rate April to June (cfs)[a]	Basis for Conclusion
Aquia Creek Stafford county, VA	23	USGS data for Aquia Creek
South Anna River Louisa county, VA	250	USGS data for South Anna River
Po River Spotsylvania county, VA	40	USGS data for Po River

[a] Flow rates represent median daily rate over the time frame of available data (30 to 70 years of data). Data obtained from http://waterdata.usgs.gov/nwis/rt

As a test against the exposure estimates provided in Section 3.2, the daily information from the USGS flow rates from Aquia Creek were compared. The list of water body's names and reaches within which the mussel resides was obtained. Available flow data from the USGS for Aquia Creek near Garrisonville, Virginia (station 01660400) and South Anna River near Ashland, Virginia (station 01672500) were obtained, which are two of the water bodies where the mussel is found within the Chesapeake Bay watershed.

Flow data was provided as a mean daily value for each day of the year based on measurements recorded between 1972 and 2004 for Aquia Creek and between 1931 and 2004 for South Anna River (http://waterdata.usgs.gov/nwis/rt). The data are reported for each day as percentiles at the 5^{th}%, 10^{th} %, 20^{th}%, 25^{th}%, 50^{th}%, 75^{th}%, 80^{th}%, 90^{th}%, and 95^{th}% of all recorded data. These data was compared with estimated flow rates from the modeling discussed above and estimated an appropriate flow rate for additional modeling specific to the dwarf wedgemussel. The average yearly and four quarter seasonal flow rates for January to March, April to June, July to September, and October to December were calculated. In general, the highest flow rates were found to be the first quarterly rates (January to March), followed by the second quarterly rates (April to June). The lowest rates were found during the summer months represented by the third quarter. In general, the flow rates for the Aquia Creek site are between the 7Q10 and Q50 values presented in Table 3.14 for the coastal plain, while the South Anna River flow rates are higher than those previously modeled for coastal plain. This exercise is intended to provide specific characterization as to the representative nature of the EECs modeled using the static water body relative to the flowing water system where the mussels are known to be located.

Annual average rates at the 50^{th} percentile of all years of flow for both data sets were selected for additional flow modeling. It is believed that this is a reasonable value to represent an annual average and is slightly more conservative (defined by lower flow) than the second quarterly value that represents the window when most atrazine is expected to be applied as a pre-emergent herbicide. Table 3.14 summarizes the data for Aquia Creek, and Table 3.15 summarizes the flow data for South Anna River.

Table 3.14. Aquia Creek Flow Information (ft^3/sec)[2]

	5th %	10th %	20th %	25th %	50th %	75th %	80th %	90th %	95th %
Annual average of daily mean values	4.09	6.43	9.48	11.01	18.89[1]	34.89	41.93	81.08	195.29
Average of first quarter seasonal daily mean values (January to March)	7.95	13.41	18.42	20.71	33.35	56.47	66.80	120.95	252.46
Average of second quarter seasonal daily mean values (April to June)	6.24	8.52	12.61	14.67	23.14	38.29	45.56	86.00	202.01
Average of third quarter seasonal daily mean values (July to September)	0.29	0.54	1.10	1.44	4.40	13.00	16.71	43.61	129.54
Average of fourth quarter seasonal daily mean values (October to December)	2.00	3.34	5.93	7.36	14.86	32.06	38.96	74.26	198.48

[1]Highlighted column represents the flow data used in the refined modeling for characterization

[2] Data source: http://waterdata.usgs.gov/nwis/rt

Table 3.15. South Anna River Flow Information (ft^3/sec) [2]

	5th %	10th %	20th %	25th %	50th %	75th %	80th %	90th %	95th %
Annual average of daily mean values	72.09	90.51	123.40	137.91	215.74[1]	372.66	442.44	791.56	1411.49
Average of first quarter seasonal daily mean values (January to March)	130.04	165.34	230.68	256.53	385.11	650.10	773.46	1311.54	1981.67
Average of second quarter seasonal daily mean values (April to June)	100.30	118.55	150.40	163.32	245.52	400.82	463.23	782.62	1324.74
Average of third quarter seasonal daily mean values (July to September)	20.54	26.73	38.41	45.01	81.47	153.63	186.91	394.70	1076.04
Average of fourth quarter seasonal daily mean values (October to December)	39.05	52.53	75.59	88.36	153.02	289.39	349.97	682.93	1274.96

[1]Highlighted column represents the flow data used in the refined modeling for characterization
[2]Data source: http://waterdata.usgs.gov/nwis/rt

Modeling with the median (50^{th} percentile) of the annual average flow rates from both sites yields considerably lower EECs, particularly longer-term EECs, than those predicted using the static water body.

As evidenced by the summary in Table 3.14., the daily flow within Aquia creek varies significantly from season to season. Previous analysis suggests that flow through the receiving water body will have a dramatic impact on the longer-term averages of exposure. Analysis shows that using the Index Reservoir as a receiving water body and modifying the flow rate to represent different conditions drops the long-term averages below levels of concern (See Section 5). This analysis is intended to characterize the influence of flow on EECs. A closer look at the daily flow data indicates that periods of much lower flow are possible. At the lower percentiles, there are periods of time within Aquia Creek when flow rates are consistently below 2 cfs for extended periods of time. For example, at the 5^{th} percentile, daily flow rates drop below 2 cfs on June 17 and do not increase above 2 cfs until November 22. By way of comparison, the daily flow rates at the 50^{th} percentile only drop below 2 cfs between September 3 and September 11.

This analysis indicates that it is possible that there can be long periods of low flow within Aquia Creek (and other waters inhabited by the dwarf wedgemussel). However, this is believed to be an unlikely occurrence. First, although it is expected that low flow conditions will generally occur during the summer and early autumn in this area, it is unlikely that the duration will be as long as suggested above. This is supported by the fact that the minimum flow year (the year from the daily distribution with the lowest flow) varies from day to day, suggesting that continuous low flow conditions in any year are unlikely. Second, even if the low flow conditions described above did occur continuously in a given year, the return frequency at the 5^{th} percentile would be 1 in 20 years, suggesting a relatively unlikely occurrence. Third, low flow conditions generally represent conditions when runoff is unlikely to occur, and analysis suggests runoff is the dominant mechanism of atrazine exposure. Finally, the time period for low flow conditions do not coincide with the main time of application of atrazine. The dominant use of atrazine is on corn and sorghum as a pre-plant and pre-emergent application and is generally applied in the early- to mid-spring.

In general, EECs predicted using the Index Reservoir and the flow rate from the Aquia Creek data (18.9 ft^3/s) are roughly two times lower for peak exposures and one to two orders of magnitude lower (depending on duration) for the longer term averages when compared to the EECs generated using the static water body. Using the flow rate (215.7 ft^3/s) for South Anna River yields an EEC roughly two times lower for the peak and two to three orders of magnitude lower (depending on duration) than the longer-term averages than the static water body. The results of this alternate modeling compared to the original EECs predicted using the static water body are presented in Table 3.16.

Scenario	Flow (ft^3/sec)	Peak	96 hour	21 day	60 day	90 day	Yearly
CB sorghum with static water body[1]	0	55.4	55.6	54.9	53.7	52.9	42.9
CB sorghum IR with Aquia Creek Flow Data	18.9	33.1	21.6	7.5	2.7	1.8	0.5
CB sorghum IR with South Anna River Flow Data	215.7	32.6	6.1	1.3	0.45	0.30	0.07

Table 3.16. Summary of Alternative PRZM Modeling Using the Index Reservoir and Site Specific Flow data from Aquia Creek and South Anna River Compared with PRZM EEC Modeling Using a Static Water Body

[1]EEC generated using PE4v01.pl in this table are slightly different from those presented in Table 3.5. due to different duration of exposure and slight differences in the manual estimation technique used in Table 3.5.

This analysis suggests that, in streams with flowing water, the predicted EECs using the static water body are over-estimating exposure for longer duration periods. The modeled values using flow rates from two of the streams that this species inhabit suggests that the peak exposure EEC is roughly two times less than that for the static water body and that the longer term exposures are several orders of magnitude below the static water body EECs. Also, a single sample was analyzed for atrazine in Aquia Creek once (station id = 01660350) on August 23, 1994 with a detection of 0.006 µg/L (Figure 3-2) as part of the Chesapeake Bay monitoring program (discussed in Section 3.4.). The results of the alternative modeling coupled with the fact that the only atrazine detection within any of the water bodies where the dwarf wedgemussel resides is well below both the peak and longer-term EECs predicted both from the static water body and the alternative flowing water body suggests that the EECs presented above are more representative of the types of exposures to which the dwarf wedgemussel is likely to encounter.

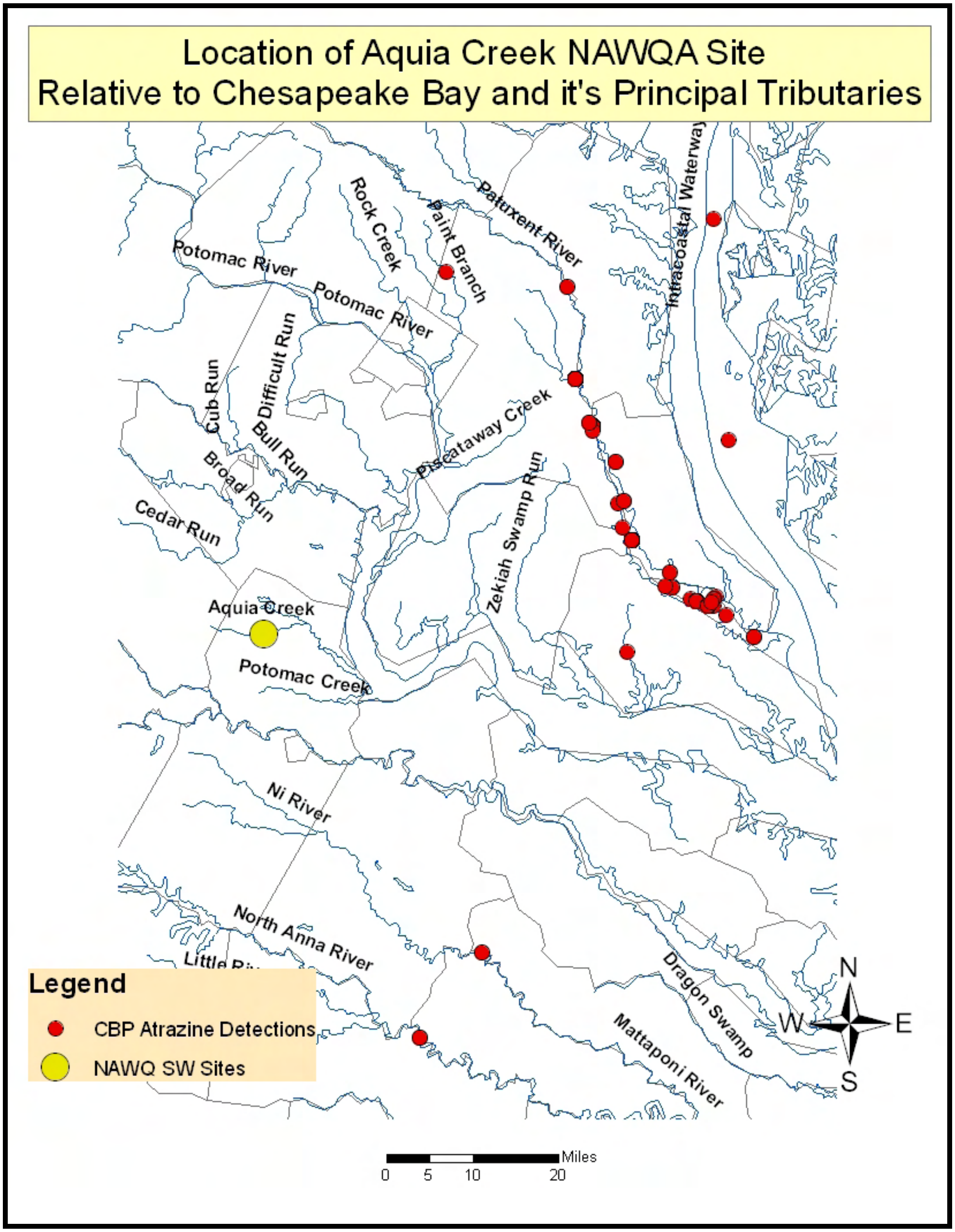


Figure 3-3. Location of NAWQA Surface Water Sites on Aquia Creek Relative to Chesapeake Bay Program (CBP) Surface Water Sites.

(Source: http://water.usgs.gov/nawqa and http://www.chesapeakebay.net/data/index.cfm, respectively)

3.3.4. Summary of Alternative Modeling Exercises

The modeling data suggest that peak EECs estimated using the standard water body remained relatively consistent across the modeled scenarios, and that peak atrazine concentrations calculated using the standard water body are high-end approximations. However, alternative modeling exercises, which incorporate flow rates representative of locations of the assessed species, suggests that longer-term atrazine EECs (days to weeks) in flowing water bodies are likely overestimated by the standard water body, and the inclusion of flow on estimated atrazine concentrations increases with increasing duration of exposure.

3.4. Monitoring Data

Unlike many pesticides, atrazine has a fairly robust data set of surface water monitoring from a variety of sources. Included in this assessment are atrazine data from the USGS National Water Quality Assessment Program (NAWQA) (http://water.usgs.gov/nawqa/), Chesapeake Bay Program (http://www.chesapeakebay.net/data/index.cfm), and Heidelberg College (http://wql-data.heidelberg.edu/). In each case, the data was characterized in terms of general statistics (number of samples, frequency of detection, maximum concentration, and mean from all detections). In addition, several sample sites were evaluated from each data set for more detailed analysis including calculation of annual maximum and annual time weighted mean concentrations by site by year. For all data described below, each site characterized by maximum and annual average (or time weighted mean) concentration represent sites with multiple samples all of which were evaluated and include in selecting maximum and annual average concentrations. The sample sites chosen for this additional analysis were selected by choosing those locations from the national and local data with the highest detected concentrations of atrazine. Finally, an interpolation of single year's worth of data from one sample site in the Heidelberg College data to estimate 14-day, 30-day, 60-day, and 90-day averages was conducted.

NAWQA groundwater data was evaluated to determine the importance of groundwater on potential loadings to the Chesapeake Bay. Groundwater data from Maryland, Pennsylvania, and Virginia was downloaded from the USGS NAWQA data warehouse (http://water.usgs.gov/nawqa/) on May 11, 2006. The three states were selected because of proximity to the Chesapeake Bay. Delaware was excluded from this analysis because most of the state lies outside the boundaries of the watershed.

A total of 725 well samples were analyzed for atrazine in groundwater between 1993 and 2004. Of these samples, a total of 373 had positive detections of atrazine with 25 of those estimated at below the limit of quantitation (LOQ). The frequency of detection for all detections was 51%. The maximum concentration detected was an estimated value of 4.2 µg/L (above the LOQ) in an urban setting in Virginia Beach, Virginia while the highest non-estimated value was 2.2 µg/L in an urban setting in Lebanon, Pennsylvania. Of all detections, only 8 samples had detections greater than 1.0 µg/L. Overall, the data suggest that atrazine recharge to the waters of the Chesapeake Bay watershed are possible but that the detection frequency, travel times, and magnitude of exposures are such that they are likely to be dwarfed by the surface runoff route.

3.4.1. National USGS NAWQA Data

An analysis of the entire USGS NAWQA data set for atrazine was conducted. A data download from the USGS data warehouse (http://water.usgs.gov/nawqa) on May 11, 2006 provided the data set used in this analysis. Overall, a total of 20,812 samples were analyzed for atrazine and of these, 16,742 had positive detections (including concentrations estimated above or below the limit of quantitation) yielding a frequency of detection of roughly 80%. The maximum detection from all samples was 201 µg/L from the Bogue Chitto Creek in Alabama in 1999. Overall, the average concentration detected was 0.26 µg/L when considering only detections and 0.21 µg/L when considering all detections and non-detections (using the detection limit as the value for estimation).

Using the top ten sites with the highest atrazine concentration more refined analysis of the detections was conducted. All values from the national data set were ranked and the top ten sites were selected based on maximum concentration. Each location was analyzed separately by year and the annual maximum and annual time weighted mean concentrations were calculated. The minimum criterion for calculating time weighted means for each sampling station was at least 4 samples in a single year. The equation used for calculating the time weighted annual mean is as follows:

$$[((T_{0+1}-T_0) + ((T_{0+2}-T_{0+1})/2))*C\ t_{0+1})] + (((T_{i+1}-T_{i-1})/2)*C_i) + [((T_{end}-T_{end-1}) + ((T_{end-1}-T_{end-2})/2)*C_{Tend-1})]/365$$

where: Ci = Concentration of pesticide at sampling time (Ti)
Ti = Julian time of sample with concentration Ci
T_0 = Julian time at start of year = 0
T_{end} = Julian time at end of year = 365

Generally, the maximum values from this analysis are similar to, or above (by as much as two to three times) the model predictions from PRZM/EXAMS from the Chesapeake Bay watershed, while the annual time weighted mean (TWM) concentrations are roughly an order of magnitude below the static water body model predictions for annual average and are roughly two to three times below the flow influenced model predictions described above.

The modeling and national NAWQA monitoring data are not directly comparable because the monitoring data are generally from high atrazine use areas in the Midwest and South vulnerable to runoff while the modeling was conducted exclusively for the action area of the Chesapeake Bay. In the Chesapeake Bay watershed the atrazine use intensity is similar to the areas in the Midwest and South but the runoff vulnerability is lower as identified by Williams et al (2004). The Ecological Exposure in Flowing Water Bodies (Williams, et al, 2004) utilized the WARP model to identify highly vulnerable watersheds for sampling and determined that the top 20% watersheds were predominantly in the Midwest and South while the watersheds in the immediate vicinity of the Chesapeake Bay are between the 40$^{\%}$ and 50%.

Given the fact that the watersheds surrounding the Chesapeake Bay are less vulnerable to atrazine runoff a comparison with monitoring data from more vulnerable areas was conducted to provide context to the modeled exposures. Modeled concentrations that exceed monitoring data in highly runoff vulnerable atrazine use areas would suggest that the modeling is either overly conservative or the monitoring is not representative. Conversely, modeled concentrations that are less than the monitoring data from the highly runoff vulnerable atrazine use area suggest that modeling is not conservative. In the case of atrazine, the modeling tends to under predict the highest single day concentrations and over predicts the annual average concentration from the national NAWQA data. This is not unexpected given that the majority of the high atrazine detections are from the 1990s. Also, because runoff vulnerability is much lower in the area surrounding the Chesapeake Bay. The analysis suggests that modeling in the action area for atrazine is providing a reasonable estimate of short term exposure but is over-estimating longer term exposure.

No comparison has been made between these data and model predictions for the intermediate durations exposures (14-day, 30-day, etc.) because the NAWQA data generally do not have the frequency needed to conduct a meaningful interpolation between data points. Table 3.17 presents a summary of the annual time weighted mean concentrations, and Table 3.18 presents a summary of the annual maximum concentrations.

Table 3.17. Annualized Time Weighted Mean (TWM) Concentration (µg/L) for the Top Ten NAWQA Surface Water Sites Ranked by Maximum Concentration Detected

Year	Station Name (ID)								
	Bogue Chitto Creek, near Memphis, TN (02444490)	Tributary to S Fork Dry Creek, near Schuyler, NE (06799750)	Sugar Creek, New Palestine, IN (394340085524601)	Kessinger Ditch, near Monroe City, IN (03360895)	LaMoine River @ Colmar, IL (05584500)	Sugar Creek @ Milford, IL (05525500)	Tensas River @ Tendal, LA (07369500)	Maple Creek near Nickerson, NE (06800000)	Auglaize River near Ft Jennings, OH (04186500)
1991									
1992			0.98					1.32	
1993			0.77	3.80				1.43	
1994			0.87	2.56					
1995			2.28	0.74					
1996			1.30				4.32		2.18
1997			5.36		3.45		5.55	1.03	2.82
1998			0.82		1.79		2.94	1.21	1.88
1999	9.62		0.28				2.50	0.68	
2000	6.49		0.56			1.26		0.15	
2001	1.20		0.83			0.78		0.22	1.28
2002	2.88		0.51			2.22		1.26	0.80
2003	2.14	4.46	0.70			7.83		2.23	1.42
2004	1.77	68.78[1]	0.67			1.24		3.31	1.93

[1] TWM concentration likely biased due to fact that first sample on May 8 is the peak sample from this year, and interpolation method likely resulted in an inflated time weighted mean value.

Table 3.18. Maximum Concentration (µg/L) for the Top Ten NAWQA Surface Water Sites Ranked by Maximum Concentration Detected

	Station Name (ID)									
Year	Bogue Chitto Creek, near Memphis, TN (02444490)	Tributary to S Fork Dry Creek, near Schuyler, NE (06799750)	Sugar Creek, New Palestine, IN (394340085524601)	Kessinger Ditch, near Monroe City, IN (03360895)	LaMoine River @ Colmar, IL (05584500)	Sugar Creek @ Milford, IL (05525500)	Tensas River @ Tendal, LA (07369500)	Maple Creek near Nickerson, NE (06800000)	Auglaize River near Ft Jennings, OH (04186500)
1991									
1992			14					25	
1993			8.5	120				11.2	
1994			11	24					
1995			27	2.6					
1996			14.2				30		18
1997			129		108		92.3	10.3	85.2
1998			7.88		27.7		19.3	30	9.96
1999	201		2.39				13.9	10.7	
2000	136		3.84			23		0.87	
2001	4.5		14.4			6.96		1.21	10.4
2002	24.8		4.01			21.3		16.4	2.58
2003	18.8	21.3	10.5			108		34.8	13.4
2004	14.6	191	28.3			10.9		91.9	18.7

3.4.2. USGS Watershed Regression of Pesticides (WARP) Data

The NAWQA data were then compared against the percentiles used to develop the USGS WARP (provided by Charlie Crawford of USGS on June 2, 2006 via email). Comparison against WARP percentiles was conducted because the WARP model has been to be a valuable tool for site selection and assessing overall vulnerability. More information on the WARP model may be found at:

http://pubs.usgs.gov/wri/wri034047/wrir034047.pdf

The WARP data were developed using a subset of the national data described above (all WARP data are included in the national data analysis described above). The USGS National Stream Water Quality Accounting Network (NASQAN) data was also included in the WARP dataset; however, it is not included in this assessment as it represents major rivers. Data collected between 1992 and 1999 from a total of 113 sample sites were used to create the model. Sample sites were selected based on the robustness of the data available at a given site. The model yields predicted daily exposures at various percentiles of occurrence. National NAWQA data and the model predictions against the mean and 95th percentile values from the data used were compared. The maximum 95th percentile value from the WARP data was 20.2 µg/L compared to a maximum of 201 µg/L from all data. The maximum mean value used in the WARP model development data was 3.82 µg/L which is consistent with the annual TWM values discussed above.

3.4.3. Regional USGS NAWQA Data

The PRZM/EXAMS EECs were compared to data from surface water sites specific to the Chesapeake Bay watershed (defined by the Lower Susquehanna River Study Unit and the Potomac River Study Unit). The same technique as applied to the national data (maximum and TWM) was applied to these two study units to provide a more regionally specific snapshot of the available NAWQA data. Generally, these data are well below the national data for maximum exposures with a peak concentration of 25 µg/L compared to 201 µg/L nationwide, while the average concentration from all data are comparable with an average for all detections of 0.28 µg/L and an average for all data (detects and non-detects) of 0.27 µg/L. The results of the refined analysis indicate that, as expected, the overall exposures in the Chesapeake Bay watershed, typified by the peak and annual TWM are generally less than those seen in the national and WARP data. A summary of the results for the Potomac River Study Unit is presented in Table 3.19, and the Lower Susquehanna River Study Unit results are provided in Table 3.20.

Table 3.19. Annual Time Weighted Mean and Maximum Concentration from the Top Three USGS NAWQA Surface Water Sites Located in the Potomac Study Unit

	Station Name (ID)					
	MUDDY CREEK AT MOUNT CLINTON, VA (01621050)		MONOCACY RIVER AT BRIDGEPORT, MD (01639000)		MORGAN CREEK NEAR KENNEDYVILLE, MD (01493500)	
Year	**TWM**	**Max**	**TWM**	**Max**	**TWM**	**Max**
1993	0.31	18.60				
1994	0.13	0.16	0.38	6.90		
1995	0.11	0.14	0.97	8.00		
1996	0.24	1.66	1.97	4.30		
1997	0.26	2.14				
1998	0.18	1.96				
1999	1.59	25.00				
2000	0.22	1.55				
2001	0.28	2.87				
2002	0.44	2.73			0.36	4.08
2003	0.16	0.74			0.46	6.53
2004					0.89	7.95

Table 3.20. Annual Time Weighted Mean and Maximum Concentration from the Top Three USGS NAWQA Surface Water Sites Located in the Lower Susquehanna River Study Unit						
	Station Name (ID)					
	EAST MAHANTANGO CREEK AT KLINGERSTOWN, PA (01555400)		SUSQUEHANNA RIVER AT HARRISBURG, PA (01570500)		MILL CREEK AT ESHELMAN MILL ROAD NEAR LYNDON, PA (01576540)	
Year	**TWM**	**Max**	**TWM**	**Max**	**TWM**	**Max**
1993	0.16	0.90			0.13	0.78
1994	0.31	3.20			0.21	1.50
1995			0.04	0.81		
1996						
1997	0.12	0.39				
1998	0.33	3.37				
1999	0.14	0.58				
2000	0.37	3.33				

3.4.4. Chesapeake Bay Program Data

A similar analysis of the limited monitoring data obtained on May 3, 2006 from the Chesapeake Bay Program (CBP) office website was also conducted. The data may be found at the following website.

http://www.chesapeakebay.net/data/index.htm

The data consists of 686 samples analyzed for atrazine between the years of 1978 (one sample) through 1999 with the bulk of the samples collected in the 1990s. A total of 74 stations were present within the data set and all data were analyzed as part of this assessment. An analysis of the distribution of the site locations relative to the characterization of the watershed (described in Section 2.4) was completed and is summarized in **Table 3.21**. The analysis confirms that the bulk of the monitoring data, including the highest detections described below, were collected from streams and rivers interior of the bay proper. The analysis also confirms that the general trend is decreasing concentrations as water moves from headwater streams adjacent to treated fields to the open waters of the bay where concentrations were well below 1 ug/l.

Table 3.21 Summary of Chesapeake Bay Program (CBP) Sample Sites Relative to Watershed Characterization Described in Section 2.4					
Watershed Region	**CBP Sample Locations**	**Number of Samples**	**Number of Detections**	**Average Concentration (ppb)**	**Maximum Concentration (ppb)**
Open Bay	L0002598, L0002599, L0002600, L0002601, L0002602	4	4	0.04	0.05
Bay Inlet	L0001829, L0002632, L0002638	20	20	0.18	0.43
Estuarine Mouth of Rivers	L0001822, L0001824, L0001825, L0001826, L0001827, L0001828, L0002624, L0002229, L0002630, L0002631	72	71	0.04	0.09
Main Stem River	L0001084, L0001085, L0001817, L0001818, L0001819, L0001820, L0001821, L0001823, L0002598, L0002625, L0002626, L0002627	367	364	0.10	3.06
River Tributary	L0001083, L0002463, L0002470, L0002473, L0002475, L0002476, L0002479, L0002478, L0002481, L0002510, L0002628, L0002633, L0002634, L0002635, L0002636	105	70	0.13	1.00
Headwater Streams	L0002450, L0002451, L0002452, L0002453, L0002454, L0002455, L00002456, L00002457, L0002458, L0002459, L0002460, L0002461, L0002462, L0002463, L0002464, L0002465, L0002466, L0002467, L0002468, L0002469, L0002471, L0002474, L0002477, L0002478, L0002487, L0002488, L0002489, L0002511, L0002637	98	46	0.74	30.00

The peak concentration detected in any sample was 30 µg/L (station L0002488), which appears to be located in a tributary on the Eastern Shore. In fact, the general trend in these data are that the main detections of atrazine are in the tributaries while significantly lower concentrations have been found in the Bay itself. This pattern suggests that an emphasis on predicting exposures in the tributaries is the most conservative approach for assessing both direct and indirect effects to the named species. Five of the top sample locations were selected based on the highest detected concentration of atrazine, to calculate annual TWM and maximum concentration. Of these five sites, only two were deemed to have a sufficient number of samples (minimum needed is four annually) to analyze for TWM. The data, which are summarized below, indicate that for the location with the peak concentration (30 µg/L), the annual TWM concentration is similar to other monitoring data analyzed previously. The results are summarized in Table 3.22.

Analysis of the data indicate that the maximum atrazine level found was 30 µg/L, while the 99th, 95th, 90th, 75th, and 50th percentile values were 2.5 µg/L, 0.5 µg/L, 0.28 µg/L, 0.1 µg/L, and 0.05 µg/L respectively. A summary of the total data are presented in Figure 3-4 while the general locations of the sampling stations are presented in Figure 3-5.

Table 3.22. Annual Time Weighted Mean and Annual Maximum Concentrations from Selected Sample Locations from the Surface Water Monitoring Data from the Chesapeake Bay Program

	Station ID			
	L0001818		L0002488	
Year	**TWM**	**Max**	**TWM**	**Max**
1991			2.03	30.00
1992			0.01	0.06
1993				
1994				
1995	1.93	3.06		
1996	0.25	1.29		

Overall, there is only limited monitoring data from the Chesapeake Bay itself. The data from the Chesapeake Bay Program indicate that of the 686 samples analyzed for atrazine, only a handful are actually from the Bay, with most of the samples collected from tributaries and rivers feeding into the Bay. Of the samples in the main stem of the Bay, most of the detections for atrazine are well below 1 µg/L. The location of the maximum atrazine detection of 30 µg/L is located in a headwater stream near the edge of the Chesapeake Bay watershed (Figure 3-6).

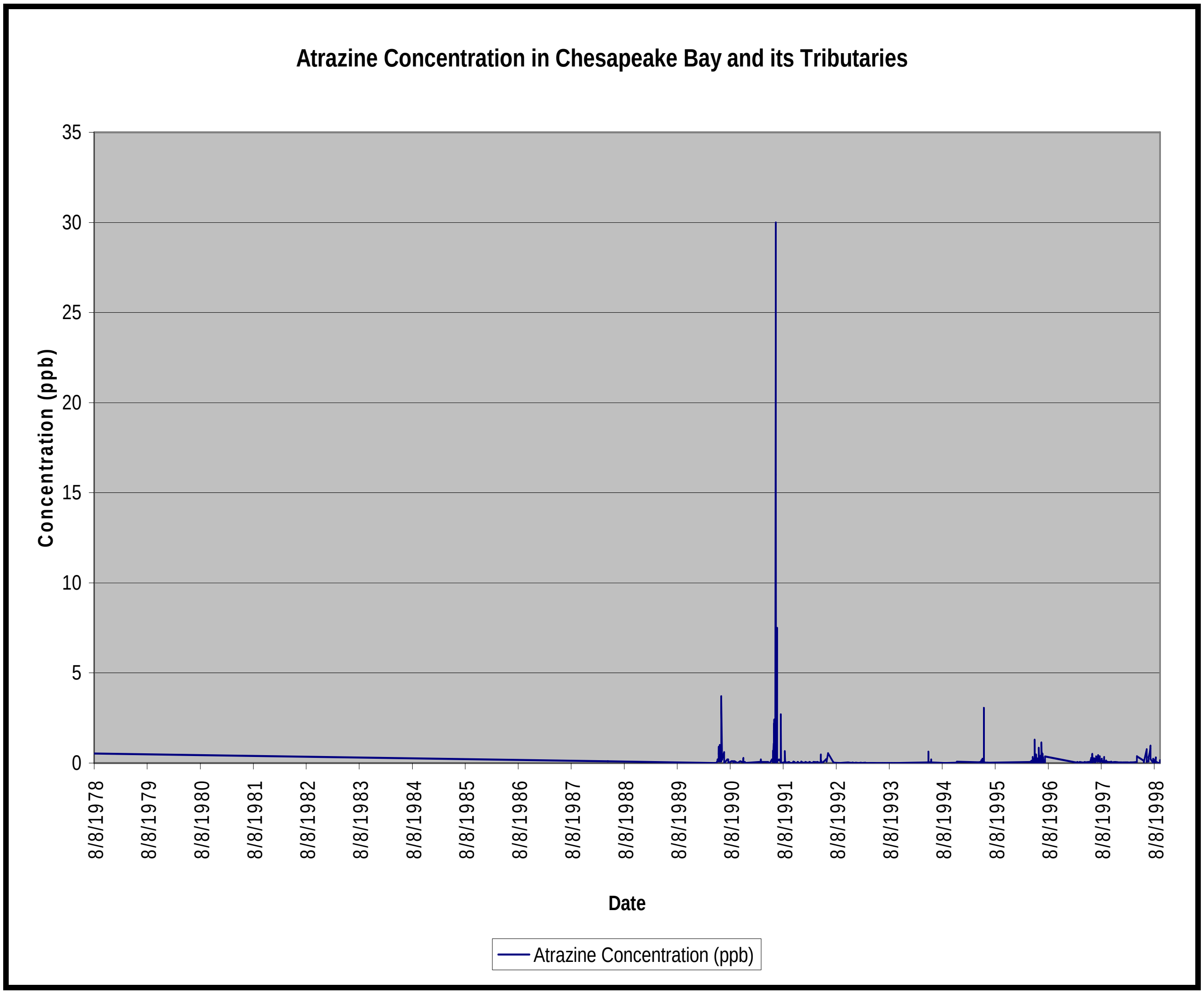


Figure 3-4. Range of Atrazine Concentrations Detected in the Chesapeake Bay and its Immediate Tributaries.

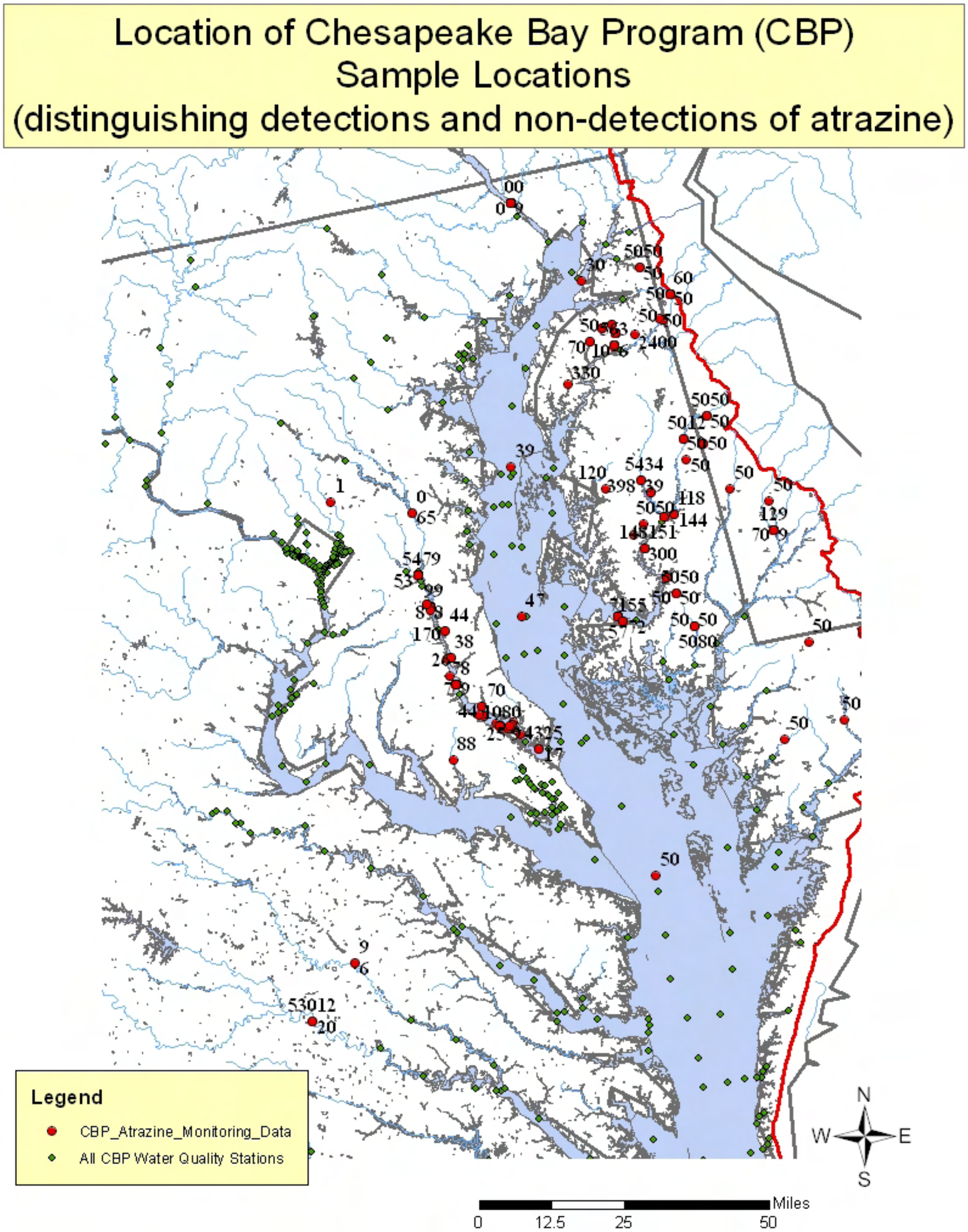


Figure 3-5. Location of Surface Water Monitoring Sites in the Chesapeake Bay and Its Immediate Tributaries.

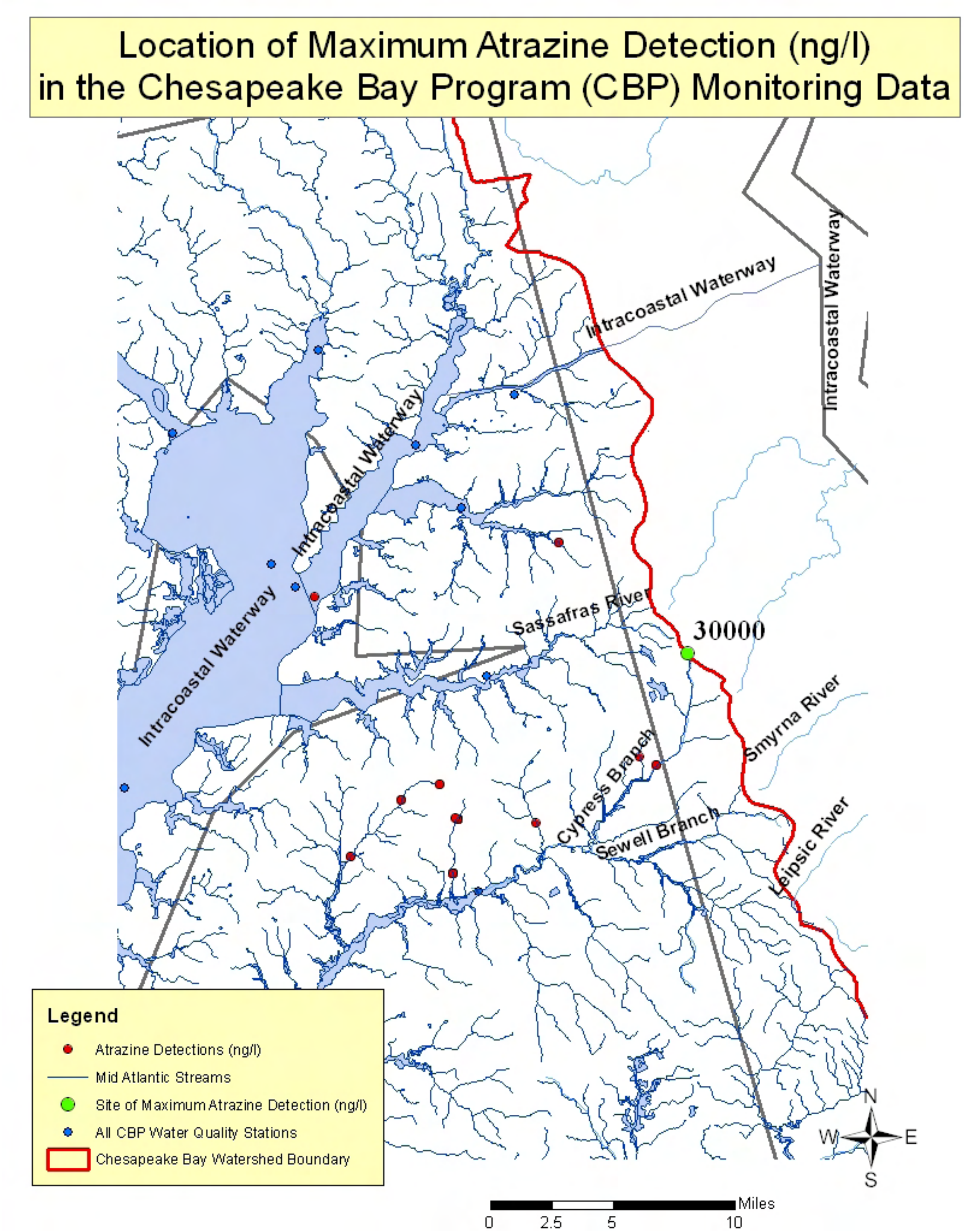


Figure 3-6. Location of Maximum Atrazine Detection (ng/l) in the Chesapeake Bay Watershed in Data from the Chesapeake Bay Program.

3.4.5. Heidelberg College Data

Data from Heidelberg College, consisting of two intensively sampled watersheds (Maumee and Sandusky) in Ohio were also analyzed. Like the national NAWQA data, these data are outside of the action area but is included in this analysis to provide context to the modeled exposures. More information on the water quality monitoring program at Heidelberg College may be found at the following website:

http://wql-data.heidelberg.edu/

The Heidelberg data are collected more frequently than other data included in this assessment. The study design was specifically established to capture peak and longer term trends in pesticide exposures. Data were collected between 1983 and 1999 and consist of an average of roughly 100 samples per year with several days of multiple sampling.

For the Sandusky watershed, a total of 1,597 samples were collected with 1,444 detections of atrazine (90.4% frequency of detection). The maximum concentration detected in the Sandusky watershed was 52.2 µg/L, and the overall average concentration was 4.5 µg/L. For the Maumee watershed, a total of 1,437 samples were collected with 1,305 detections of atrazine (90.8% frequency of detection). The maximum concentration detected in the Maumee watershed was 38.7 µg/L with an overall average concentration of 3.7 µg/L.

This analysis was furthered refined by deriving the annual TWM and maximum concentrations by sampled watershed by year. The results of this analysis are presented in **Table 3.23.** The results show a consistent pattern with that seen in other data collected from high atrazine use areas with general TWM concentrations between 1 and 3 µg/L.

Table 3.23. Annual Time Weighted Mean and Maximum Concentrations (µg/L) for Atrazine in Two Ohio Watersheds from the Heidelberg College Data

	Sandusky		Maumee	
Year	**TWM**	**Max**	**TWM**	**Max**
1983	1.34	7.97	0.98	5.42
1984	1.08	8.73	1.27	11.71
1985	1.83	19.46	1.00	6.21
1986	3.32	24.61	1.64	10.01
1987	1.76	16.45	1.80	9.92

Table 3.23. Annual Time Weighted Mean and Maximum Concentrations (µg/L) for Atrazine in Two Ohio Watersheds from the Heidelberg College Data				
	Sandusky		Maumee	
Year	TWM	Max	TWM	Max
1988	0.41	1.53	0.43	2.15
1989	1.30	15.71	1.07	8.49
1990	1.96	19.31	1.69	14.78
1991	1.49	20.59	2.044	21.45
1992	0.39	40.53	0.51	7.35
1993	1.27	26.34	1.21	22.66
1994	0.86	10.10	0.82	4.02
1995	1.39	15.46	1.30	14.06
1996	1.56	23.40	1.19	16.19
1997[1]	**2.16**	**53.21**	2.09	38.74
1998	1.49	40.03	1.41	27.62
1999	1.57	17.11	1.88	19.37

[1] Sample year 1997 from Sandusky selected for data infilling by interpolation in order to calculate CASM duration exposure values

Unlike other data sets included in this assessment an effort at interpolation between data points was completed in order to estimate 14-day, 30-day, 60-day, and 90-day average concentrations. A final analysis of the data was completed by selecting one year worth of data from the Heidelberg data. 1997 was selected because it was one of the more recent data sets and because the maximum and TWM concentrations were higher than most other year’s data. To process these data it was necessary to “fill in the gaps”. A total of 126 samples were collected during 1997 with 50 days with multiple samples yielding a time series of roughly 75 days. A step-wise approach was used to estimate daily concentrations between sampling dates that consisted of simply extending an analytical result from the date of analysis to the next date. For example, on January 6, 1997, atrazine was detected at a concentration of 0.475 µg/L with the next sample date on January 20, 1997 with a concentration of 0 µg/L. In the step-wise interpolation all dates between January 6 and January 20 were assigned the concentration of 0.475 µg/L. Also, because

January 6 was the first sample date of the year, all previous days were also assigned a value of 0.475 µg/L. This process was repeated throughout the year to fill in the time series and yield 365 days worth of data. In addition, where multiple samples were analyzed on any given day, the highest of the values on that day was assigned. There is significant uncertainty with this type of interpolation because there is no information to suggest whether the interpolated value represents actual exposure. For example, where a significant gap in time exists between two samples, it is unlikely that a continuous concentration exists. More likely is that there are upward and downward fluctuations in exposure, with a greater likelihood that higher exposures are missed between sample times with larger gaps in data points.

Table 3.24 presents the results of this analysis. The analysis suggests that at least for the Sandusky watershed in 1997 the estimated longer-term exposures are less than the modeled estimates for the sorghum scenario by a factor of two to three times.

Table 3.24. Magnitude and Duration Estimates from the 1997 Data from Sandusky Watershed[1]

	14 day	21 day	30 day	60 day	90 day
Maximum	28.26	21.11	18.30	12.38	8.89
90^{th} Percentile	7.55	7.08	7.82	10.23	8.22

[1] Stepwise interpolation was used between samples

3.4.6. U.S. EPA ORD Great Lakes Program – Lake Michigan Mass Balance Project

The U.S. EPA's Office of Research and Development (ORD) and the Great Lakes National Program Office (GLNPO), working with state and local partners, has been collecting and analyzing atrazine data for Lake Michigan in the Lake Michigan Mass Balance (LMMB) study. In addition, ORD has developed and implemented a model to predict future trends in atrazine occurrence in the Lake. The LMMB project and data are not directly comparable to the action area because of its location in the upper Midwest. However, it is included in this assessment because of the fact that it represents a large water body comparable in size to the Chesapeake Bay

The LMMB study includes analytical results for atrazine occurrence in the atmosphere (vapor phase, dry deposition, and wet deposition), surface water tributaries to the Lake, and within Lake Michigan itself. The LMMB modeling framework includes computational transport, mass balance, and bioaccumulation and has 3 levels of spatial resolution (whole Lake, 10 surface segments and 41 water segments, and a high- resolution model consisting of 2,318 surface segments and 44,042 water segments).

In the Lake Michigan basin, atrazine is usually applied to cornfields in the spring to control broadleaf and some grassy weeds, and approximately 850,000 kg is applied annually in the Lake Michigan basin. In the atmospheric component, study results indicate that the predominant atmospheric source of atrazine is precipitation. Atrazine was only detected in 3.7% of vapor phase samples, and while the detection frequency was higher for particulate samples (dry deposition), the mean concentrations during spring (peak atrazine use season) were found at concentrations up to 370 pg/m^3. In precipitation, atrazine was detected at concentrations as high as 2,800 ng/l (2.8 µg/L).

In the tributaries feeding Lake Michigan, atrazine was detected in 99% of samples, and concentrations ranging from 0.064 to 2.7 µg/L were strongly influenced by geography with higher concentrations in the south near atrazine use sites and lower in the north. In the open water portion of the study, the Lake monitoring data showed fluctuating values of atrazine between 0.03 to 0.06 µg/L. Seasonal loadings (spring to early summer) tend to be focused on the southeast and northwest shores. Modeling predicts that, with no increase or decrease in loadings, concentrations in the Lake will increase slightly and level off thereafter.

In general, the monitoring data and modeling from the LMMB study found that overall loadings are expected to be similar to those seen in the CBP data in the open Bay with concentrations generally in the sub-µg/L range in Lake Michigan and Chesapeake Bay. Similarly, significantly higher concentrations of atrazine were found in both settings in the tributaries feeding both Lake Michigan and the Chesapeake Bay. More recent work has been done to develop a model to predict long-term exposures to atrazine throughout the entire Lake and indicate that long term atrazine concentrations tend to be seasonal and higher near shore than in the central portions of the lake.

More details on the LMMB study and atrazine can be found at

http://www.epa.gove/glnpo/lmmb/results/atra_datarpt.html

3.4.7. Summary of Open Literature Sources of Monitoring Data for Atrazine

Atrazine is likely to be persistent in ground water and in surface waters with relatively long hydrologic residence times (such as in some reservoirs) where advective transport (flow) is limited. The reasons for atrazine's persistence are its resistance to abiotic hydrolysis and direct aqueous photolysis, its only moderate susceptibility to biodegradation, and its limited volatilization potential as indicated by a relatively low Henry's Law constant. Atrazine has been observed to remain at elevated concentrations longer in some reservoirs than in flowing surface water or in other reservoirs with presumably much shorter hydrologic residence times in which advective transport (flow) greatly limits its persistence.

A number of open literature studies have been cited in the 2003 IRED (U.S. EPA, 2003a) which document the occurrence of atrazine and its degradates in both surface water and groundwater. These data support the general conclusion of the analysis above that higher exposures tend to occur in the most vulnerable areas in the Midwest and South and that the most vulnerable water bodies tend to be headwater streams and water bodies with little or no flow.

The analysis in the IRED also documents the occurrence of atrazine in the atmosphere. The data indicate that atrazine can enter the atmosphere via volatilization and spray drift. The data also suggest that atrazine is frequently found in rain samples and tends to be seasonal probably related to application timing. Finally, the data suggest that although frequently detected, the concentrations being detected are less than those seen in the monitoring data and modeling conducted as part of this assessment and support the contention that runoff and spray drift are the principal routes of exposure. In general, these detections are located in areas of high atrazine use such as the Midwestern US It is expected that lower amounts will be present in the action area due to lower relative use. More details on these data can be found in the 2003 IRED (U.S. EPA, 2003a).

3.5 Summary of Modeling vs. Monitoring Data

Overall, comparison of the monitoring data with the modeling indicates that, in general, the peak concentrations are reasonably well predicted by modeling with PRZM/EXAMS for all scenarios and iterations of the modeling but that the longer-term average concentrations are over-estimated for flowing water bodies. For this analysis, only the peak and annual average (approximated by averaging across the sample range from the monitoring data) from the monitoring data were comparable to the model output, with the exception of the analysis from the Heidelberg data. The Heidelberg analysis, although highly uncertain due to the nature of the interpolation necessary, suggests that in a highly vulnerable watershed, the longer-term exposures will be less than predicted in streams and rivers with even moderate flow rates.

A number of uncertainties should be considered when comparing the modeled EECs from the static water body with various habitat types and monitoring data. Specifically, the modeled water body represents static water; however, in reality, many water bodies have some amount of flow. For the Chesapeake Bay watershed, it is expected that no-flow, and low-flow water bodies are representative of the headwater streams adjacent to agricultural fields. In addition, water bodies in the Chesapeake Bay watershed increase in flow rate, volume, salinity, and the influence of tidal fluxes and increasing watershed size will result in some dilution due to the influx of non-impacted water. None of these factors are accounted for in the modeled estimates presented in Table 3.5 used for risk estimation. In general, it is expected that modeled atrazine concentrations in the static water body will over-estimate exposure in settings where flow is greater than those modeled and where the volume of the water body is greater than that modeled (20,000,000 liters). It is uncertain what impact differences in water chemistry and tidal influences would have on modeled exposures.

Overall, the uncertainties inherent in the exposure assessment tend to result in over estimation of exposures. This is apparent when comparing modeling results with monitoring data. In general, the monitoring data should be considered a lower bound on exposure while modeling represents an upper bound. Factors influencing the over-estimation of exposure include the assumption of no degradation, dilution, or mixing in the subsurface transport from edge of field to springs. The modeling exercises presented is in actuality assuming the assessed water bodies and application sites are adjacent. In reality, there are likely to be processes at work which cannot be accounted for in the modeling which will reduce the predicted exposures. In addition, the impact of setbacks on runoff estimates have not been quantified while acknowledging that these buffers, especially well-vegetated buffers, are likely to result in considerable reduction in runoff loading of atrazine.

3.6. Oral Exposure to Sea Turtles

3.6.1. Dietary Exposure from Contaminated Food Items

Dietary exposure to the four species of sea turtles considered in this assessment was estimated using the highest reported bioconcentration factor of atrazine of 8.5 L/kg (U.S. EPA, 2003c) and the peak EEC from PRZM/EXAMS reported in Section 3.2. Atrazine concentration in sea turtle food items was estimated using the following calculation:

BCF of 8.5 L/kg x EEC of 0.055 mg/L = 0.47 mg/kg = 0.47 ppm

The dietary concentration of 0.47 ppm was compared with the avian LC50 (ppm) and NOAEC (ppm) for derivation of dietary based risk quotients.

Daily doses (mg/kg-bw) of atrazine were estimated for sea turtles by assuming that a turtle consumes approximately 100% of its weight daily (see below for explanation of this assumption) using the following equation:

Dietary concentration (0.47 mg/kg = 0.47 ppm) / 100% bw consumed = 0.47 mg/kg-bw

The assumption that sea turtles consume 100% of their body weight daily was based on a report by Lutcavage and Lutz (1986), who reported that hatchling leatherbacks consume their weight in food daily. Duron (1978) estimated that adult leatherbacks would need to consume approximately 200 lbs jellyfish daily to satisfy their energy requirements, resulting in consumption of considerably less than 100% of body weight daily, given a small adult leatherback turtle of 260 kg (approximately 570 lbs). Therefore, the assumption that turtles consume 100% of their body weight daily would result in a conservative estimation of exposure. Food consumption data were not located for other sea turtle species; therefore, it was assumed that other sea turtle species also consume no more than 100% of their body weight daily.

3.6.2. Potential Exposure to Sea Turtles from Water Intake

Exposure from water flow-through was estimated based on water turnover rates reported by Wallace et al (2005) in leatherback turtles and Ortiz *et al.* (2000) in Kemp's ridley turtles using the following equation:

Water turnover rate (mL / kg-bw) x (1 L / 1000 mL) x EEC (µg/L) = Dose (µg/kg-bw)

Water influx data were not located for green turtles and loggerhead turtles. Therefore, there is additional uncertainty in the EECs for these turtle species. Results from this analysis are in Table 3.24.

Table 3.24. Water Intake Exposure Estimations for Sea Turtles			
Species	**Water Influx (mL/kg-bw)[c]**	**Atrazine EEC (µg/L)[d]**	**Atrazine Dose in Turtle (µg/kg-bw)**
Leatherback[a]	233	55	13 (0.013 mg/kg-bw)
Kemp's Ridley[b]	123	55	6.8 (0.0068 mg/kg-bw

[a] Maximum of 5 values, which ranged from 106 to 233
[b] Average of 4 values; range not reported
[c] Water influx data reported from Wallace et al (2005) and Ortiz *et al.* (2000)
Atrazine EECs are from Section 3.2 and were derived using PRZM/EXAMS.
[d] Peak EEC from PRZM/EXAMS using the standard water body scenario

These data will be compared with acute avian LD50 values (mg/kg-bw) for direct effects risk estimation in turtles.

4.0 Effects Assessment

This ecological risk assessment evaluates the potential for atrazine to affect six species: shortnose sturgeon, dwarf wedgemussel, and four sea turtle species. Assessment endpoints include direct toxic effects on the survival, reproduction, and growth of the species as well as indirect effects such as reduction of the food supply and/or habitat modification. Direct effects include reduced survival and reproductive impairment from both direct acute (short-term) and direct chronic (long-term) exposures to atrazine. These assessment endpoints, while measured at the individual level, provide insight about risks at higher levels of biological organization (e.g., populations) as described in U.S. EPA (2004).

With respect to atrazine degradates, including hydroxyatrazine (HA), deethylatrazine (DEA), deisopropylatrazine (DIA), and diaminochloroatrazine (DACT), it is assumed that each of the degradates are less toxic than the parent compound. As shown in Table 4.2, comparison of available toxicity information for HA, DIA, and DACT indicates lesser aquatic toxicity than the parent for freshwater fish, invertebrates, and aquatic plants.

Table 4.1 Comparison of Acute Freshwater Toxicity Values for Atrazine and Degradates			
Substance Tested	**Fish LC_{50} (µg/L)**	**Daphnid EC_{50} (µg/L)**	**Aquatic Plant EC_{50} (µg/L)**
Atrazine	5,300	3,500	1
HA	>3,000 (no effects at saturation)	>4,100 (no effects at saturation)	>10,000
DACT	>100,000	>100,000	No data
DIA	17,000	126,000	2,500
DEA	No data	No data	1,000

Although degradate toxicity data are not available for terrestrial plants, lesser or equivalent toxicity is assumed, given the available ecotoxicological information for other taxonomic groups including aquatic plants and the likelihood that the atrazine degradates are expected to lose efficacy as an herbicide.

Therefore, given the lesser toxicity of the degradates, as compared to the parent, concentrations of the atrazine degradates are not assessed, and the focus of this assessment is limited to parent atrazine. The available information also indicates that aquatic organisms are more sensitive to the technical grade (TGAI) than the formulated products of atrazine; therefore, the focus of this assessment is on the TGAI. A detailed summary of the available ecotoxicity information for all atrazine degradates and formulated products is presented in Appendix A.

In addition to registrant-submitted and open literature toxicity information, the community-level endpoints were also used in the evaluation of the potential for atrazine to induce indirect effects to the assessed species via impacts to aquatic plant community structure and function (See Section 4.6). Other sources of information, including use of the acute probit dose response relationship to establish the probability of an individual effect and reviews of the Ecological Incident Information System (EIIS), are conducted to further refine the characterization of potential ecological effects associated with exposure to atrazine. A summary of the available aquatic and terrestrial plant ecotoxicity information, the community-level endpoints, use of the probit dose response relationship, and the incident information for atrazine are provided in Sections 4.1 through 4.8.

As previously discussed in the problem formulation, the available toxicity data show that other pesticides may combine with atrazine to produce synergistic, additive, and/or antagonistic toxic interactions. The results of available toxicity data for mixtures of atrazine with other pesticides are presented in Section A.6 of Appendix A. Synergistic effects with atrazine have been demonstrated for a number of organophosphate insecticides including diazanon, chlorpyrifos, and methyl parathion, as well as herbicides including alachlor. If chemicals that show synergistic effects with atrazine are present in the environment in combination with atrazine, the toxicity of the atrazine mixture may be increased relative to the toxicity of each individual chemical, offset by other environmental factors, or even reduced by the presence of antagonistic contaminants if they are also present in the mixture. The variety of chemical interactions presented in the available data set suggest that the toxic effect of atrazine, in combination with other pesticides used in the environment, can be a function of many factors including but not necessarily limited to (1) the exposed species, (2) the co-contaminants in the mixture, (3) the ratio of atrazine and co-contaminant concentrations, (4) differences in the pattern and duration of exposure among contaminants, and (5) the differential effects of other physical/chemical characteristics of the receiving waters (e.g. organic matter present in sediment and suspended water). Quantitatively predicting the combined effects of all these variables on mixture toxicity to any given taxa with confidence is beyond the capabilities of the available data. However, a qualitative discussion of implications of the available pesticide mixture effects data involving atrazine on the confidence of risk assessment conclusions for the assessed species is addressed as part of the uncertainty analysis for this effects determination.

4.1. Toxicity Data Used to Evaluate Assessment Endpoints

Toxicity endpoints are established based on data generated from guideline studies submitted by the registrant and from open literature studies that meet the criteria for inclusion into the

ECOTOX database maintained by EPA/ORD. Open literature data presented in this assessment were obtained from the 2003 atrazine IRED as well as new information obtained from the ECOTOX database on February 16, 2006. The February 2006 ECOTOX search included all open literature data for atrazine (i.e., pre- and post-IRED). In order to be included in the ECOTOX database, papers must meet the following minimum criteria:

(1) the toxic effects are related to single chemical exposure;
(2) the toxic effects are on an aquatic or terrestrial plant or animal species;
(3) there is a biological effect on live, whole organisms;
(4) a concurrent environmental chemical concentration/dose or application rate is reported; and
(5) there is an explicit duration of exposure.

Data that pass the ECOTOX screen are evaluated along with the registrant-submitted data, and may be incorporated qualitatively or quantitatively into this endangered species assessment. In general, effects data in the open literature that are more conservative than the registrant-submitted data are considered. Based on the results of the 2003 IRED for atrazine, potential adverse effects on sensitive aquatic plants and non-target aquatic organisms including their populations and communities, are likely to be greatest when atrazine concentrations in water equal or exceed approximately 10 to 20 µg/L on a recurrent basis or over a prolonged period of time. Given the large amount of microcosm/mesocosm and field study data for atrazine, only effects data that are less than or more conservative than the 10 µg/L aquatic-community effect level identified in the 2003 atrazine IRED were considered. The degree to which open literature data are quantitatively or qualitatively characterized is dependent on whether the information is relevant to the assessment endpoints (i.e., maintenance of survival, reproduction, and growth) identified in the problem formulation. For example, endpoints such as behavior modifications are likely to be qualitatively evaluated, because it is not possible to quantitatively link these endpoints with reduction in species survival, reproduction, and/or growth (e.g., the magnitude of effect on the behavioral endpoint needed to result in effects on survival, growth, or reproduction is not known).

Citations of all open literature not considered as part of this assessment because it was either rejected by the ECOTOX screen or accepted by ECOTOX but not used (e.g., the endpoint is less sensitive and/or not appropriate for use in this assessment) are included in Appendix J. Appendix J also includes a rationale for rejection of those studies that did not pass the ECOTOX screen and those that were not evaluated as part of this ESA.

The most sensitive endpoint for each taxa evaluated was used for risk quotient calculation (U.S. EPA, 2004). For this assessment, the toxicity data were used to assess endpoints listed in Table 4.1 for the six species considered in this analysis. A description of all effects data considered for this assessment is in Appendix A. Currently, no studies have been conducted on sea turtles, sturgeon, or freshwater mussels. Therefore, surrogate species were used as outlined in U.S. EPA (2004) for characterization of atrazine toxicity to the assessed species and toxicity to other animals on which the assessed species rely for sustenance. Avian studies were used for surrogates for reptiles; the most sensitive fish and bivalve species tested were used to assess potential direct effects to the shortnose sturgeon and dwarf wedgemussel, respectively. In addition, studies located in the open literature were considered for use in the characterization of potential toxicity of atrazine to each of the species assessed as the data allow. A summary of the toxicity data used for this assessment is in Table 4.1.

Table 4.2. Summary of Toxicity Data Used to Evaluate the Assessment Endpoints for the Six Assessed Listed Species.

Toxicity Data[a]	Assessment Endpoint	Species Assessed
Freshwater and saltwater invertebrates EC50, LC50, and NOAEC	Survival, growth, and reproduction via direct effects	Dwarf wedgemussel
	Survival, growth, and reproduction via indirect effects on food supply	Shortnose sturgeon, dwarf wedgemussel, sea turtles (all four species assessed)
Freshwater and saltwater fish LC50 and NOAEC	Survival, growth, and reproduction via direct effects	Shortnose sturgeon
	Survival, growth, and reproduction via effects on food supply	Sea turtles (ambient exposure, all four species assessed)
	Survival, growth, and reproduction via effects on host fish needed to complete life cycle	Dwarf wedgemussel
Acute avian LD50, LC50, and reproduction NOAEC	Survival, growth, and reproduction via direct effects	Sea turtles (oral exposure, all four species assessed)
Freshwater and saltwater aquatic plant EC50	Survival, growth, and reproduction via indirect effects on habitat and/or primary productivity	Shortnose sturgeon, dwarf wedgemussel, sea turtles (all four species assessed)
	Survival, growth, and reproduction via indirect effects on food supply	Green turtle, dwarf wedgemussel
Terrestrial plant EC25	Survival, growth, and reproduction via indirect effects on terrestrial vegetation (riparian habitat) required to maintain acceptable water quality and spawning habitat	All species assessed

[a] Most sensitive single species was initially used in risk estimation for indirect effects; however, dietary requirements and behavior of the assessed species were used as the data allow to refine potential risks if use of the most sensitive food item species resulted in LOC exceedances.

4.2. Toxicity Classification Scheme

Toxicity to fish and aquatic invertebrates is categorized using the following system as outlined in U.S. EPA (2004):

LC_{50} (ppm)	Toxicity Category
< 0.1	Very highly toxic
> 0.1 - 1	Highly toxic
> 1 – 10	Moderately toxic
> 10 - 100	Slightly toxic
> 100	Practically nontoxic

The following classification system (U.S. EPA, 2004) was used to characterize toxicity of atrazine to birds (surrogate for sea turtles):

LC_{50} (ppm)	LD_{50} (mg/kg-bw)	Toxicity Category
<50	<10	Very highly toxic
50 – 500	10 – 50	Highly toxic
501 – 1000	51- 500	Moderately toxic
1001 – 5000	501 – 2000	Slightly toxic
>5000	>2000	Practically nontoxic

Toxicity categories are currently not defined for plants.

4.3. Laboratory Effects Data

This assessment considered both EPA guideline studies and studies located in the open literature. A summary of registrant-submitted and open literature data used in risk estimation is provided in this section. Additional information is in Appendix A.

4.3.1. Toxicity to Fish

4.3.1.1. Acute Exposure (Mortality) Studies

Fish toxicity studies were used to assess potential direct effects to the shortnose sturgeon and potential indirect effects to the dwarf wedgemussel and sea turtles. Dwarf wedgemussels depend on fish to complete their life cycle, and each of the four turtle species may consume fish as part of their diet during all or part of their life cycle (see Appendix D).

Atrazine toxicity has been evaluated in numerous fish species, and the results of these studies demonstrate a wide range of sensitivities to atrazine. LC50 values range from 2000 to 60,000 µg/L (2 mg/L to 60 mg/L, see Appendix A for additional details on these studies). Therefore, atrazine is classified as moderately toxic to fish on an acute basis.

Atrazine has been tested in both saltwater and freshwater species. The most sensitive species was used to calculate risk quotients regardless of the salinity environment because this assessment includes the Chesapeake Bay and its tributaries, which encompass both freshwater and saltwater environments. However, species habitat would be considered if LOCs are exceeded based on RQs derived using the most sensitive LC50. Therefore, the lowest LC50, 2,000 µg/L reported in the estuarine fish sheepshead minnows (MRID 45208303) was used for risk quotient calculations (Table 4.2).

4.3.1.2. Chronic Exposure Studies

The most sensitive chronic studies in fish used in this assessment indicate that atrazine is associated with reduced juvenile survival in the estuarine fish, sheepshead minnows (LOAEC = 3400 µg/L) and with reduced growth (7% reduction in length and 16% reduction in weight relative to controls) in freshwater brook trout (LOAEC = 120 µg/L). No effects (NOAEC) were observed in these species at 1900 µg/L and 65 µg/L, respectively. The most sensitive NOAEC of 65 µg/L (MRID 00024377) was used to calculate risk quotients.

Table 4.3. Summary of Fish Toxicity Studies Used In Risk Quotient Calculations.

Reference (MRID)	Species Tested	Study Type/Endpoints	Toxicity Value	Comment
Hall *et al* . 1994 (MRID 45208303)	Sheepshead Minnow	96-hour acute / mortality	LC50: 2000 µg/L Probit slope: 4.4[a] (95% CI: 2.8 – 5.9)	None
Macek *et al.* 1976 (MRID 00024377)	Brook trout	44-Week life-cycle / growth and reproduction	NOAEC: 65 µg/L LOAEC: 120 µg/L	NOAEC based on reduced size and weight
Ward & Ballantine 1985 (MRID 45202920)	Sheepshead Minnow	Early life stage / growth and reproduction	NOAEC: 1900 µg/L LOAEC: 3400 µg/L	89% reduction in juvenile survival was observed at the LOAEC of 3400 µg/L.

[a] A reliable probit slope could not be estimated for the most sensitive study; therefore, a slope of 4.4 was used from a different study in sheepshead minnows of equivalent duration (MRID 43344901). This analysis is consistent with methods described in U.S. EPA (2004a) and results in a more conservative estimation of the probability of an individual effect than the default slope recommended in U.S. EPA (2004a) of 4.5.

4.3.1.3. Sublethal Effects and Additional Open Literature Information In Freshwater Fish

In addition to submitted studies, data were located in the open literature that report sublethal effect levels to freshwater fish that are less than the selected measures of effect summarized in Table 4.1. Although these studies report potentially sensitive endpoints, effects on survival, growth, or reproduction were not observed in the four available life-cycyle studies at concentrations that induced the reported sublethal effects described below and in Appendix A. Therefore, these sublethal endpoints were not used for risk estimation purposes. In the life-cycle study design, fish are exposed to atrazine from one stage of the life cycle to at least the same stage of the next generation (e.g. egg to egg). Therefore, exposure occurs during the most sensitive life stages and during the entire reproduction cycle.

Reported sublethal effects in adult largemouth bass show increased plasma vitellogenin levels in both female and male fish at 50 µg/L and decreased plasma testosterone levels in male fish at atrazine concentrations greater than 35 µg/L (Wieser and Gross, 2002 [MRID 456223-04]). Vitellogenin (Vtg) is an egg yolk precursor protein expressed normally in female fish and dormant in male fish. The presence of Vtg in male fish is used as a molecular marker of exposure to estrogenic chemicals. It should be noted, however, that there is a high degree of variability with the Vtg effects in these studies, which confounds the ability to resolve the effects of atrazine on plasma steroids and vitellogenesis.

Effects of atrazine on freshwater fish behavior, including a preference for the dark part of the aquarium following one week of exposure (Steinberg et al., 1995 [MRID 452049-10]) and a reduction in grouping behavior following 24-hours of exposure (Saglio and Trijase, 1998 [MRID 452029-14]), have been observed at atrazine concentrations of 5 μg/L. In addition, alterations in rainbow trout kidney histology have also been observed at atrazine concentrations of 5 μg/L and higher (Fischer-Scherl et al., 1991 [MRID 452029-07]).

In salmon, potentially sensitive endpoints that have been reported included effects on gill physiology and endocrine-mediated olfactory functions. Data from Waring and Moore (2004; ECOTOX #72625) suggest that salmon smolt gill physiology, represented by changes in Na-K-ATPase activity, was altered at 2 μg/L atrazine and higher. Survival was evaluated after transfer to full salinity sea water (33 $^{o}/_{oo}$). Atrazine exposure for 5 to 7 days in freshwater followed by transfer to full salinity sea water resulted in higher mortality at atrazine concentrations of 14 ug/L (14% mortality) and higher in one study and at 1 ug/L (15% mortality) and higher in a separate experiment presented in the publication (no controls died; statistical significance was not indicated). As noted in Appendix D, observational and experimental evidence suggests that shortnose sturgeon prefer habitats with less than 5$^{o}/_{oo}$ for all life history stages during summer months (U.S. EPA, 2003b). Based on distributional evidence, older juvenile and adult shortnose sturgeon are limited to oligohaline and low mesohaline regions of estuaries (<15$^{o}/_{oo}$). The salinity used in by Waring and Moore (2004) simulated full strength seawater (33 $^{o}/_{oo}$). Therefore the relevance of findings from this study to the shortnose sturgeon is questionable.

Moore and Lower (2001; ECOTOX #67727) reported that endocrine-mediated functions of male salmon parr were affected at 1 μg/L atrazine. The reproductive priming effect of the female pheromone prostaglandin $F_{2\alpha}$ on the levels of expressible milt in males was reduced relative to controls after exposure to atrazine at 0.5 μg/L. Although the hypothesis was not tested, the study authors suggest that exposure of smolts to atrazine during the freshwater stage may potentially affect olfactory imprinting to the natal river and subsequent homing of adults. However, no quantitative relationship is established between reduced olfactory response of male epithelial tissue to the female priming hormone in the laboratory and reduction in salmon reproduction (i.e., the ability of male salmon to detect, respond to, and mate with ovulating females). A negative control was not included as part of the study design; therefore, potential solvent effect cannot be evaluated. Furthermore, the study did not determine whether the decreased response of olfactory epithelium to specific chemical stimuli would result in similar responses in intact fish.

Although these studies raise questions about the effects of atrazine on plasma steroid levels, behavior modifications, gill physiology, and endocrine-mediated functions in freshwater and anadromous fish, the data do not allow for a derivation of a quantitative link between these sublethal effects and the selected assessment endpoints for the assessed species (i.e., survival, growth, and reproduction of individuals). Also, effects on survival, growth, or reproduction were not observed in the four available life-cycle studies at concentrations that induced these reported sublethal effects. Therefore, potential sublethal effects to fish are considered qualitatively in

Section 5.2, but are not used as part of the quantitative risk characterization. Further detail on sublethal effects to fish is provided in Sections A.2.4a and A.2.4b of Appendix A.

4.3.2. Toxicity to Aquatic Invertebrates

Aquatic invertebrate toxicity studies were used to assess potential direct effects to the dwarf wedgemussel and potential indirect effects to the shortnose sturgeon, all four sea turtles, and the dwarf wedgemussel as outlined in Table 4.3.

4.3.2.1. Acute Toxicity Studies

Atrazine is classified as very highly toxic to slightly toxic to aquatic invertebrates on an acute exposure basis with LC50 and EC50 values ranging from 88 µg/L to >33,000 µg/L. A chemical is considered very highly toxic if the LC50 is less than 100 µg/L and slightly toxic if the LC50 is between 10,000 and 100,000 µg/L. The acute toxicity data in invertebrates indicate a wide range of sensitivity across species. Furthermore, considerable variability in sensitivity was observed across studies conducted using the same species (Figure 4-1). The most sensitive (lowest) LC50 value for a given species was used for risk estimation. Therefore, this risk assessment may overestimate or underestimate toxicity to some taxa under some environmental conditions.

Data Used for Direct Effects Assessment

The dwarf wedgemussel is the only listed aquatic invertebrate included in this assessment for direct effects. The Eastern oyster was used as a surrogate species for the dwarf wedgemussel. The acute toxicity data demonstrated that the shell deposition EC50 value in Eastern oyster (*Crassostrea virginica*) was >1,700 µg/L (MRID 46648201); no treatment related effects were observed in this study at any concentration. A second study in the Eastern oyster also produced no effects at the highest concentration tested of 1000 µg/L (MRID 46648201). In addition, the Pacific oyster was tested with a wettable powder formulated product. That study (MRID 45227722) produced an EC50 >100 µg/L.

Because none of the studies produced definitive EC50 values (no clear treatment-related effects at any concentration tested), an EC50 of >1700 µg/L was used for risk estimation for direct effects to the dwarf wedgemussel.

One additional acute study in freshwater mussels was located in the open literature. The results of the study by Johnson *et al.* (1993) suggest that 48-hour exposures at atrazine concentrations up to 60 mg/L (60,000 µg/L) do not affect the survival of juvenile and mature freshwater mussels, *Anodonta imbecilis*. This study was not considered suitable for use in RQ calculations; however, it was considered to be of good quality and useful in risk characterization discussion. The study in the freshwater mussel, *A. imbecilis,* do not suggest that use of Eastern Oysters in risk estimation resulted in an underestimation of potential risk of direct effects to dwarf wedgemussels.

Data Used for Indirect Effects Assessment

Aquatic invertebrate toxicity data were also used to evaluate potential indirect effects to each of the six listed species because each assessed species depends on aquatic invertebrates for sustenance. For the indirect effects assessment, the most sensitive aquatic invertebrate species was initially used for risk estimation, which is consistent with U.S. EPA (2004). The most sensitive organism tested was the marine copepod. The lowest LC50 in this species was 88 µg/L; however, a wide range of LC50 values have been reported in copepods from studies that tested technical grade atrazine (LC50 values of 88, 94, 140, 500, 4300, and 7900 µg/L have been reported, see Appendix A). Reasons for the disparity across the reported acute toxicity values in the copepod are unknown. However, similar variability has been observed in other species that have been tested by multiple laboratories. For example, studies conducted in the midge produced LC50s that spanned 2 orders of magnitude (values ranged from 720 to >33,000 µg/L). Other than the copepod, all reported acute toxicity values for the other 12 aquatic invertebrate species tested are 720 µg/L and higher.

The distribution of available toxicity data are summarized in Figure 4-1 below. These studies are described in greater detail in Appendix A.

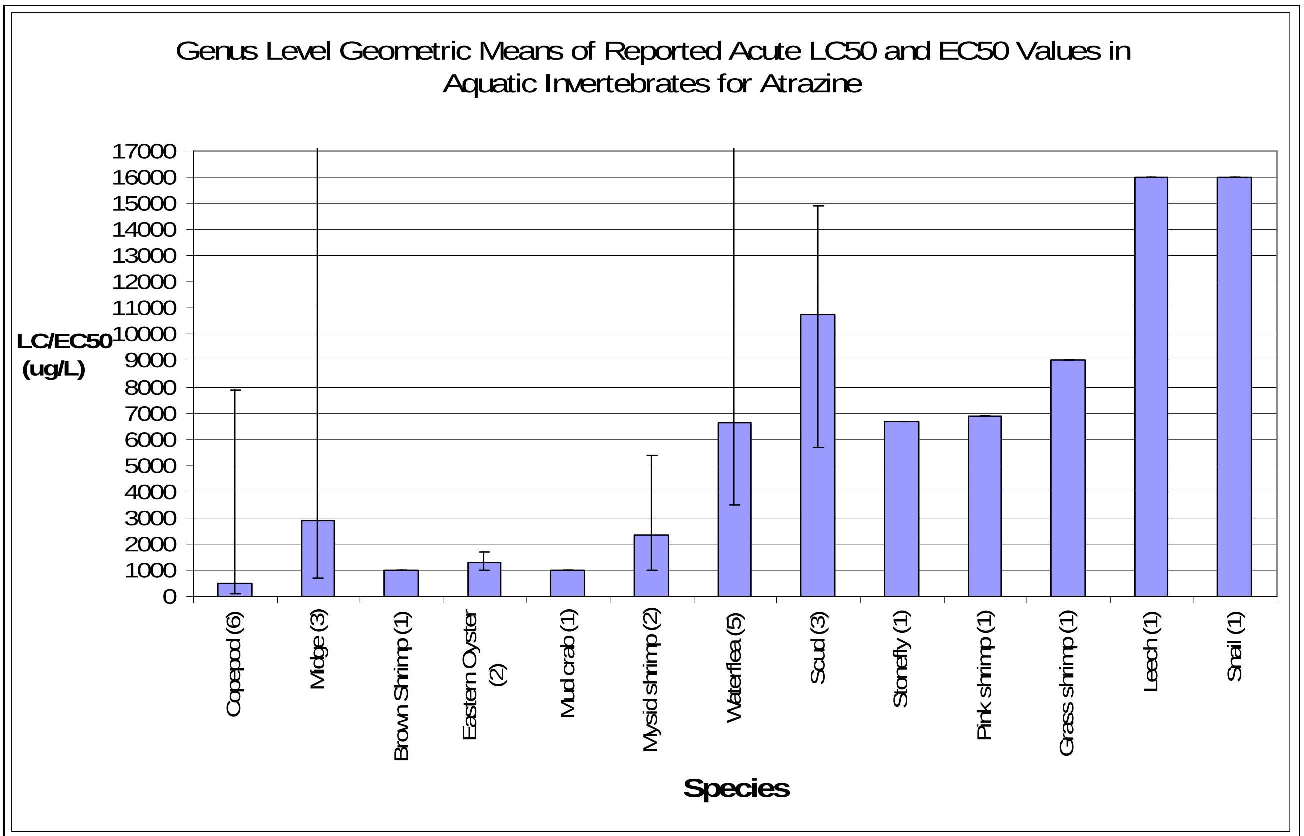


Figure 4-1. Range of Aquatic Invertebrate Acute Toxicity Values Reported for Atrazine

The columns in the above graph represent geometric means of the acute toxicity values (genus levels). The error bars represent the range of reported values. Error bars higher than the maximum value of 17,000 µg/L were reported for two species. These values are 33,000 µg/L for the midge and 30,000 µg/L for the waterflea. Values in parentheses represent the number of studies included in the analysis. No effects were observed in the mud crab and eastern oyster studies. See Appendix A for a description of the studies used in the generation of Figure 4-1.

4.3.2.2. Chronic Exposure Studies

The most sensitive chronic endpoint for freshwater invertebrates was based on a 30-day flow-through study on the scud, which showed a 25% reduction in the development of F_1 to the seventh instar at atrazine concentrations of 140 µg/L; the corresponding NOAEC was 60 µg/L (MRID 00024377).

The most sensitive chronic bioassay in saltwater species was a 28-day study in mysid shrimp (*Americamysis bahia*) that reported a NOAEC of 80 µg/L; a 37% reduction in juvenile survival occurred at the LOAEC of 190 µg/L. Additional details on this study (MRID 45202920) and other chronic bioassays are described in Appendix A.

An uncertainty in the chronic bioassay data is that chronic toxicity data suitable for risk quotient derivation are not available on the most acutely sensitive marine invertebrate (copepod). The potential impact of this uncertainty in risk estimation is described in Section 6. However, the absence of a chronic NOAEC in copepods is not expected to change conclusions of this risk assessment.

Also, a chronic study in bivalves was not available. However, the direct effects assessment to the dwarf wedgemussel was considered protective because the acute data in the Eastern oyster and in a freshwater mussel (*A. imbecilis*) demonstrated low sensitivity to atrazine relative to other aquatic invertebrates tested. Therefore, use of the most sensitive aquatic invertebrate (scud, MRID 00024377) chronic NOAEC in risk estimation for direct effects to the dwarf wedgemussel is considered protective. Acute and chronic studies used to calculate risk quotients for aquatic invertebrates are summarized in Table 4.3.

Table 4.4. Acute and Chronic Aquatic Invertebrate Toxicity Values Used in Initial Risk Estimation of Atrazine					
Reference (MRID)	**Species Tested**	**Species Assessed**	**Study Type / Endpoints**	**Toxicity Value (µg/L)**	**Comment**
Thursby *et al.*, 1990 (MRID 45202918)	Saltwater invertebrate, Copepod (*Acartia tonsa*)	Shortnose sturgeon and sea turtles / indirect effects from reduction in animal food supply	Acute toxicity / mortality	LC50: 88 Probit Slope: 0.95	Data used as initial screen to assess indirect effects to listed species from reduction of animal food supply.
Ward & Ballantine, 1985 (MRID 45202920)	Saltwater invertebrate, Mysid shrimp *(Americamysis bahia)*	Sea turtles / indirect effects from reduction in animal food supply	Chronic exposure / growth and survival	NOAEC: 80	37% Reduction in survival occurred at the LOAEC of 190 µg/L. Data used as initial screen to assess indirect effects to listed species from reduction of animal food supply.
Macek *et al.* 1976 (MRID 00024377)	Freshwater invertebrate, Scud	Dwarf wedgemussel / direct chronic effects Shortnose sturgeon / indirect effects from reduction in animal food supply	Chronic exposure / 25 % red. in development of F_1 to seventh instar.	NOAEC: 60	Chronic bivalve data were not available; therefore, this study, as the most sensitive aquatic invertebrate chronic study, was used to characterize potential chronic toxicity of atrazine to the dwarf wedgemussel.
Johnson 1986 (MRID 45087413)	Freshwater invertebrate, daphnid	Dwarf wedgemussel / indirect effects from reduction in food supply	Acute exposure / immobilization	EC50: 3500 Probit slope: Sufficient data not available[a]	Raw data not included.
Caferalla, 2005b (MRID 46648201); Mayer 1986 (MRID 40228-01)	Bivalve, Eastern oyster	Dwarf wedgemussel / direct acute effects	Acute exposure / shell deposition	EC50: >1000 and >1700 Probit slope: None (no effects occurred)	Endpoint chosen to assess potential direct effects to the dwarf wedgemussel was 1700 µg/L for risk quotient calculations because no treatment related effects occurred in either study.

a Slope information on the toxicity study that was used to derive the RQ for freshwater invertebrates is not available. Therefore, the probability of an individual effect was calculated using a probit slope of 4.4, which is the only technical grade atrazine value reported in the available freshwater invertebrate acute studies; 95% confidence intervals could not be calculated based on the available data (Table A-18). Use of a probit slope of 4.4 would result in a more conservative estimation of the probability of an individual effect than the default slope recommended in U.S. EPA (2004a) of 4.5.

4.3.3. Toxicity to Sea Turtles

Available toxicity data in turtles or other reptiles are limited (summarized in Table 4.4), and no data in sea turtles were located. Therefore, birds were used as a surrogate species for the characterization of atrazine effects to turtles, in accordance with the Overview Document (U.S. EPA, 2004). Birds are considered a conservative surrogate species for the evaluation of potential risks to sea turtles the following reasons:

- Reptiles are poikilotherms (body temperature varies with environmental temperature) while birds are homeotherms (temperature is regulated, constant, and largely independent of environmental temperatures). As a consequence, the caloric requirements of reptiles are markedly lower than birds. Therefore, on a daily dietary intake basis, birds consume more food than reptiles. This can be seen when comparing the caloric requirements for free living iguanid lizards to Passeriformes (song birds) (U.S. EPA, 1993):

  iguanid FMR (kcal/day)= 0.0535 (bw g)^0.799

  passerine FMR (kcal/day) = 2.123 (bw g)^0.749

  With relatively comparable slopes to the allometric functions, one can see that, given a comparable body weight, the free living metabolic rate of birds can be 40 times higher than reptiles, though the requirement differences narrow with high body weights.

  Because the existing risk assessment process is driven by the dietary route of exposure, a finding of safety for birds, with their much higher feeding rates and therefore higher dietary exposure, is reasoned to be protective of reptiles. For this not to be the case, a reptile would have to be 40 times more sensitive than birds for the differences in dietary uptake to be negated. The existing reptile toxicity data (Table 4.4), although limited in its utility, do not suggest that reptiles are more sensitive than birds to atrazine. In addition, conservative assumptions were made to estimate exposure to sea turtles (Section 3).

  For these reasons, the assessment based on toxicity studies in birds as a surrogate species is considered protective of sea turtles. Toxicity values used to calculate risk quotients for sea turtles are summarized in Table 4.4 below.

The available data in birds suggest that atrazine is slightly toxic to avian species on an acute oral exposure basis. The lowest reported LD50 is 940 mg/kg-bw. Signs of intoxication in mallards first appeared 1 hour after treatment and persisted up to 11 days (U.S. EPA, 2003a). In pheasants, remission of signs of intoxication occurred by 5 days after treatment. Signs of intoxication included weakness, hyper-excitability, ataxia, and tremors; weight loss occurred in mallards.

One degradate (desethyl atrazine, DEA) has been shown to be roughly as toxic as atrazine to birds on an acute oral basis. Other degradates evaluated, including deisopropyl atrazine (DIA) and hydroxyatrazine (HA) are considerably less toxic than atrazine to birds on an acute oral basis (Appendix A). However, DACT, which has been shown to be of equivalent toxicity compared with atrazine in mammals, has not been tested in birds.

Because all subacute avian LC_{50} values are greater than 5,000 ppm, atrazine is categorized as practically non-toxic to avian species on a subacute dietary basis. In the subacute dietary study in mallard ducks, 30% mortality was observed at the highest test concentration of 5,000 ppm (MRID 00022923). The time to death was Day 3 for the one Japanese quail and Day 5 for three mallard ducks (U.S. EPA, 2003).

Reproduction studies in birds have reported reproductive effects at atrazine concentrations as low as 675 ppm. In bobwhite quail, the following endpoints were affected at 675 ppm atrazine: egg production, embryo viability, hatchling and 14-day weight, and number of defective eggs (MRID 42547102). Bobwhite and mallard tests show similar toxic effects on reduced egg production and embryo viability/hatchability with LOAEC and NOAEC values of 675 and 225 ppm, respectively, for both species. Although the bobwhite test showed a 7 to 18% reduction in 14-day body weight in the 75 ppm treatment group relative to the control group, the reproductive endpoints were considered to be more biologically significant, given the use of the avian data as a surrogate for sea turtles in the Chesapeake Bay. However, use of 75 ppm instead of 225 ppm would not impact conclusion in this assessment as discussed in Section 5.

Several studies in turtles were located in the open literature. These studies, which are described in Appendix A, suggest that atrazine does not permeate the outer egg shell of reptiles including turtles and alligators after direct application to the egg (MRIDs 45545303 and 45545302) or cause significant alteration in gonadal development and aromatase activity in the snapping turtle or alligator under the conditions of the available studies (De Solla *et al.*, 2005; Crain *et al.*, 1999). Although these data do not allow for derivation of risk quotients, they suggest that reptiles are not more sensitive than birds to potential atrazine effects.

Table 4.5. Summary of Available Acute Oral, Subacute Dietary, and Reproduction Toxicity Studies in Birds, and Available Studies in Reptiles.

Test material/ Reference (MRID)	Species Tested	Study Type/Endpoints	Toxicity Value	Comment
Technical grade atrazine Fink 1976 (MRID 00024721)	Northern bobwhite quail (*Colinus virginianus*)	Acute oral gavage toxicity / mortality	LD50: 940 mg/kg-bw Slope = 3.8 (95% CI: 2.0 – 5.7)	The range of acute oral gavage LD50s in birds is 940 mg/kg-bw to 4200 mg/kg-bw (Appendix A).
Degradate Desethyl Atrazine (DEA) Stafford, 2005c (MRID 46500009)	Northern bobwhite quail (*Colinus virginianus*)	Acute oral gavage toxicity / mortality	LD50: 768 mg/kg-bw slope = 6.2 (95% CI: 3.2 – 9.3)	These data suggest that the degradate DEA is approximately as toxic to birds on an acute oral basis as atrazine.
Technical grade atrazine Hill *et al.* 1975 (MRID 00022923)	Mallard duck (*Anas platyrhynchos*)	Subacute dietary / mortality	LC50: > 5,000 (30 % mortality at 5,000 ppm)	All submitted subacute dietary studies in birds report LC50s that are higher than 5,000 ppm.
Technical grade atrazine Pedersen & DuCharme 1992 (MRID 42547102)	Northern bobwhite (*Colinus virginianus*)	Dietary Exposure / Reproduction effects	NOAEC: 225 ppm LOAEC : 675 ppm	At the LOAEC, egg production, embryo viability, and hatchling weight were affected.
Technical grade atrazine Pedersen & DuCharme 1992 (MRID 42547101)	Mallard duck (*Anas platyrhynchos*)	Dietary Exposure / Reproduction effects	NOAEC: 225 ppm LOAEC : 675 ppm	At the LOAEC, egg production, egg hatchability, and food consumption were affected.
Technical grade atrazine De Solla *et al.*, 2005; Ecotox Reference No. 82032	Snapping turtles	4-Month exposure study in developing embryos / gonad development	NOAEC: 13.2 lbs a.i./Acre (8.1 ppm soil), highest rate tested	No treatment-related effects were observed at the highest concentration tested.
Technical grade atrazine Gross, 2001 (MRIDs 45545303 and 45545302)	Red-eared slider turtle and American Alligator	10-Day egg exposure / endocrine effects	NOAEC: 500 µg/L, highest concentration tested	No treatment-related effects were observed at the highest concentration tested.

4.4. Terrestrial Plant Toxicity

Terrestrial plant toxicity data are used to evaluate the potential for atrazine to affect the riparian zone. Riparian zone effects could impact habitat and stream water quality as discussed in detail in Section 5.2.

Plant toxicity data from both registrant-submitted studies and studies in the scientific literature were reviewed for this assessment. Registrant-submitted studies are conducted under conditions and with species defined in EPA toxicity test guidelines. Sub-lethal endpoints such as plant growth, dry weight, and biomass are evaluated for both monocots and dicots, and evaluate effects at both seedling emergence and vegetative life stages. A guideline study generally evaluates toxicity to ten crop species. A drawback to these tests is that they are conducted on herbaceous agricultural crop species only, and extrapolation of effects to other species, such as woody shrubs and trees and wild herbaceous species, contributes uncertainty to risk conclusions. However, atrazine is labeled for use in forestry production; therefore effects to these types of trees are not anticipated at concentration anticipated in the environment. In addition, preliminary data (discussed below) suggests that sensitive woody plant species exist; however, damage to most woody species at labeled application rates is not expected.

Commercial crop species have been selectively bred, and may be more or less resistant to particular stressors than wild herbs and forbs. The direction of this uncertainty for specific plants and stressors, including atrazine, is largely unknown. Homogenous test plant seed lots also lack the genetic variation that occurs in natural populations, so the range of effects seen from tests is likely to be smaller than would be expected from wild populations.

Based on the results of the submitted terrestrial plant toxicity tests, it appears that emerged seedlings are more sensitive to atrazine via soil/root uptake exposure than emerged plants via foliar routes of exposure. However, all tested plants, with the exception of corn in the seedling emergence and vegetative vigor tests and ryegrass in the vegetative vigor test, exhibited adverse effects following exposure to atrazine.

For Tier II seedling emergence, the most sensitive dicot is the carrot and the most sensitive monocots are oats. EC_{25} values, on an equivalent application rate basis, for oats and carrots, which are based on a reduction in dry weight, are 0.003 and 0.004 lb ai/A, respectively; NOAEC values for both species are 0.0025 lb ai/A. Table 4.5 summarizes the Tier II terrestrial plant seedling emergence toxicity data.

For Tier II vegetative vigor studies, the most sensitive dicot is cucumber and the most sensitive monocot is onion. In general, dicots appear to be more sensitive than monocots via foliar routes of exposure with all tested monocot species showing a significant reduction in dry weight at EC_{25} values ranging from 0.008 to 0.72 lb ai/A. In contrast, two of the four tested monocots showed no effects from atrazine (corn and ryegrass), while EC_{25} values for oats and onion were 0.61 and 2.4 lb ai/A, respectively. Table 4.6 summarizes the terrestrial plant vegetative vigor toxicity data used to derive risk quotients in this assessment.

Table 4.5. Nontarget Terrestrial Plant Seedling Emergence Toxicity (Tier II).

Surrogate Species	% ai	EC_{25} / NOAEC (lbs ai/A)	Endpoint Affected	MRID No. Author/Year	Study Classification
Monocot - Corn (*Zea mays*)	97.7	> 4.0 / > 4.0	No effect	420414-03 Chetram 1989	Acceptable
Monocot - Oat (*Avena sativa*)	97.7	**0.004 / 0.0025**	red. in dry weight	420414-03 Chetram 1989	Acceptable
Monocot - Onion (*Allium cepa*)	97.7	0.009 / 0.005	red. in dry weight	420414-03 Chetram 1989	Acceptable
Monocot - Ryegrass (*Lolium perenne*)	97.7	0.004 / 0.005	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Root Crop - Carrot (*Daucus carota*)	97.7	**0.003 / 0.0025**	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Soybean (*Glycine max*)	97.7	0.19 / 0.025	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Lettuce (*Lactuca sativa*)	97.7	0.005 / 0.005	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Cabbage (*Brassica oleracea alba*)	97.7	0.014 / 0.01	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Tomato (*Lycopersicon esculentum*)	97.7	0.034 / 0.01	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Cucumber (*Cucumis sativus*)	97.7	0.013 / 0.005	red. in dry weight	420414-03 Chetram 1989	Acceptable

Table 4.6. Nontarget Terrestrial Plant Vegetative Vigor Toxicity (Tier II).

Surrogate Species	% ai	EC25 / NOAEC (lbs ai/A)	Endpoint Affected	MRID No. Author/Year	Study Classification
Monocot - Corn (*Zea mays*)	97.7	> 4.0 / > 4.0	No effect	420414-03 Chetram 1989	Acceptable
Monocot - Oat (*Avena sativa*)	97.7	2.4 / 2.0	red. in dry weight	420414-03 Chetram 1989	Acceptable
Monocot - Onion (*Allium cepa*)	97.7	**0.61 / 0.5**	red. in dry weight	420414-03 Chetram 1989	Acceptable
Monocot - Ryegrass (*Lolium perenne*)	97.7	> 4.0 / > 4.0	No effect	420414-03 Chetram 1989	Acceptable
Dicot - Root Crop - Carrot (*Daucus carota*)	97.7	1.7 / 2.0	red. in plant height	420414-03 Chetram 1989	Acceptable
Dicot - Soybean (*Glycine max*)	97.7	0.026 / 0.02	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Lettuce (*Lactuca sativa*)	97.7	0.33 / 0.25	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Cabbage (*Brassica oleracea alba*)	97.7	0.014 / 0.005	red. in dry weight	420414-03 Chetram 1989	Acceptable
Dicot - Tomato (*Lycopersicon esculentum*)	97.7	0.72 / 0.5	red. in plant height	420414-03 Chetram 1989	Acceptable
Dicot - Cucumber (*Cucumis sativus*)	97.7	**0.008 / 0.005**	red. in dry weight	420414-03 Chetram 1989	Acceptable

In addition, a report on the toxicity of atrazine to woody plants (Wall *et al.*, 2006; MRID 4687040001) was reviewed by the Agency. A total of 35 species were tested at application rates ranging from 1.5 to 4.0 lbs ai/A. Twenty-eight species exhibited either no or negligible phytotoxicity. Seven of 35 species exhibited >10% phytotoxicity. However, further examination of the data indicate that atrazine application was clearly associated with severe phytotoxicity in only one species (Shrubby Althea). These data suggest that, although sensitive woody plants exist, atrazine exposure to most woody plant species at application rates of 1.5 to 4.0 lbs ai/A is not expected to cause adverse effects. A summary of the available woody plant data is provided in Table A-39b of Appendix A.

4.5. Aquatic Plant Toxicity Data

Aquatic plant toxicity studies were used to evaluate whether atrazine may affect primary productivity in the Chesapeake Bay and its source waters or direct food source for the dwarf wedgemussel and green turtles, both of which use plants as a primary component of their diets. Two types of studies were used to evaluate the potential of atrazine to affect primary productivity. The most sensitive EC50 from available laboratory studies was initially used to derive risk quotients to determine whether atrazine may affect aquatic plants. Threshold concentrations predictive of potential community level effects to aquatic plants were also used to further characterize indirect effects to the assessed species. Laboratory data are described in Section 4.5.1., field studies are described in Section 4.5.2., and community-level threshold concentrations are described in Section 4.5.3.

Recovery from the effects of atrazine and the development of resistance to the effects of atrazine in some vascular and non-vascular aquatic plants has been reported and may add uncertainty to these findings. However, reports of recovery are often based on differing interpretations. Thus, before recovery can be considered as an uncertainty, an agreed upon interpretation is needed. For the purposes of this assessment, recovery is defined as a return to pre-exposure levels for the *affected community*, not for a replacement community of more tolerant species. Further research is needed to quantify the impact that recovery and resistance would have on aquatic plants.

4.5.1. Laboratory Data

Numerous aquatic plant toxicity studies have been submitted. A summary of these studies is presented below. See Appendix A for a more comprehensive description of these data. The Tier II results for freshwater aquatic plants indicate that atrazine causes a 41 to 98% reduction in chlorophyll production of freshwater algae; the corresponding EC_{50} value for four different species of freshwater algae is 1 µg/L, based on data from a 7-day acute study (MRID 00023544). Vascular plants are less sensitive to atrazine than their freshwater non-vascular plants with an EC_{50} value of 37 µg/L, based on reduction in duckweed growth (MRID 43074804).

In marine species, the marine algae *Isochrysis galbana* is the most sensitive nonvascular aquatic plant (EC_{50} = 22 µg/L; MRID 41065204), and the most sensitive vascular aquatic plant is Sago pondweed (EC50 = 7.5 µg/L; MRID 45088231). EC50s for sea grasses, which are important forage material for green turtles, range from approximately 70 µg/L (MRID 45227729) to 30,000 µg/L (MRID 45205101) in laboratory studies.

4.5.2. Field Data

Microcosm and mesocosm studies with atrazine provide measurements of primary productivity that incorporate the aggregate responses of multiple species in aquatic plant communities. Because plant species vary widely in their sensitivity to atrazine, the overall response of the plant community may be different from the responses of the individual species measured in laboratory toxicity tests. Mesocosm and microcosm studies allow observation of population and community recovery from atrazine effects and of indirect effects on higher trophic levels. In addition, mesocosm and microcosm studies, especially those conducted in outdoor systems, incorporate partitioning, degradation, and dissipation, factors that are not usually accounted for in laboratory toxicity studies, but that may influence the magnitude of ecological effect.

Atrazine has been the subject of many mesocosm and microcosm studies in ponds, streams, lakes, and wetlands. The duration of these studies have ranged from a few weeks to several years in duration at exposure concentrations from 0.1 µg/L to 10,000 µg/L. Most of the studies have focused on atrazine effects on phytoplankton, periphyton, and macrophytes; however, some have also included measurements on animals.

Based on the results of the 2003 IRED for atrazine, potential adverse effects on sensitive aquatic plants and non-target aquatic organisms including their populations and communities are likely to be greatest when atrazine concentrations in water equal or exceed approximately 10 to 20 µg/L on a recurrent basis or over a prolonged period of time. A summary of all the freshwater aquatic microcosm, mesocosm, and field studies that were summarized as part of 2003 IRED is included in Appendix A. In addition, a number of estuarine/marine field studies are available, which are also discussed in Appendix A (Section A.3.7). Given the large amount of microcosm and mesocosm and field study data for atrazine, only effects data less than or more conservative than the 10 µg/L aquatic community effect level identified in the 2003 IRED were considered as part of the open literature search. Based on the selection criteria for review of new open literature, all of the available studies show effects levels to freshwater and estuarine/marine fish and invertebrates at concentrations greater than 10 µg/L.

4.6. Community-Level Endpoints: Threshold Concentrations

In this ESA, direct and indirect effects to the assessed listed species are evaluated in accordance with the screening-level methodology described in the Agency's Overview Document (U.S. EPA, 2004). If aquatic plant RQs exceed the Agency's non-listed species LOC (because the assessed species do not have an obligate relationship with any one particular plant species, but rather rely on multiple plant species), based on available EC_{50} data for vascular and non-vascular plants, risks to individual aquatic plants are assumed.

It should be noted, however, that the indirect effects and components of the critical habitat impact analyses in this assessment are unique, in that the best available information for atrazine-related effects on aquatic communities is significantly more extensive than for other pesticides. Hence, atrazine effects determinations can utilize more refined data than is generally available to the Agency. Specifically, a robust set of microcosm and mesocosm data and aquatic ecosystem models are available for atrazine that allowed EPA to refine the indirect effects and critical habitat impact analysis associated with potential aquatic community-level effects (via aquatic plant community structural change and subsequent habitat modification) to the listed species. Use of such information is consistent with the guidance provided in the Overview Document (U.S. EPA, 2004), which specifies that "the assessment process may, on a case-by-case basis, incorporate additional methods, models, and lines of evidence that EPA finds technically appropriate for risk management objectives" (Section V, page 31 of EPA, 2004). This information, which represents the best scientific data available, is described in further detail below and in Appendix B. This information is also considered a refinement of the 10-20 µg/L range reported in the 2003 IRED (U.S. EPA, 2003a).

The Agency has selected an atrazine level of concern (LOC) in the 2003 IRED (U.S. EPA, 2003a and b) that is consistent with the approach described in the Office of Water's (OW) draft atrazine aquatic life criteria (U.S. EPA, 2003c). Through these previous analyses (U.S. EPA, 2003a, b, and c), which reflect the current best available information, predicted or monitored aqueous atrazine concentrations can be interpreted to determine if a water body is likely to be significantly affected via indirect effects to the aquatic community. Potential impacts of atrazine to plant community structure and function that are likely to result in indirect effects to the rest of the aquatic community, including the listed species, are evaluated as described below.

As described further in Appendix B, responses in microcosms and mesocosms exposed to atrazine were evaluated to differentiate no or slight, recoverable effects from significant, generally non-recoverable effects (U.S. EPA, 2003e). Because effects varied with exposure duration and magnitude, there was a need for methods to predict relative differences in effects for different types of exposures. The Comprehensive Aquatic Systems Model (CASM) (Bartell et al., 2000; Bartell et al., 1999; DeAngelis et al., 1989) was selected as an appropriate tool to predict these relative effects, and was configured to provide a simulation for the entire growing season of a 2nd and 3rd order Midwestern stream as a function of atrazine exposure. CASM simulations conducted for the concentration/duration exposure profiles of the micro- and mesocosm data showed that CASM seasonal output, represented as an aquatic plant community similarity index, correlated with the micro- and mesocosm effect scores, and that a 5% change in this index reasonably discriminated micro- and mesocosm responses with slight versus significant effects. The CASM-based index was assumed to be applicable to more diverse exposure conditions beyond those present in the micro- and mesocosm studies.

To avoid having to routinely run the CASM model, simulations were conducted for a variety of actual and synthetic atrazine chemographs to determine 14-, 30-, 60-, and 90-day average concentrations that discriminated among exposures that were unlikely to exceed the CASM-based index (i.e., 5% change in the index). It should be noted that the average 14-, 30-, 60-, and 90-day concentrations were originally intended to be used as screening values to trigger a CASM run (which is used as a tool to identify the 5% index change LOC), rather than actual thresholds to be used as an LOC (U.S. EPA, 2003e). The following threshold concentrations for atrazine were identified (U.S. EPA, 2003e):

- 14-day average = 38 μg/L
- 30-day average = 27 μg/L
- 60-day average = 18 μg/L
- 90-day average = 12 μg/L

Effects of atrazine on aquatic plant communities that have the potential to subsequently pose indirect effects to the listed species and their designated critical habitat are best addressed using the robust set of micro- and mesocosm studies available for atrazine and the associated risk estimation techniques (U.S. EPA, 2003a, b, c, and e). The 14-, 30-, 60-, and 90-day threshold concentrations developed by EPA (2003e) are used to evaluate potential indirect effects to aquatic communities for the purposes of this ESA. Use of these threshold concentrations is considered appropriate because: (1) the CASM-based index meets the goals of the defined assessment endpoints for this assessment; (2) the threshold concentrations provide a reasonable surrogate for the CASM index; and (3) the additional conservatism built into the threshold concentration, relative to the CASM-based index, is appropriate for an endangered species risk assessment (i.e., the threshold concentrations were set to be conservative, producing a low level (1%) of false negatives relative to false positives). Therefore, these threshold concentrations are used to identify potential indirect effects (via aquatic plant community structural change) to the listed species and their designated critical habitat. If modeled atrazine EECs exceed the 14-, 30-, 60- and 90-day threshold concentrations following refinements of potential atrazine concentrations with available monitoring data, the CASM model could be employed to further characterize the potential for indirect effects. A step-wise data evaluation scheme incorporating the use of the threshold concentrations is provided in Figure 4.2. Further information on threshold concentrations is provided in Appendix B.

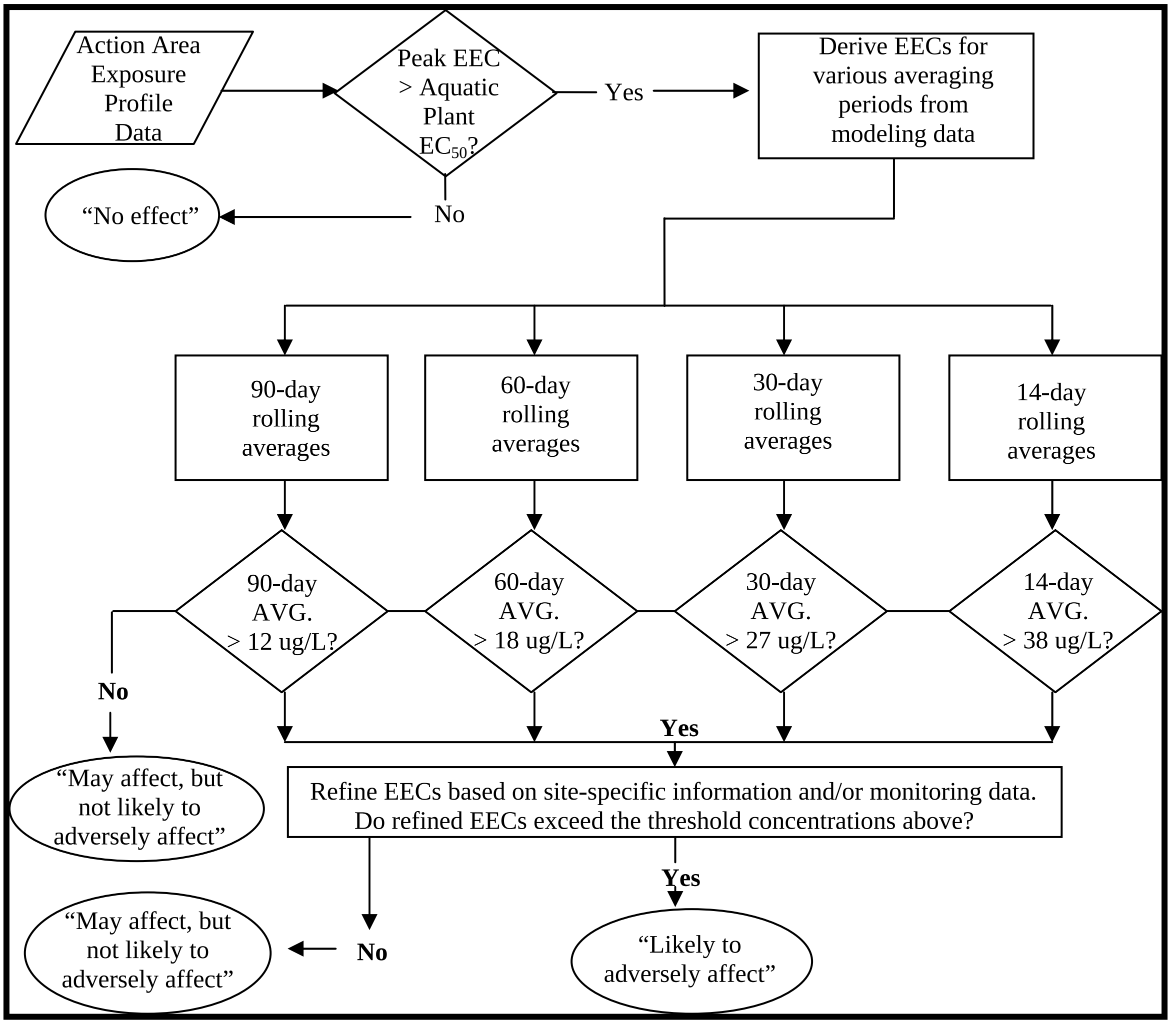


Figure 4.2 Use of Threshold Concentrations in Endangered Species Assessment

4.7. Use of Probit Slope Response Relationship to Provide Information on the Endangered Species Levels of Concern

The Agency uses the probit dose-response relationship as a tool for providing additional information on the potential for acute direct effects to individual listed species and aquatic animals that may indirectly affect the listed species of concern (U.S. EPA, 2004). As part of the risk characterization, an interpretation of acute RQ for listed species is discussed. This interpretation is presented in terms of the chance of an individual event (i.e., mortality or immobilization) should exposure at the EEC actually occur for a species with sensitivity to atrazine on par with the acute toxicity endpoint selected for RQ calculation. To accomplish this interpretation, the Agency uses the slope of the dose response relationship available from the toxicity study used to establish the acute toxicity measures of effect for each taxonomic group that is relevant to this assessment (i.e., freshwater fish used as a surrogate for aquatic-phase amphibians and freshwater invertebrates). The individual effects probability associated with the acute RQ is based on the mean estimate of the slope and an assumption of a probit dose response relationship. In addition to a single effects probability estimate based on the mean, upper and lower estimates of the effects probability are also provided to account for variance in the slope, if available. The upper and lower bounds of the effects probability are based on available information on the 95% confidence interval of the slope. A statement regarding the confidence in the estimated event probabilities is also included. Studies with good probit fit characteristics (i.e., statistically appropriate for the data set) are associated with a high degree of confidence. Conversely, a low degree of confidence is associated with data from studies that do not statistically support a probit dose response relationship. In addition, confidence in the data set may be reduced by high variance in the slope (i.e., large 95% confidence intervals), despite good probit fit characteristics.

Individual effect probabilities are calculated based on an Excel spreadsheet tool IECV1.1 (Individual Effect Chance Model Version 1.1) developed by the U.S. EPA, OPP, Environmental Fate and Effects Division (June 22, 2004). The model allows for such calculations by entering the mean slope estimate (and the 95% confidence bounds of that estimate) as the slope parameter for the spreadsheet. In addition, the acute RQ is entered as the desired threshold.

4.8. Incident Database Review

A number of incidents have been reported in which atrazine has been associated with some type of environmental effect. Incidents are maintained and catalogued by EFED in the Ecological Incident Information System (EIIS). Each incident is assigned a level of certainty from 0 (unrelated) to 4 (highly probable) that atrazine was a causal factor in the incident. As of the writing of this assessment, 358 incidents are in EIIS for atrazine spanning the years 1970 to 2005. Most (309/358, 86%) of the incidents involved damage to terrestrial plants, and most of the terrestrial plant incidences involved damage to crops treated directly with atrazine. Of the remaining 49 incidents, 47 involved aquatic animals and 2 involved birds. Because the species included in this effects determination are aquatic species, incidents involving aquatic animals assigned a certainty index of 2 (possible) or higher (N=33) were re-evaluated. Results are summarized below, and additional details are provided in Appendix E. The 33 aquatic incidents were divided into three categories:

1. Aquatic incidents in which atrazine concentrations were confirmed to be sufficient to either cause or contribute to the incident, including directly via toxic effects to aquatic organisms or indirectly via effects to aquatic plants, resulting in depleted oxygen levels;
2. Aquatic incidents in which insufficient information is available to conclude whether atrazine may have been a contributing factor – these may include incidents where there was a correlation between atrazine use and a fish kill, but the presence of atrazine in the affected water body was not confirmed; and
3. Aquatic incidents in which causes other than atrazine exposure are more plausible (e.g., presence of substance other than atrazine confirmed at toxic levels).

The presence of atrazine at levels thought to be sufficient to cause either direct or indirect effects was confirmed in 3 (9%) of the 33 aquatic incidents evaluated. Atrazine use was also correlated with 11 (33%) additional aquatic incidents where its presence in the affected water was not confirmed, but the timing of atrazine application was correlated with the incident. Therefore, a definitive causal relationship between atrazine use and the incident could not be established. The remaining 19 incidents (58%) were likely caused by some factor other than atrazine. Other causes primarily included the presence of other pesticides at levels known to be toxic to affected animals. Although atrazine use was likely associated with some of the reported incidents for aquatic animals, they are of limited utility to this assessment for the following reasons:

- No incidents in which atrazine is likely to have been a contributing factor have been reported after 1998. A number of label changes, including cancellation of certain uses, reduction in application rates, and harmonization across labels to require setbacks for applications near waterbodies, have occurred since that time. For example, several incidents occurred in ponds that are adjacent to treated fields. The current labels require a 66-foot buffer between application sites and water bodies.

- The habitat of the assessed species is not consistent with environments in which incidents have been reported. For example, no incidents in streams or rivers were reported.

Although the reported incidents suggest that high levels of atrazine may result in impacts to aquatic life in small ponds that are in close proximity to treated fields, the incidents are of limited utility to the current assessment. However, the lack of recently reported incidents in flowing waters does not indicate that effects have not occurred. Further information on the atrazine incidents and a summary of uncertainties associated with all reported incidents are provided in Appendix E.

5.0 Risk Characterization

Risk was estimated by calculating the ratio of the EEC to the appropriate toxicity endpoint as outlined in U.S. EPA (2004). The resulting value is the risk quotient (RQ), which is then compared to pre-established levels of concern (LOC) for each category evaluated (Appendix F). The highest EEC and most sensitive acute and chronic toxicity endpoints from laboratory studies were used to determine the screening level RQ. However, exceedance of one or more LOC does not necessarily result in a "likely to adversely effect" determination. In cases where RQs exceed one or more of the established LOCs, additional factors including biological and ecological factors of the assessed species and additional characterization of potential exposures were used to characterize the potential for atrazine to affect the assessed species. RQs were initially calculated for the use that resulted in the highest EEC (sorghum); other uses were evaluated if RQs based on the highest EEC result in a "likely to adversely affect" determination.

Potential direct effects to the six listed species from use of atrazine in the action area are evaluated in Section 5.1. Potential indirect effects to the assessed species from direct effects to animals and plants are evaluated in Section 5.2. The risk characterization approach used in this assessment to evaluate direct and indirect effects to listed species is endorsed by the Services (USFWS/NMFS, 2004b).

As previously discussed in the effects assessment, the toxicity of the atrazine degradates has been shown to be less than the parent compound based on the available toxicity data for freshwater fish, invertebrates, and aquatic plants; therefore, the focus of the risk characterization is parent atrazine (i.e., RQ values were not derived for the degradates).

5.1. Direct Effects Assessment

5.1.1. Risk Estimation

5.1.1.1. Shortnose Sturgeon

RQs used to estimate direct effects to the shortnose sturgeon are in Table 5.1. These RQs are further characterized in Section 5.1.2.

Table 5.1. Summary of Aquatic RQs to Assess Potential Direct Effects to the Shortnose Sturgeon.						
Effect	**Surrogate Species**	**Toxicity Value (µg/L)**	**EEC (µg/L)**	**RQ**	**Probability of Individual Effect[a,b]**	**LOC Exceedance**
Direct Acute Effects to Shortnose Sturgeon	Sheepshead Minnow	LC50: 2000	Peak: 55	0.028	1 in $2x10^{11}$	None
Direct Chronic Effects to Shortnose Sturgeon	Brook Trout	NOAEC: 65	60-Day: 54	0.83	Not calculated for chronic endpoints	None

a No slope was available for the most sensitive study; therefore, a slope of 4.4 (95% CI of 2.8 – 5.9) was used from a different acute study in sheepshead minnows of equivalent duration (MRID 43344901).
b Based on the 95% CI on the slope, the probability of an individual effects would be from 1 in 146,000 to 1 in $4x10^{19}$.

5.1.1.2. Dwarf Wedgemussel

RQs used to estimate direct effects to the dwarf wedgemussel are in Table 5.2 below. RQs are further interpreted in the risk description, Section 5.1.2.

Table 5.2. Summary of RQs Used to Estimate Direct Effects to the Dwarf Wedgemussel.						
Effect	**Surrogate Species**	**Toxicity Value µg/L**	**EEC µg/L**	**RQ**	**Probability of individual Effect**	**LOC Exceedance**
Direct Acute Effects to Dwarf Wedgemussel	Eastern oyster	**EC50:** >1700 µg/L[a]	Peak: 55	<0.032	<1 in $4x10^{10}$[c]	None
Direct Chronic Effects to Dwarf Wedgemussel	Scud	**NOAEC**: 60 µg/L[b]	21-Day: 55	0.92	Not calculated for chronic endpoints	None

[a] Two studies in the Eastern Oyster were located. No treatment-related effects were observed in either study. Therefore, the study that tested the highest concentration (1700 µg/L) was used to estimate risk.
[b] A chronic study in mussels was not located for this assessment; therefore, the most sensitive chronic invertebrate NOAEC was used. It is uncertain if use of scud as a surrogate species results in an under or overestimation of risk. However, toxicity studies in invertebrates that are closer in taxonomy to mussels (e.g., snail and leech; Section 5.1.2.) suggest that use of the scud as a surrogate species is protective.
[c] The probability of an individual effect was calculated using a probit dose response slope of 4.4; this is the only slope for technical grade atrazine reported in available ecotoxicity data for freshwater invertebrates (MRID 45202917, scud). Use of a slope of 4.4 results in a more conservative estimation of the probability of an individual effect than the default slope recommended in U.S. EPA (2004a) of 4.5.

5.1.1.3. Sea Turtles

RQs used to estimate potential direct effects to sea turtles are provided in Table 5.3 below. These RQs are further characterized in Section 5.1.2.

Table 5.3. Summary of RQs Used to Assess Potential Direct Effects to Sea Turtles.[a]						
Effect	**Surrogate Species**	**Toxicity Value**	**EEC**	**RQ**	**Probability of Individual Effect**	**LOC Exceedance**
Direct Acute Effects to Sea Turtles	Bobwhite quail and mallard duck	Dietary LC50: >5000 ppm	0.47 ppm	<0.01	Not calculated; sufficient dose-response not available.	None
		LD50: 940 mg/kg-bw Probit slope: 3.8 (95% CI: 2.0 – 5.7)	0.48 mg/kg-bw[b]	<0.01	<1 in 1,000,000 (95% CI: <1 in 1,000,000)	
Direct Chronic Effects to Sea Turtles	Bobwhite quail and mallard duck	Dietary NOAEC: 225 ppm	0.47 ppm	<0.01	Not estimated for chronic endpoints	None

[a] Sea turtles include green, loggerhead, leatherback, and Kemp's ridley sea turtles.

[b] The dose-based EEC represents addition of the dietary EEC (0.46 mg/kg-bw) + the water flow through EEC (0.01 mg/kg-bw) as presented in Section 3.

5.1.2. Risk Description, Direct Effects

5.1.2.1. Shortnose Sturgeon

RQs were derived using standard laboratory studies and PRZM/EXAMS estimated standard water body EECs. No acute or chronic concern levels were exceeded for direct effects to fish. The highest acute RQ for fish was 0.028. At this RQ, the estimated probability of an individual effect (i.e., mortality) would be 1 in $2x10^{11}$. This analysis is based on an assumption of a probit dose response relationship with an estimated slope of 4.4 for sheepshead minnows (MRID 43344901). The acute LC50 for sheepshead minnows was 2000 µg/L. It is recognized that extrapolation of very low probability events is associated with considerable uncertainty in the resulting estimates. In order to explore the possible bounds to such estimates, the upper and lower 95% confidence limits of 2.8 to 5.9 were used to calculate upper and lower estimates of the effects probability associated with the acute RQ. Probability of an individual effect based on the upper and lower confidence intervals are 1 in 146,000 to 1 in $4x10^{19}$. Based on the lack of acute and chronic LOC exceedance and the low probability of an individual mortality, atrazine is not likely to cause direct adverse effects to the shortnose sturgeon.

The highest chronic RQ was 0.83 based on a 60-day EEC of 54 µg/L and a NOAEC of 65 µg/L in brook trout. Although an RQ of 0.83 approaches the chronic LOC of 1.0, the exposure value used in the RQ calculation is expected to produce a conservative measure of exposure for habitats of the shortnose sturgeon (major rivers, river mouths). The EECs used to derive chronic RQs were estimated using PRZM/EXAMS EECs, which is based on a standard water body scenario. Additional modeling and the available monitoring data presented in Section 3 collectively suggest that long-term EECs used to derive RQs for locations of the shortnose sturgeon (major river systems and river mouths) are expected to be considerably lower than 54 µg/L.

As discussed in Section 4, several open literature studies raise questions about sublethal effects of atrazine on plasma steroid levels, behavior modifications, gill physiology, and endocrine-mediated functions in freshwater fish and anadromous fish. Consideration of the sublethal data indicates that effects associated with alteration of gill physiology and endocrine-mediated olfactory functions may occur in salmon at atrazine concentrations lower than the lowest NOAEC reported from submitted life-cycle studies (Waring and Moore, 2004; Moore and Lower, 2001). However, there are a number of factors that limit the utility of these studies for this assessment, which are addressed in detail in Sections A.2.4 of Appendix A. For example, Moore and Lower (2001) measured olfaction responses in exposed epithelial tissue (after removal of skin and cartilage) and not intact fish to atrazine, and potential solvent effects could not be reconciled (i.e., no negative (solvent free dilution water) control was tested). Furthermore, no quantitative relationship is established between reduced olfactory response (measured as electrophysiological response) of male epithelial tissue to the female priming hormone in the laboratory and reduction in salmon reproduction (i.e., the ability of male salmon to recognize and mate with ovulating females). Also, Waring and Moore (2004) evaluated survival of salmon in full salinity seawater after atrazine exposure in freshwater. However, the relevance of direct transfer from freshwater to full-salinity seawater to the assessed species is questionable given the habitats of the assessed species (Appendix D). Other sublethal effects observed in fish studies have included behavioral modifications, alterations of plasma steroid levels, and changes in kidney histology at atrazine concentrations ranging from 5 to 35 µg/L (see Section 4). However, a number of uncertainties were also identified with each of the studies, which are discussed in Section A.2.4 of Appendix A.

In summary, it is not possible to quantitatively link the sublethal effects to the selected assessment endpoints for the assessed listed species (i.e., survival, growth, and reproduction of individuals). Also, effects to reproduction, growth, and survival were not observed in the four submitted fish life-cycle studies at levels that produced the reported sublethal effects (Appendix A). In addition, there are a number of factors in the design of these studies, which are addressed in detail in Sections A.2.4a and A.2.4b of Appendix A, that preclude quantitative use of the data in risk assessment.

Based on the lack of LOC exceedance for acute and chronic effects to the most sensitive fish species tested in acute and life-cycle studies and PRZM/EXAMS standard water body, the best available information suggests that atrazine use in the Chesapeake Bay watershed will have “no effect” on the shortnose sturgeon via direct effects.

5.1.2.2. Dwarf Wedgemussel

No acute or chronic LOCs were exceeded for the dwarf wedgemussel. The acute RQ is based on an EC50 of >1700 µg/L. No effects were observed in this study resulting in an acute RQ of <0.032. The probability of individual effect could not be calculated based on the dose-response from this study because no effects were observed in the acute toxicity study used in RQ calculation. Therefore, the probability of an individual effect was calculated using a probit dose response slope of 4.4; this is the only slope for technical grade atrazine reported in available ecotoxicity data for freshwater invertebrates (MRID 45202917, scud). Use of a probit slope of 4.4 results in a more conservative estimation of the probability of an individual effect than the default slope recommended in U.S. EPA (2004a) of 4.5. Based on a probit slope of 4.4, the probability of an individual mortality at an RQ of <0.032 is <approximately 1 in $4x10^{10}$. In addition, data located in the open literature (Johnson *et al.*, 1993) suggest use of the saltwater Eastern Oyster as a surrogate for the dwarf wedgemussel (a freshwater mussel) in risk estimation was protective because the EC50 in the only freshwater mussel tested (*A. imbecilis*) of >60,000 µg/L is considerably higher than the EC50 used in RQ derivation of 17,000 ug/L in Eastern Oysters. Based on the lack of acute and chronic LOC exceedance, a "no effect" determination is made for potential direct effects to dwarf wedgemussels.

Chronic toxicity data in mussels were not located for use in this assessment; therefore, the most sensitive chronic invertebrate NOAEC was used (60 µg/L in the scud, MRID 00024377) to derive RQs. It is uncertain if use of the scud as a surrogate species results in an under or overestimation of risk. However, acute toxicity studies in invertebrates that are closer in taxonomy than the scud (phylum arthropoda) to the dwarf wedgemussel (i.e., snail and leech; phylum mollusca) suggest that use of the scud as a surrogate species is likely protective of acute effects to the dwarf wedgemussel. Acute LC50 values in both the snail and leech are >16,000 µg/L (although effects occurred in the leech study after approximately 30 days, see Appendix A), compared with scud LC50s, which range from 5700 µg/L (MRID 00024377) to 15,000 µg/L (MRID 45202917). The slightly lower LC50s in the scud, compared with the snail and leech, suggest that use of a chronic NOAEC in the scud is unlikely to result in an underestimation of risk to the dwarf wedgemussel.

In addition, as previously discussed (Section 3), EECs used to derive RQs likely overestimate potential long-term exposures to the dwarf wedgemussel. Incorporation of location-specific factors into modeling including representative flow rate from water bodies where the mussels are expected to occur (Section 3.3.) together with monitoring data from the Chesapeake Bay tributaries (Section 3.4) suggest that longer term EECs (days to weeks) are expected to be considerably lower than the modeled values using the standard water body and are likely in the low µg/L range.

Based on the lack of LOC exceedances for acute effects to the Eastern oyster, chronic effects to the most sensitive invertebrate species tested, and EECs derived from the PRZM/EXAMS standard water body, the best available information suggests that atrazine use in the Chesapeake Bay watershed will have "no effect" on the dwarf wedgemussel via direct effects.

5.1.2.3. Sea Turtles

No acute or chronic LOCs are exceeded. All acute RQs are less than 0.01. Although there is uncertainty associated with the RQs, the methods used to derive surrogate effects endpoints for reptiles and derivation of EECs for sea turtles is considered protective (conservative). Key uncertainties are as follows:

1. Dietary exposure to turtles was estimated using the highest BCF of 8.5, which was reported in fathead minnows and the peak PRZM/EXAMS EEC of 55 µg/L from the standard water body. Use of a fish BCF to estimate concentrations in turtle dietary items such as aquatic invertebrates may result in an over or under estimation of atrazine intake

by sea turtles. Bioconcentration of neutral organic chemicals such as atrazine is typically influenced by lipid content of the bioaccumulating organism. Therefore, food items with lipid content higher than the fish species used in the BCF study may accumulate more atrazine, resulting in an underestimation of exposure. Conversely, organisms with lipid contents lower than the fish species used in the BCF study may accumulate less atrazine, resulting in an underestimation of exposure. However, given the low magnitude of RQs, variations in lipid content of dietary items relative to fathead minnows are not expected to alter the conclusions of this assessment. In addition, the resulting dietary EEC is considered conservative because dietary EECs for food items were derived based on an assumption of continuous exposure to atrazine at 55 µg/L. As discussed in Section 3, short-term peak atrazine concentrations are expected to be similar to the 55 µg/L estimate; however, longer term atrazine concentrations at locations within the Bay where turtles are expected to feed are estimated to be considerably lower than 55 µg/L.

2. Water intake data used to estimate EECs were available for only two of the four turtle species, and food intake levels were only available for one of the four species assessed. An assumption was made in this assessment that the food and water intake values are representative of all four turtle species. Given the low magnitude of the RQs and the conservative nature of the EECs used to derive RQs, differences in food or water intake levels across the four turtle species are not to likely impact conclusions of this assessment.

3. Ecotoxicity data from birds were used as surrogates for turtles. Use of birds as a surrogate species for reptiles is considered protective, as discussed in Section 4.

4. One degradate, DEA, was found to be approximately as toxic as atrazine to birds on an acute oral basis. DACT has been shown to be of equivalent acute toxicity compared with atrazine in mammals; however, DACT has not been tested in birds. However, both of these degradates are expected to form a maximum of 18% of atrazine in the environment (See Section 2). Even if both degradates of concern were found at concentrations equal to atrazine (55 µg/L atrazine, 55 µg/L DEA, and 55 µg/L DACT) and assuming equivalent toxicity for all three degradates, acute and chronic RQs would remain lower than 0.01, which is well below the level of concern (EEC = 1.4 ppm; LC50 = 940 ppm; NOAEC = 225 ppm). The assumption that all three compounds are present at concentrations equal to the peak atrazine concentration is conservative given that the degradates are expected to have similar physicochemical properties and have been shown to form no more than 18% of atrazine in available degradation studies. Therefore, this analysis suggests that quantification of exposure to degradates would have negligible impact on this assessment.

5. A NOAEC of 225 ppm was used for the chronic RQ calculation. However, reduced body weight gain was observed at 75 ppm (the lowest concentration tested; MRID 42547102). As discussed in Section 4, the reproduction NOAEC of 225 ppm was considered more biologically relevant for this assessment. However, use of 75 ppm as the NOAEC in RQ calculations would not change the risk conclusions, and chronic RQs would remain < 0.01, well below the LOC.

Based on an assumption of a probit dose response relationship with an estimated slope of 3.8, 95% confidence intervals of 2.0 to 5.7 (MRID 00024721), and RQs presented in Table 5.3, probability of an individual effect based on the slope and the 95% confidence intervals would be <1,000,000.

Based on the lack of LOC exceedances and the conservative assumptions of exposure to sea turtles, the best available information suggests that atrazine use in the Chesapeake Bay watershed will have “no effect” to any of the four sea turtle species assessed via direct effects.

5.1.3. Summary of Direct Effects Conclusions.

Table 5.4. Summary of Direct Effects Determinations to the Six Assessed Listed Species.		
Species	**Direct Effects Conclusion**	**Basis for Conclusion**
Dwarf wedgemussel	“No effect”	No acute or chronic LOCs are exceeded
Shortnose sturgeon	“No effect”	No acute or chronic LOCs are exceeded
Four sea turtle species	“No effect”	No acute or chronic LOCs are exceeded

5.2. Indirect Effects

Pesticides have the potential to exert indirect effects upon the listed organisms by inducing changes in structural or functional characteristics of affected communities (U.S. EPA, 2004). For example, perturbation of forage or prey availability and alteration the extent and nature of nesting habitat are examples of indirect effects.

In conducting a screen for indirect effects, the direct effects LOCs for each taxonomic group are used to make inferences concerning the potential for indirect effects upon listed species that rely upon non-endangered organisms in these taxonomic groups as resources critical to their life cycle (U.S. EPA, 2004). If no direct effect RQs exceed any LOCs for a taxonomic group (presented in Section 5.1), then the concern for indirect effects to the assessed species that rely on the taxonomic group is presumed to be lower than LOCs. If direct effects LOCs are exceeded, then further analysis on the potential for indirect effects to occur depends on the taxa for which LOCs were exceeded as described below.

When LOCs are exceeded for animals that may be food items of the six assessed animals, there is a potential for atrazine to indirectly affect the assessed animals by reducing available food supply. In such cases, the dose-response relationship from the toxicity study used for calculating the RQ of the surrogate prey item is evaluated to estimate the probability of acute effects associated with an exposure equivalent to the EEC. The greater the probability that exposures will produce effects on a taxa, the greater the concern for potential indirect effects for listed species dependant upon that taxa. Indirect effects RQs were initially calculated using PRZM/EXAMS EEC and the most sensitive laboratory studies for the broad taxonomic groups outlined in Section 4. Therefore, when direct effects LOCs were exceeded for food items, additional analysis was conducted to allow for a determination of potential effects to dietary items more relevant to the assessed species, and potential exposures were further characterized for waters more reflective of habitats of the assessed species.

When LOCs are exceeded for plants, then the potential exists for indirect effects to occur from reduction in food source or habitat alteration. The initial plant LOCs were interpreted using the following (U.S. EPA, 2004):

- plant RQ < endangered species LOC: a no effect determination to listed species that rely either on a specific plant species (plant species obligate) or multiple plant species (plant dependant) for some important aspect of their life cycle are not expected;
- plant RQ > endangered species LOC and < non-endangered species LOC: a no effect determination is made for listed species that rely on multiple plant species to successfully complete their life cycle (plant dependent species);
- plant RQ > non-endangered species LOC: potential for adverse effects to listed species that rely either on a specific plant species (plant species obligate) or multiple plant species (plant dependant) for some important aspect of their life cycle.

If aquatic plant LOCs are exceeded, further evaluation is conducted to determine whether effects to plants are likely to result in adverse effects to the assessed species. Further evaluation included analyses of the geographical and temporal nature of the exposure and characterization of the biological and ecological requirements of potentially impacted listed species.

A summary of the methods used to evaluate the potential for atrazine to adversely affect the six assessed species via indirect effects from potential adverse effects to animals and plants is discussed below. Methods are consistent with those presented in U.S. EPA (2004).

- Potential indirect effects on the assessed species from direct effects on animal food items were evaluated by considering the diet of the assessed species, the potential magnitude of effect to dietary species, and the potential number of species affected relative to the number of species that serve as dietary items.

- Potential indirect effects on the assessed species from effects on habitat and/or primary productivity were assessed using RQs based on standard water body EECs and the most sensitive aquatic plant EC50 as a screen. If aquatic plant RQs exceed any LOC, potential community level effects were evaluated using community-level effect threshold concentrations, as described in Section 4.6; if standard water body EECs do not exceed the threshold concentrations, a "may effect, but not likely to adversely affect" determination is made. However, if EECs based on the standard water body exceed the threshold concentrations, further characterization of the EECs is performed using additional modeling and monitoring information for locations where the species is expected to occur.

- The potential for atrazine to affect the assessed species indirectly by affecting riparian zones in the Chesapeake Bay watershed and areas adjacent to the habitat or spawning areas of the assessed species is evaluated using submitted terrestrial plant toxicity data and preliminary studies on woody plants. If terrestrial plant RQs exceed the LOC for direct effects to non-endangered species based on EECs derived using Terrplant (Version 1.2.1) and submitted guideline terrestrial plant toxicity data, a conclusion that atrazine may affect the assessed species is made. Further analysis of the potential for atrazine to affect the assessed species via reduction in riparian habitat includes an evaluation of the magnitude of the potential effect to riparian habitat, type of riparian area most vulnerable to atrazine, and relevance of sensitive riparian zones to water quality in the Chesapeake Bay watershed.

5.2.1. Summary of Biological and Ecological Information Used to Evaluate Potential Indirect Effects of Atrazine

Location and dietary information on the shortnose sturgeon, dwarf wedgemussel, and the four sea turtles used to perform the indirect effects assessment are summarized below. These data are summaries of information provided in Section 2.2 and Appendix D. Additional information can be obtained from those sections.

5.2.1.1. Shortnose Sturgeon

Data for the shortnose sturgeon were primarily obtained from NMFS (1998); U.S. EPA (2003b); and Gilbert, 1989).

Diet: Shortnose sturgeon are non-selective continuous benthic omnivores. Dietary items consist of insect larvae, worms, and mollusks; however, the dietary preferences appear to change with age. Insect larvae (e.g. *Hexagenia*, *Chaobrus*, and *Chironomus*), and small crustaceans (e.g. *Gammarus*, *Asellus*, and *Cyathura*) are the predominate food items for juveniles (NMFS, 1998). Adults feed primarily on small mollusks. In freshwater, these mollusks include *Physa*, *Helisoma*, *Corbicula*, *Amnicola*, *Valvata*, *Pisidium*, and small *Elliptio* (NMFS, 1998). In saline areas, molluscan prey include small *Mya* and *Macoma*. Recent data show that adult sturgeon feed on gammarid amphipods and zebra mussels (NMFS, 1998). Juveniles are located in freshwater systems from hatching until adulthood (approximately 3 to 7 years); therefore, only toxicity data in freshwater organisms were used to assess potential effects to juvenile sturgeon prey items. Both marine and freshwater toxicity data were used to assess potential effects to adult food items because they may reside in freshwater or low salinity areas of the Chesapeake Bay.

Habitat: Shortnose sturgeon are found primarily in the main stems of larger rivers and river mouths and the Chesapeake Bay. They reproduce in deeper freshwater rivers with a swift current and remain in this environment until maturity, when they migrate to mouths of rivers with slow to no current.

Reproduction: Shortnose sturgeon depend on free-flowing rivers and seasonal floods to provide suitable spawning habitat. For shortnose sturgeon, spawning grounds have been found to consist mainly of gravel or rubble substrate in regions of fast flow. Flowing water provides oxygen, allows for the dispersal of eggs, and assists in excluding predators. Seasonal floods scour substrates free of sand and silt, which might suffocate eggs (U.S. EPA, 2003b).

Shortnose sturgeon spawn in upper, freshwater sections of rivers and feed and overwinter in both fresh and saline habitats. In populations that have free access to the total length of a river (absent of dams), spawning areas are located at the farthest accessible upstream reach of the river, often just below the fall line (U.S. EPA, 2003b). Tributaries of the Chesapeake Bay that appear to have suitable spawning habitat for the shortnose sturgeon include the Potomac, Rappahannock, James, York, Susquehanna, Gunpowder and Patuxent rivers (U.S. EPA, 2003b). Other scientists believe that very little, if any, suitable spawning habitat remains for shortnose sturgeon, due to past sedimentation in tidal freshwater spawning reaches (U.S. EPA, 2003b).

Table 5.5. Summary of Shortnose Sturgeon Dietary Items

Life Stage	Location	Dietary items	Examples	Surrogate Species with Toxicity Data	Range of Toxicity Values (µg/L)
Juveniles	Freshwater Rivers	Insect larvae	*Hexagenia*, *Chaobrus*, and *Chironomus*	Chironomus, stonefly	Chironomus LC50: 720 - >33,000 Stonefly: 6700
		Small crustaceans	*Gammarus*, *Asellus*, and *Cyathura*	Scud, waterflea	Scud LC50: 4700 to 15,000 Waterflea EC50: 3500 to >30,000
Adults	Low salinity areas including river mouths	Freshwater Mollusks	*Physa*, *Helisoma*, *Corbicula*, *Amnicola*, *Valvata*, *Pisidium*, and small *Elliptio*.	Snail (*Ancylus fluviatilis*), leech	Acute Snail and Leech: LC50: >16,000
		Saltwater Mollusks	Small *Mya* (a soft shelled clam) and *Macoma* (a clam)	Eastern Oyster	EC50: >1700

5.2.1.2. Dwarf Wedgemussel

Primary data sources on the dwarf wedgemussel include USFWS (1993) and U.S. EPA (2003b).

Diet: Little specific information is available on the food items of the dwarf wedgemussel. In general, mussels are filter-feeders that feed on detritus (dead organic matter), zooplankton, and phytoplankton. Phytoplankton is typically considered to be the most important food item for sustenance of mussels. Waterfleas and algae were used as surrogate food species for freshwater invertebrates and zooplankton, respectively.

Habitat: The dwarf wedgemussel is a freshwater mussel that lives on muddy sand, sand, and gravel bottoms in creeks and rivers of varying sizes. Its habitat is also characterized by slow to moderate current with little silt deposition.

Several locations within the Chesapeake Bay watershed are known for the dwarf wedgemussels. These locations were described in Section 2 (Table 2.3) and include waters in the Potomac River drainage (McIntosh Run, Nanjemoy Creek, and Aquia Creek), the York River drainage (South Anna and Po Rivers), the Tuckahoe Creek Drainage (Norwich Creek, Long Marsh Ditch, Mason’s Branch), the Southeast Creek watershed (Browns branch, Granny Finley, Corsica River tributary, and Southeast Creek tributary), and the Rappahannock River drainage (Rappahannock River and Carter Run). All locations are small, flowing streams that are consistent with the description of headwater streams or mid-level reaches presented in Section 2.4.

5.2.1.3. Sea turtles

A summary of the information on sea turtle’s diet and habitat is presented in this section. These data are a summary of information presented in Section 2.2 and Appendix D and were obtained from a number of sources including the recovery plans for the four sea turtle species and information from the Virginia Institute of Marine Science (VIMS), the Chesapeake Bay Program, and Fish and Wildlife Services. Specific references are presented in Section 2.2 and in Appendix D.

Diet: The diet of each sea turtle is described in detail in Appendix D and summarized in Table 5.6. Loggerhead and Kemp’s ridley sea turtles eat food items including crustaceans, plants, mollusks, other invertebrates, and fish. Leatherback turtles consume mainly jelly fish, but also ingest other invertebrates. Green turtles, however, eat invertebrates as hatchlings, but feed almost exclusively on aquatic plants (i.e., sea grasses) as adults.

Table 5.6 Summary of Dietary Items of the Assessed Sea Turtles			
Species	**Food Item (taxa)**	**Food Item (common name)**	**Surrogate Species with Toxicity Data**
Loggerhead and Kemp's ridley	Chelicerate	Horseshoe crab	Mud crab Brown shrimp Grass shrimp Pink shrimp
	Crustacean	Blue crab	
		Hermit crab	
		Mantis shrimp	
	Plant	Eelgrass	Aquatic plant EC50; Community level effects thresholds
		Widgeon grass	
	Invertebrate	Jellyfish	No adequate surrogate species; Most sensitive species (copepod) was used for initial screen. [a]
		Other invertebrates including sea urchins	
	Mollusk	Oysters and clams	Eastern oyster
		Other bivalves	
	Fish	Various Species	Sheepshead minnow Brook trout
		Atlantic menhaden	
		Spot	
		Atlantic croaker	
		Bluefish	
		Striped bass	
		Oyster toadfish	
Leatherback	Invertebrates	Pink comb (jellyfish)	No adequate surrogate species; Most sensitive species (copepod) was used for initial screen.[a]
		Sea walnut (jellyfish)	
		Other jellyfish	
		Other invertebrates	
Green, Adults	Plants	Algae	Aquatic plant EC50; community level effects thresholds
		Sea grass	
Green, Juveniles	Invertebrates	Variety of invertebrates	Most sensitive species (copepod) was used for initial screen.[a]

[a] Distribution of toxicity values in aquatic invertebrates was used to characterize potential risks if LOCs were exceeded based on use of the copepod LC50 in RQ calculations.

Habitat: A summary of the expected habitats of the four sea turtles included in this assessment is in Section 2. Generally, these turtles are expected to be found in the main portion of the Bay, but may also be found in main stems of major rivers, river mouths, and estuarine inlets. A more thorough description of the expected locations of these turtles is in Appendix D.

5.2.2. Evaluation of the Potential for Atrazine to Induce Indirect Effects on the Shortnose Sturgeon, Dwarf Wedgemussel, and Sea Turtles from Reduction in Animal Food Items

5.2.2.1. Potential Effects to Aquatic Invertebrate Food Items

Potential effects to the six assessed species from reduction in food (aquatic animals) availability are presented below. As shown in Table 5.7, aquatic animals consumed by the six species assessed include fish, a variety of aquatic invertebrates, and zooplankton. However, an indirect effects assessment was not performed from direct acute or chronic effects to fish or chronic effects to aquatic invertebrates, because the direct effects assessment presented in Section 5.1 was conducted using all available toxicity data for these endpoints, and a concern for direct effects was not identified. Therefore, indirect effects from potential direct effects to these surrogate species are also presumably lower than LOCs. Potential indirect effects to the six assessed listed species from potential effects to aquatic plants are evaluated in Section 5.2.4.

Table 5.7. Summary of Animal Prey Items of the Six Assessed Species

Species	Life Stage	Location	Animal Dietary Items	Surrogate Species with Available Toxicity Data[a]
Shortnose sturgeon	Juveniles	Freshwater rivers	Insect larvae	Chironomus, stonefly
			Small crustaceans	Scud, waterflea
	Adults	River mouths, main stem of the Chesapeake Bay, and large rivers	Freshwater mollusks	Snail, leech
			Saltwater mollusks	Eastern oyster
Dwarf wedgemussel	All[b]	Freshwater headwater streams and mid-level reaches	Freshwater zooplankton	Daphnia
Sea turtles	All	Main stem of the Chesapeake Bay, major rivers, river mouths, estuarine inlets	Crustaceans, mollusks, other invertebrates, and fish	Brown shrimp, pink shrimp, grass shrimp, mud crabs, oysters, fish

[a] Copepods, the most sensitive aquatic invertebrate tested, were not considered an appropriate surrogate food item for any of the assessed species. However, the lowest copepod LC50 was used for risk estimation because copepods were the most sensitive invertebrate species tested as outlined in U.S. EPA (2004). Toxicity data on more appropriate dietary invertebrate species were used to further characterize potential risks to the assessed species if RQs based on the lowest copepod LC50 exceed acute LOCs.

[b] Glochidial stage receives sustenance from fish. No direct effects RQs exceeded LOCs for fish; therefore, indirect effects to glochidial stage mussels from potential effects to fish are also presumably lower than LOCs.

RQs initially used to screen whether atrazine may indirectly affect the six listed species considered in this assessment via reduction in available prey were based on acute ecotoxicity data from the most sensitive invertebrates tested and the PRZM/EXAMS estimated peak EEC of 55 µg/L in the standard water body. Results of this analysis are presented in Table 5.8.

Table 5.8. Summary of RQs Used to Estimated Indirect Effect to Shortnose Sturgeon and Sea Turtles via Potential Direct Effects on Dietary Items

Surrogate Food Item	Assessed Species for Indirect Effects	Toxicity Value (µg/L)	EEC (µg/L)	RQ	Probability of individual Effect[a]	Risk Interpretation
Aquatic invertebrates, Copepod	Shortnose sturgeon; Sea turtles	LC50: 88	55	0.62	1 in 2	Availability of aquatic invertebrate food items may be affected by atrazine use.

a Probability based on the assumption of a probit dose-response relationship and an estimated slope of 0.95 (MRID 45202918)

Table 5.8 indicates that atrazine may affect the shortnose sturgeon and sea turtles via potential direct effects on sensitive aquatic invertebrate food items. However, this analysis was based only on the most sensitive aquatic invertebrate species in laboratory studies and did not consider specific dietary characteristics of the assessed species or the expected location of the species within the Bay and its source waters. Therefore, additional characterization of the potential for atrazine to affect aquatic invertebrate food items of the six assessed species is presented below.

The potential for atrazine to elicit indirect effects via effects on food items is dependent on several factors including: (1) the potential magnitude of effect on invertebrate populations; and (2) the number of prey species potentially affected relative to the number of prey species needed for sustenance. Together, these data provide a basis to evaluate whether a sufficient number of individuals within a prey species and the number of prey species may be reduced such that an adverse effect to the shortnose sturgeon, dwarf wedgemussel, or sea turtles is likely, unlikely, or unable to be determined. Therefore, the sensitivity of all aquatic invertebrates to atrazine and the types of organisms consumed by the assessed listed species were considered.

Table 5.9 below presents RQs for surrogate food items that are representative of dietary items of the assessed species. The sensitivity of all aquatic invertebrates tested to atrazine is represented in Figure 5-1. This analysis considers only acute risk to aquatic invertebrate food items. Acute and chronic RQs for fish and chronic RQs for invertebrates were lower than LOCs for direct effects; therefore, potential indirect effects to listed species from direct effects on these endpoints were presumably lower than LOCs. Although marine copepods were the most sensitive aquatic invertebrate tested, more suitable surrogate food items for the assessed species were used in the refined analyses (Table 5.9).

Table 5.9. Summary of RQs Used to Assess Potential Risk to Animal Food Items of the Shortnose Sturgeon, Dwarf Wedgemussel, and Four Sea Turtles.					
Surrogate Food Item Species	**Listed Species That Receive Sustenance From Food Item Or Similar Food Item**	**Acute Toxicity Value Range (No. of Studies)**	**RQ Range Based on an EEC of 55 (µg/L)**	**Probability of Individual Effect [a]**	**Risk Interpretation**
Midge	Juvenile shortnose sturgeon	720 - >33,000 (3)	<0.01 - 0.076	<1 in 1,000,000[a]	Based on LOC exceedance, atrazine may affect food items that are as sensitive as the midge; however, the magnitude of potential effects on food availability (<1 in 1,000,000) is not likely sufficient to induce indirect effects to juvenile shortnose sturgeon.
Brown shrimp	Sea turtles	1000 (1)	0.055	<1 in 1,000,000[c]	Based on LOC exceedance, atrazine may affect food items that are as sensitive as the brown shrimp; however, the magnitude of potential effects on food availability (<1 in 1,000,000) would not likely be sufficient to induce indirect effects to sea turtles.
Mysid shrimp	Juvenile shortnose sturgeon	1000 – 5400 (2)	0.01 – 0.055	<1 in 1,000,000[c]	Based on LOC exceedance, atrazine may affect food items that are as sensitive as the mysid shrimp; however, the low probability of an individual effect suggests that the magnitude of potential effects would not likely be sufficient to induce indirect effects to predators.
Waterflea	Dwarf wedgemussel; juvenile shortnose sturgeon	3500 - >30,000 (5)	0.02	<1 in 1,000,000[a]	Based on low probability of individual effects and lack of LOC exceedance, atrazine is not likely to affect food items that are as sensitive as scud or waterfleas to the extent that indirect effects on predators are expected
Scud	Juvenile shortnose sturgeon	5700 – 15,000 (3)	0.01	<1 in 1,000,000	

Table 5.9. Summary of RQs Used to Assess Potential Risk to Animal Food Items of the Shortnose Sturgeon, Dwarf Wedgemussel, and Four Sea Turtles.					
Surrogate Food Item Species	**Listed Species That Receive Sustenance From Food Item Or Similar Food Item**	**Acute Toxicity Value Range (No. of Studies)**	**RQ Range Based on an EEC of 55 (µg/L)**	**Probability of Individual Effect [a]**	**Risk Interpretation**
Eastern oyster	Adult shortnose sturgeon; sea turtles	>1000 - >1700 (2)	<0.03 - <0.055 [b]	Not estimated based on lack of dose-response	Based on lack of LOC exceedance and/or lack of effects in the study at highest concentrations tested, atrazine is not likely to affect food items that are as sensitive as these organisms to the extent that indirect effects to predators would be expected
Leech	Adult shortnose sturgeon	>16000 (2)	<0.01	Not estimated based on lack of acute dose-response	
Mud crab	Sea turtles	>1000(1)	<0.055[b]	Not estimated based on lack of dose-response	
Snail	Adult shortnose sturgeon	>16,000 (1)	<0.01	Not estimated based on lack of dose-response	
Grass shrimp	Sea turtles	9000 (1)	<0.01	<1 in 1,000,000[c]	
Pink shrimp	Sea turtles	6900 (1)	<0.01	<1 in 1,000,000[c]	
Stonefly	Juvenile shortnose sturgeon	6700 (1)	<0.01	<1 in 1,000,000[c]	

[a] The probability of an individual effect was calculated using a probit dose response slope of 4.4 (MRID 45202917, scud); this is the only slope for technical grade atrazine reported in available ecotoxicity data for freshwater invertebrates. 95% Confidence intervals could not be calculated based on the available data (Table A-18).
[b] No effects were observed in the mud crab or the eastern oyster studies, and the EECs used in risk estimation were considered conservative; therefore, these slight LOC exceedances, should be interpreted with caution.
[c] Dose-response based on probit slope of 4.5 from study in mysid shrimp (MRID 43344902).

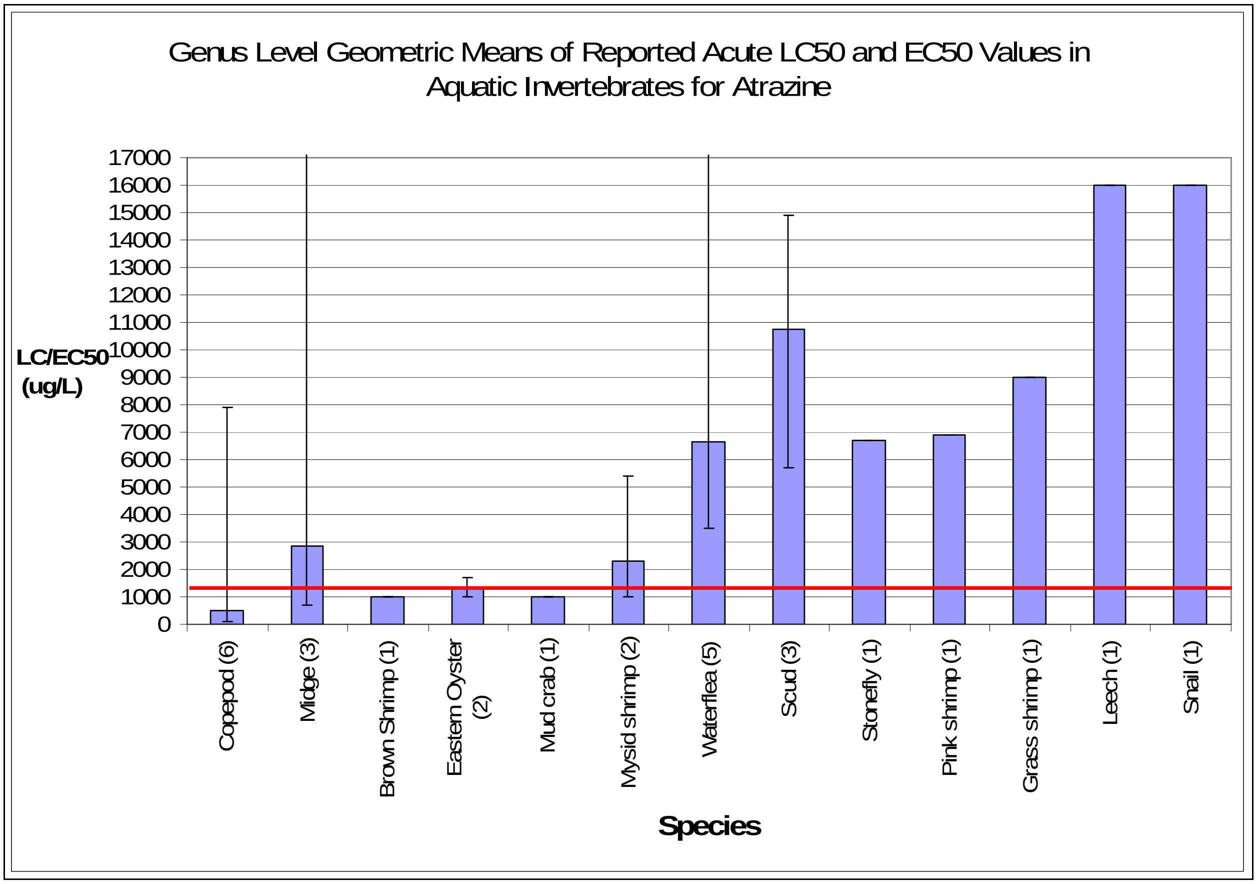


Figure 5-1. Range of Aquatic Invertebrate Acute Toxicity Values Reported for Atrazine.

The columns in the above graph represent the geometric means of values reported across the same genus. The y error bars represent the range of reported values. Error bars higher than the graph's maximum value of 17,000 µg/L were reported for two species. These values are 33,000 µg/L for the midge and 30,000 µg/L for the waterflea. Values in parentheses represent the number of studies included in the analysis. The horizontal line at 1100 µg/L represents toxicity values below which would result in LOC exceedances for aquatic invertebrates. Toxicity values higher than the horizontal line would not result in the endangered species LOC exceedance based on a PRZM/EXAMS estimated EEC of 55 µg/L (55 / 1100 = endangered species LOC of 0.05). Although toxicity values for the mud crab and oyster are less than 1100, RQs for these species should be interpreted with caution because no effects were observed in those studies (Table 5.9.).

5.2.2.2 Conclusion – Potential Indirect Effects to the Six Assessed Listed Species Resulting From Direct Effects to Aquatic Animals

Dwarf Wedgemussel

Direct effects LOCs were not exceeded for the surrogate freshwater zooplankton species (the acute RQ for daphnids was 0.02). Based on the assumption of probit dose response relationship with a mean estimated slope of 4.4, the estimated probability of an individual effect is <1 in 1,000,000. The probability of an individual mortality to food items as sensitive as waterfleas was estimated using a probit dose response slope of 4.4 (MRID 45202917); this is the only slope for technical grade atrazine reported in available ecotoxicity data for freshwater invertebrates and results in a more conservative analysis than use of the default slope of 4.5 recommended by U.S. EPA (2004a).

Based on the lack of LOC exceedance for the surrogate freshwater zooplankton species used in this assessment, the best available data suggests that atrazine use in the Chesapeake Bay watershed is expected to have “no effect” to the dwarf wedgemussel via reduction in zooplankton as food supply.

Shortnose Sturgeon

Shortnose sturgeon are omnivores and continuously feed on benthic and epibenthic invertebrates including mollusks, crustaceans, and oligochaete worms (U.S. EPA 2003b). Table 5.9 indicates that the endangered species LOC was exceeded for several surrogate food species for juveniles (midge, RQ = 0.076; mysid shrimp, RQ = 0.055).

The probability of an individual mortality to food items as sensitive as the midge was estimated using a probit dose response slope of 4.4 (MRID 45202917); this is the only slope for technical grade atrazine reported in available ecotoxicity data for freshwater invertebrates. Based on an assumption of a probit dose response relationship with a mean estimated slope of 4.4, the probability of an individual effect to this food item species at an RQ of 0.076 would be approximately 1 in 2,000,000.

In addition, the endangered species LOC is exceeded for mysid shrimp (RQ = 0.055). Based on the assumption of a probit dose-response relationship and a mean estimated slope of 4.5 in mysid shrimp (MRID 43344902), the probability of an individual effect at an RQ of 0.055 is 1 in $1x10^8$ (represented by <1 in 1,000,000 in Table 5.9). No LOCs were exceeded for the other juvenile shortnose sturgeon dietary organisms or for any surrogate adult shortnose sturgeon dietary organisms tested including mollusks (snail, leech, oyster), small crustaceans (daphnia), and stoneflies (Table 5.9).

The EECs used to estimate potential effects to invertebrate food items were based on PRZM/EXAMS estimated values using the standard water body, which are considered to be representative of short-term atrazine concentrations in headwater streams and small estuarine inlets. However, juvenile sturgeon are located in relatively deep river channels with high flow rates, and adult sturgeon are located in river mouths and in the main stem of the Chesapeake Bay. Based on additional modeling exercises and monitoring data presented in Section 3.4, EECs in these locales are likely to be lower than those estimated using the standard water body scenario; therefore, the results of this indirect effects analysis are considered conservative.

Based on the non-selective nature of feeding behavior, the conservative nature of the EECs used to derive RQs, and the low estimated magnitude (<1 in 1,000,000) of anticipated effects on the availability of food items as sensitive as those with LOC exceedances, the availability of dietary items of juvenile or adult shortnose sturgeon is not likely to be affected to an extent that would constitute a “take”, as defined in Section 2.1. Therefore, the data suggest that the significance of potential effects to the shortnose sturgeon from reduction in food is negligible, and that atrazine is "not likely to adversely affect" the shortnose sturgeon via effects on aquatic animals as food supply.

Sea Turtles

Loggerhead, leatherback, Kemp's ridley, and juvenile green turtles consume a variety of crustaceans, mollusks, jellyfish, other invertebrates, and fish. The endangered species level of concern was exceeded for one surrogate food item, brown shrimp (RQ = 0.055). Sufficient dose-response data were not available to allow for an evaluation of the probability of an individual effect to the brown shrimp. However, based on the lowest reported slope for surrogate saltwater aquatic invertebrate food items of 4.5 (mysid shrimp, MRID 43344902) and an assumption of probit dose-response relationship, the probability of an individual effect at an RQ of 0.055 would be <1 in 1,000,000 for brown shrimp. LOCs were not exceeded for other surrogate species tested including grass shrimp, pink shrimp, Eastern oysters, fish, or mud crabs. Additional formulated product data suggest that effects to fiddler crabs would also not be expected at the EECs derived in this assessment. The LC50 in fiddler crabs was >198,000 µg/L (MRID 00024395; described in Appendix A).

Each of the four sea turtle species assessed are located and feed in a variety of locations within the Chesapeake Bay, including main stem of major rivers, river mouths, the main stem of the Bay, and estuarine inlets. The EECs derived using the PRZM/EXAMS water body scenario of 55 µg/L may be representative of short-term peak exposures in headwater streams and estuarine inlets, but likely overestimates exposure and risk to invertebrates found in other areas of Chesapeake Bay where turtles are expected to be located (main stem of the Bay, major rivers, and river mouths).

Therefore, the above analysis based on a conservative exposure estimate suggests that a low number of dietary species relative to the number of species that serve as food to sea turtles are may be impacted by atrazine use in the Chesapeake Bay watershed. The magnitude of potential effects to the single dietary species (brown shrimp) with an LOC exceedance (RQ = 0.055) is low (estimated probability of an individual effect was <1 in 1,000,000) such that indirect effects to predators would be insignificant. Consequently, potential effects to the assessed sea turtle species from potential effects to aquatic animals as food items are anticipated to be negligible. Based on insignificant magnitude of potential effects to sea turtles from potential reduction in food supply organisms in the Chesapeake Bay, atrazine use in the Chesapeake Bay watershed is not likely to adversely affect sea turtles.

5.2.3. Summary of Effects Determinations: Indirect Effects from Direct Effects to Aquatic Animals

Table 5.10. Summary of Indirect Effects Determinations to the Six Assessed Listed Species Resulting from Direct Effects to Aquatic Animals

Species	Inirect Effects Conclusion	Basis for Conclusion
Dwarf wedgemussel	"No effect"	No acute or chronic LOCs are exceeded for surrogate fish or freshwater zooplankton species.
Shortnose sturgeon	"May affect, but not likely to adversely affect"	Some food items of these species may be affected; however, the significance of such effects to the shortnose sturgeon and to the four sea turtles are considered negligible based on the low anticipated magnitude of such an effect on the available food supply and the conservative exposure assumptions used in this analysis.
Four sea turtle species	"May affect, but not likely to adversely affect"	

5.2.4. Evaluation of Potential Indirect Effects to the Six Listed Species from Potential Effects to Aquatic Plants

Potential indirect effects to the six assessed listed species from effects on habitat and/or primary productivity were assessed using RQs based on the most sensitive aquatic plant study and PRZM/EXAMS estimated EECs from the standard water body scenario as a screen. If aquatic plant RQs exceed the LOC of 1.0, potential community level effects were evaluated using threshold concentrations for community level effects, as described in Section 4.

RQs used to estimate potential indirect effects to shortnose sturgeon, dwarf wedgemussel, and sea turtles based on primary productivity effects and/or decrease in available plants as food are in Table 5.11. Aquatic plants serve as important food sources for the dwarf wedgemussel and adult green turtles.

Table 5.11 Summary of RQs Used to Estimate Indirect Effects to Shortnose Sturgeon, Dwarf Wedgemussel, and Sea Turtles via Direct Effects on Aquatic Plants.					
Surrogate Species	**Toxicity Value (µg/L)**	**MRID**	**EEC (µg/L)**	**RQ**	**Risk Interpretation**
Freshwater Plants					Atrazine may affect the assessed listed species via effects on aquatic plants; further analysis of potential risks is necessary.
Non-vascular plants, green algae	EC50: 1 µg/L	00023544	55 µg/L	55	
Vascular plants, Cyanophyceae *Anabaena cylindrica*	EC50: 37 µg/L	43074804	55 µg/L	1.5	
Marine Plants					
Marine algae, *Isochrysis galbana*	EC50: 22 µg/L	41065204	55 µg/L	2.5	
Sago pondweed	EC50: 7.5 µg/L	45088231	55 µg/L	7.3	

Based on the results in Table 5.11, atrazine may indirectly affect each of the six assessed listed species via effects on aquatic plants. However, this analysis was based on the most sensitive aquatic plant species tested and a PRZM/EXAMS EEC generated based on the standard water body scenario. No known obligate relationship exists between any single plant species and the assessed listed species; therefore, additional analyses were performed to determine whether potential effects to individual plant species would likely result in community level effects. As previously discussed in Section 4, threshold concentrations were determined from realistic and complex time variable atrazine exposure profiles (chemographs) for modeled aquatic community structure changes (see Section 4.6 and Appendix B). If the following threshold concentrations are exceeded based on the EECs presented in Section 3 for the standard water body, then EECs based on the expected location of the assessed species within the Chesapeake Bay are further characterized. Exceedance of the following threshold concentrations indicates that changes in the aquatic plant community structure are possible.

- 14-day average = 38 µg/L
- 30-day average = 27 µg/L
- 60-day average = 18 µg/L
- 90-day average = 12 µg/L

The only uses that resulted in exceedance of the preceding thresholds are corn, sorghum, and fallow/idle lands; 14-, 30-, 60-, and 90-day average concentrations for these uses are in **Table 5.12.**

Table 5.12. Summary of PRZM/EXAMS Output for Corn, Sorghum, and Fallow/Idle Land and Comparison of Estimated Atrazine Concentrations to Community Level Effects Thresholds								
Use Site	**14-Day EEC (µg/L)**	**14-Day Effect Threshold**	**30-Day EEC (µg/L)**	**30-Day Effect Threshold**	**60-Day EEC (µg/L)**	**60-Day Effect Threshold**	**90-Day EEC (µg/L)**	**90-Day Effect Threshold**
Corn	46.9	38	46.5	27	45.6	18	44.4	12
Sorghum	54.8		54.3		53.7		52.5	
Fallow/ idle land	45.0		45.0		45.0		44.9	

The above uses result in EECs that exceed the 14-, 30-, 60-, and 90-day thresholds for community level effects; however, the EECs from this analysis were estimated using PRZM/EXAMS and the standard water body scenario. As previously discussed in Section 3.3, the standard water body may not accurately represent EECs in expected locations of the assessed species. Therefore, additional information on the location of the assessed species was used to further characterize potential exposures relative to those presented for the standard water body scenario. This analysis was presented in detail in Section 3.2 and is summarized below for each of the assessed species.

5.2.4.1. Additional Characterization of EECs in Flowing Streams and Rivers

The six species considered in this assessment are located in headwaters with low to moderate flow (mussel); in larger rivers (shortnose sturgeon and sea turtles); at river mouths (shortnose sturgeon and sea turtles); in the main stem of the Chesapeake Bay (sea turtles); and in estuarine inlets (sea turtles). Outside of short-term concentrations in small estuarine inlets and headwater streams, none of the locations where the assessed species are located are likely well represented by the standard water body, which was used to derive EECs used in RQ calculations. The primary reason that long-term concentrations estimated using the standard water body are not representative of these environments is that creeks and rivers are flowing water bodies, inlets and the Bay are subject to extensive mixing. In contrast, the standard water body is a static water body.

As described in detail in Section 3.2, a number of additional modeling exercises were performed to allow for characterization of potential effects of flow rate on the EECs. This analysis, together with available monitoring data, was used to further characterize potential exposures to the habitat of the listed species.

First, the variable volume water model (VVWM) was used to account for the influence of input and output (flow) on model predictions. Two alternate model runs were conducted using the VVWM. The first was done using standard assumptions and environmental fate parameters generally consistent with the non-flowing standard water body. The second assumption was designed to represent a larger volume water body that maximizes flow into the water body.

Second, the impact of flow was characterized using the Index Reservoir as the receiving water body and various flow rates. Flow assumptions considered representative of the headwater streams and mid-level reaches where the dwarf wedgemussel is located were evaluated. The EECs in larger rivers would be expected to be lower than those estimated for headwater streams due to higher flow rate and greater dilution potential.

In addition to the modeling exercises, existing monitoring data were used to characterize atrazine concentrations in the Chesapeake Bay and its tributaries. A detailed description of these data are in Section 3.4. Table 5.13 below provides a brief summary of the key results from the additional modeling and monitoring data used to characterize potential exposures.

Table 5.13. Characterization of Exposures Based on Additional Modeling and Available Monitoring Data

Analysis	Results
Modeling using VVWM	EECs are below the 14- and 30-day community level effects thresholds, but above the 60- and 90-day community-level threshold concentrations.
Modeling using the Index Reservoir and various flow rates	EECs decreased as flow rate increased. Modeling with flow rates representative of dwarf wedgemussel locations results in EECs that are lower than all community-level threshold concentrations.
Monitoring data, Chesapeake Bay and its tributaries	The maximum atrazine level was 30 µg/L, while the 99^{th}, 95^{th}, 90^{th}, 75^{th}, and 50^{th} percentile values were 2.5 µg/L, 0.5 µg/L, 0.28 µg/L, 0.1 µg/L, and 0.05 µg/L, respectively.
Monitoring, other representative water bodies	High peak atrazine concentrations have been observed; however, longer-term (>14 days) durations (when the data allow for calculation) are in the low µg/L range.

Collectively, the alternative modeling exercises and the monitoring data discussed in Section 3 suggest that atrazine concentrations in the headwater streams, rivers, river mouths, and Chesapeake Bay are expected to be in the low µg/L range, and lower than the 14-, 30-, 60-, and 90-day threshold concentrations for community-level effects. Therefore, community level effects to aquatic plants are considered unlikely in the Chesapeake Bay. No obligate relationship between the assessed species and any single aquatic plant species is known to exist. Therefore, it is concluded that atrazine is not likely to adversely affect any of the six assessed listed species via effects to aquatic vegetation.

There is additional concern, however, for green turtles from effects to aquatic plants because green turtles are primarily herbivores and may be found in minor estuarine inlets where atrazine concentrations may be higher than concentrations in the Bay, main rivers, and river mouths. However, due to extensive mixing within these inlets, peak atrazine concentrations that may occur after a run-off event are expected to be diluted rather quickly, such that any effects to aquatic vegetation would be anticipated to be temporary. Also, estuarine sea grasses, a forage item of green turtles, was shown to be less sensitive to atrazine than those used to derive aquatic plant risk quotients. EC50s for sea grasses range from approximately 70 µg/L (MRID 45227729) to 30,000 µg/L (MRID 45205101) in laboratory studies, which result in RQs that are less than LOCs. In addition, recovery of aquatic plants is expected to occur from the short-term exposures to atrazine within these minor inlets (Appendix A). Also, green turtles are highly mobile and are transitory at any single location within the Bay, and these minor inlets are not expected to be important to the green turtle's feeding or reproduction. Therefore, potential temporary effects to aquatic vegetation are not likely to result in harm or harassment of green turtles. This observation combined with the transient nature of atrazine within these inlets caused by extensive water mixing and the presence of a large pool of food within the Bay as it relates to the small areas potentially impacted (See Appendix G for map of submerged aquatic vegetation in the Chesapeake Bay), suggests that atrazine is not likely to adversely affect the green turtle via potential effects to food supply.

Table 5.14. Summary of Indirect Effects Determinations to the Six Assessed Listed Species Resulting from Effects to Aquatic Plants

Species	Direct Effects Conclusion	Basis for Conclusion
Dwarf wedgemussel	"May affect, but not likely to adversely affect"	Individual aquatic plant species within the Chesapeake Bay watershed may be affected. However, atrazine concentrations are not anticipated to exceed community-level effect threshold concentrations, and no known obligate relationship between the assessed species and any single aquatic plant species exists. Therefore, potential effects are considered to be insignificant in the context of a take as defined in Section 2.1.
Shortnose sturgeon	"May affect, but not likely to adversely affect"	
Four sea turtle species	"May affect, but not likely to adversely affect"	

5.2.5. Potential Indirect Effects to the Listed Species via Direct Effects to Terrestrial Plants

Riparian plants beneficially affect water and stream quality in a number of ways in both adjacent stream reaches and areas downstream of the riparian zone. A general discussion of riparian habitat and its relevance to the assessed species is provided below followed by a discussion of potential risks to the assessed species caused by effects on riparian areas from use of atrazine in the Chesapeake Bay watershed.

5.2.5.1. Discussion of Riparian Habitat and Its Relevance to the Assessed Species

Riparian vegetation serves several functions in the stream ecosystem including: serving as an energy source; providing organic matter to the stream; providing shading, which ensures thermal stability of the stream; and service as a buffer filtering out sediment, nutrients, and contaminants before they reach the stream. Criteria based largely on professional judgment have been proposed to assess the health of riparian zones and their ability to support fish habitat that may be used to assess the health of riparian zones (Fleming *et al.* 2001). These criteria are in Table 5.15 below. General criteria are identified for the width of vegetated area (i.e. distance from cropped area to water), structural diversity of vegetation, and canopy shading.

Table 5.15. **Semi-quantitative Criteria Related to Riparian Vegetation for Assessing the Health of Riparian Areas for Supporting Aquatic Habitats.**[1]				
Criteria	**Quality**			
	Excellent	**Good**	**Fair**	**Poor**
Buffer width	>18m	12 - 18m	6 - 12m	<6m
Vegetation diversity	>20 species	15 - 20 species	5 - 14 species	<5 species
Structural diversity	3 height classes grass/shrub/tree	2 height classes	1 height class	sparse vegetation
Canopy shading	mixed sun/shade	sparse shade	90% sun	no shade

[1] Adapted from Fleming *et al.* 2001.

Additional discussion of the importance of riparian areas in the Chesapeake Bay and its source waters can be found at http://www.chesapeakebay.net/ripar1.htm. Three attributes of habitat quality were linked to riparian vegetation for this assessment: water temperature, stream bank stability, and sediment loading. Each of these attributes are discussed briefly below.

Stream bank Stabilization: Riparian vegetation typically consists of three distinct types of plants; a groundcover of grasses and forbs, an understory of shrubs and young trees, and an overstory of mature trees. These plants serve as structural components for streams, with the root systems helping to maintain stream stability, and the large woody debris from the mature trees providing in stream cover. Riparian vegetation has been shown to be essential to maintenance of a stable stream (Rosgen, 1996). Destabilization of the stream can have a severe impact on aquatic habitat quality. Following a disturbance, the stream may down cut and widen, releasing sediment from the stream banks and scouring the stream bed. Bed scour can move redds (egg nests) after spawning, and/or decrease the number of good spawning sites by changing the size of gravel available. Destabilization of the stream can have a severe effects aquatic habitat quality by increasing sedimentation within the watershed. Effects of sedimentation are summarized below.

Sedimentation: Riparian vegetation is also important in moderating the amount of sediment loading from upland sources. The roots and stems of riparian vegetation can intercept eroding upland soil (USDA NRCS, 2000) and riparian plant foliage can reduce erosion from within the riparian zone by covering soil and reducing the impact energy of raindrops onto soil (Bennett, 1939). Sediment can smother benthic plants and animals. Increased turbidity from sediment loading could also reduce light transmission, potentially affecting aquatic plants (Cloern, 1987, Weissing and Huisman 1994) that are important for shelter and food. Increased suspended solids could also affect foraging behavior of sea turtles (US NMFS, 2004).

In addition, sediment carries excess nutrients, particularly phosphorus, into Bay waters, compromising water quality (www.chesapeakebay.net). Fine sediments can physically occlude interstitial pore spaces preventing fry from emerging and altering the habitat of mussel locations. In addition, sediment can bury and effectively suffocate mussels. The U.S. Fish and Wildlife Services reported that as little as one-quarter of an inch of silt covering the substrate resulted in death of 90% of mussel species evaluated (U.S. FWS, 2006). Increased siltation in the stream may also affect spawning, by settling on spawning gravel and reducing flow of water and dissolved oxygen to the eggs and fry (Everest *et al.* 1987). Reduced oxygen levels can result in direct mortality. In addition, fine particles settling on the streambed can also disrupt the food chain by reducing habitat quality for aquatic invertebrates, and adversely affect groundwater-surface water interchange (Nelson *et al.* 1991).

Thermal stability. Riparian habitat provides shading, which provides thermal stability. However, the sensitivity of the assessed species to temperature fluctuation is unknown, although U.S. EPA (2003b) reported that temperatures >29 degrees C are stressful to the shortnose sturgeon. Studies on the temperature sensitivity of the dwarf wedgemussel were not located.

5.2.5.2. Terrestrial Plant Exposure Analysis

The potential for atrazine to affect riparian areas in the Chesapeake Bay watershed was initially evaluated using terrestrial plant RQs (U.S. EPA, 2004). However, exceedance of terrestrial plant LOCs does not imply that atrazine use would be expected to result in adverse effects to the assessed species from riparian zone alterations. Discussion and interpretation of LOC exceedances is in Sections 5.2.5.8 and 5.2.5.9.

Plants in riparian areas may be exposed to atrazine residues carried from application sites via surface water runoff or spray drift. Atrazine residues can directly expose seedlings breaking through the soil surface and expose more mature plants through root uptake or by direct deposition onto foliage. Although both seedlings and more mature plants can be exposed to atrazine residues on the soil, seedlings are understood to be the more sensitive life stage. Runoff or drift into the terrestrial riparian buffer could damage or destroy the riparian vegetation, which provides important ecosystem services previously discussed such as temperature regulation, energy input, and stream bank stabilization.

Based on the results of the submitted terrestrial plant toxicity tests, it appears that emerged seedlings are more sensitive to atrazine via soil/root uptake exposure than emerged plants via foliar routes of exposure. However, all tested plants, with the exception of corn in the seedling emergence and vegetative vigor tests and ryegrass in the vegetative vigor test, exhibited adverse effects following exposure to atrazine. Therefore, a variety of herbaceous plants that may inhabit riparian zones may be sensitive to atrazine exposure. However, most woody plants are not expected to be sensitive to atrazine at environmentally relevant concentrations (MRID 4687040001), and atrazine is labeled for use in forestry production. Therefore, most woody plants are not expected to be affected by atrazine at the labeled application rates.

Atrazine exposure to riparian vegetation was estimated using TerrPlant (version 1.2.1), considering use conditions likely to occur in the Chesapeake Bay watershed. The TerrPlant model evaluates exposure to plants via runoff and spray drift. The runoff loading of TerrPlant is estimated based on the solubility of the chemical and assumptions about the drainage and receiving areas. The spray drift component of TerrPlant assumes that 1% and 5% of the application rate deposits in the receiving area for ground boom and aerial applications, respectively.

TerrPlant calculates exposure values for terrestrial plants inhabiting two environments: dry adjacent areas and semi-aquatic areas. The 'dry, adjacent area' is considered to be representative of a slightly sloped area that receives relatively high runoff and spray drift levels from upgradient treated fields and was used as a surrogate for riparian areas.

The following input values were used to estimate terrestrial plant exposure to atrazine from all uses: solubility = 33 ppm; minimum incorporation depth = 0 (from product labels); application methods: ground boom, aerial, and granular (from product labels). The following agricultural and non-agricultural scenarios were modeled: ground/aerial application to fallow land at 2.25 lbs ai/A, granular application to residential lawns at 2 lbs ai/A, and aerial or ground application to corn or sorghum at 2 lbs a.i./Acre. Although atrazine is also labeled for forestry use on conifers at an application rate of 4 lb ai/A, this use was not modeled because the best available information indicates that atrazine is rarely used in forestry in the Chesapeake Bay watershed (Section 2). However, potential impacts to riparian vegetation resulting from atrazine use on forestry (should herbicide use patterns on forestry in the Chesapeake Bay watershed change in the future) are discussed as part of the Risk Description. If forestry uses of atrazine are considered (at an application rate of 4.0 lb ai/A), the EECs in Table 5.16 and the resulting RQs would be expected to increase by a factor of approximately two.

Terrestrial plant EECs for non-granular and granular formulations are summarized in Table 5.16. EECs resulting from spray drift are derived for non-granular applications only.

Table 5.16. Screening-Level Exposure Estimates for Terrestrial Plants to Atrazine

Use/ App. Rate (lbs/acre)	Application Method	Total Loading to Dry Adjacent Areas (lbs/acre)	Drift EEC
Fallow land / 2.25	Aerial	0.16	0.14
	Ground	0.07	0.02
Corn/Sorghum / 2.0	Aerial	0.14	0.10
	Ground	0.06	0.02
Residential / 2.0	Granular	0.04	NA

5.2.5.3. Risk Quotients for Riparian Vegetation via Runoff Exposure

Comparison of plant EECs to the seedling emergence EC25 values presented in Section 4 indicates that terrestrial plant RQs are above LOCs for non-endangered plants for all species except corn and soybeans. RQs range from <1 (corn and soybeans) to 53 (carrots). LOCs were exceeded for both ground and aerial applications and for both granular and spray applications. Monocots and dicots show similar sensitivity to atrazine; therefore, RQs were similar across both taxa. Seedling emergence RQs are in Tables 5.17.

Table 5.17. Nontarget Terrestrial Plant Seedling Emergence RQs

Surrogate Species	EC25 (lbs ai/A)	EEC Dry adjacent areas	RQ Dry adjacent areas
Monocot - Corn (*Zea mays*)	> 4.0	Aerial: 0.16 Ground: 0.07 Granular: 0.04	<LOC
Monocot - Oat (*Avena sativa*)	0.004	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 40 Ground: 18 Granular: 10
Monocot - Onion (*Allium cepa*)	0.009	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 18 Ground: 7.8 Granular: 4.4
Monocot - Ryegrass (*Lolium perenne*)	0.004	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 40 Ground: 18 Granular: 10
Dicot - Root Crop - Carrot (*Daucus carota*)	0.003	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 53 Ground: 23 Granular: 13
Dicot - Soybean (*Glycine max*)	0.19	Aerial: 0.16 Ground: 0.07 Granular: 0.04	<LOC
Dicot - Lettuce (*Lactuca sativa*)	0.005	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 32 Ground: 14 Granular: 8
Dicot - Cabbage (*Brassica oleracea alba*)	0.014	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 11 Ground: 5 Granular: 2.9
Dicot - Tomato (*Lycopersicon esculentum*)	0.034	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 4.7 Ground: 2.1 Granular: 1.2
Dicot - Cucumber (*Cucumis sativus*)	0.013	Aerial: 0.16 Ground: 0.07 Granular: 0.04	Aerial: 12 Ground: 5.4 Granular: 3.1

[1] All toxicity values from Chetram, 1989 (MRID 42041403)

Based on exceedances of seedling emergence LOCs for all species tested except corn and soybeans, the following general conclusions can be made with respect to potential harm to riparian habitat via runoff exposures:

1. Atrazine may enter riparian areas via runoff where it may be taken up through the root system of sensitive plants.
2. Comparison of EC25 values from seedling emergence studies to EECs estimated using TERRPLANT suggests that existing vegetation may be affected or inhibition of new growth may occur.

a. Inhibition of new growth could result in degradation of high quality riparian habitat over time because as older growth dies from natural or anthropogenic causes, plant biomass may be prevented from being replenished in the riparian area.
b. Inhibition of new growth may also slow the recovery of degraded riparian areas that function poorly due to sparse vegetation because atrazine deposition onto bare soil would be expected to inhibit the growth of new vegetation.

3. Because LOCs were exceeded for most species tested (8/10) in the seedling emergence studies, it would be expected that many species of herbaceous plants may be affected by atrazine exposure.

5.2.5.4. RQs for Riparian Vegetation via Spray Drift Exposure

Vegetative vigor RQs exceeded LOCs for 3 dicot species (soybeans, cabbage, and cucumber) of 10 plants tested. Vegetative vigor RQs were not exceeded for any of the monocot species tested. The highest vegetative vigor RQ was 14 (cucumbers).

Table 5.18. Nontarget Terrestrial Plant Vegetative Vigor Toxicity[1]

Surrogate Species	EC25 (lbs ai/A)	Drift EEC (lbs ai/A)	RQ
Monocot - Corn (*Zea mays*)	> 4.0	Aerial: 0.11 Ground: 0.02	<LOC
Monocot - Oat (*Avena sativa*)	2.4	Aerial: 0.11 Ground: 0.02	<LOC
Monocot - Onion (*Allium cepa*)	0.61	Aerial: 0.11 Ground: 0.02	<LOC
Monocot - Ryegrass (*Lolium perenne*)	> 4.0	Aerial: 0.11 Ground: 0.02	<LOC
Dicot - Root Crop - Carrot (*Daucus carota*)	1.7	Aerial: 0.11 Ground: 0.02	<LOC
Dicot - Soybean (*Glycine max*)	0.026	Aerial: 0.11 Ground: 0.02	Aerial: 4.2 Ground: 0.77
Dicot - Lettuce (*Lactuca sativa*)	0.33	Aerial: 0.11 Ground: 0.02	<LOC
Dicot - Cabbage (*Brassica oleracea alba*)	0.014	Aerial: 0.11 Ground: 0.02	Aerial: 7.8 Ground: 1.4
Dicot - Tomato (*Lycopersicon esculentum*)	0.72	Aerial: 0.11 Ground: 0.02	<LOC

Table 5.18. Nontarget Terrestrial Plant Vegetative Vigor Toxicity[1]			
Surrogate Species	**EC25 (lbs ai/A)**	**Drift EEC (lbs ai/A)**	**RQ**
Dicot - Cucumber (*Cucumis sativus*)	0.008	Aerial: 0.11 Ground: 0.02	Aerial: 14 Ground: 2.5

[1] All Toxicity Values are from Chetram (1989), MRID 42041403.

The vegetative vigor RQs exceeded the LOC of 1.0 for three dicot plants (soybeans, cucumbers, and cabbage) with a maximum RQ of 14 in cucumbers. This analysis suggests that some dicots in riparian habitat are expected to be at risk from foliar exposure via spray drift. Therefore, riparian habitats comprised of herbaceous plants sensitive to atrazine may be adversely affected by spray drift. RQs were not exceeded for monocots; therefore, drift would not be anticipated to affect riparian zones comprised primarily of monocot species such as grasses.

Because RQs for terrestrial plants listed in Tables 5.17 and 5.18 are above LOCs, atrazine use is considered to have the potential to affect aquatic species by impacting plants in riparian areas potentially resulting in degradation of stream water quality. These potential effects are evaluated below.

5.2.5.5. Types of Riparian Zones Sensitive to Atrazine Effects

The parameters used to assess riparian quality that are potentially sensitive to atrazine were outlined in Table 5.15 and include buffer width, vegetation diversity, vegetation cover, structural diversity, and canopy shading. Buffer width, vegetation cover, and/or canopy shading could be reduced if atrazine exposure impacted plants in the riparian zone or prevented new growth from emerging. Plant species diversity and structural diversity may also be affected if only sensitive plants are impacted (Jobin *et al.*, 1997, Kleijn and Snoeijing, 1997), leaving non-sensitive plants in place. Atrazine may also affect the long term health of high quality riparian habitats by affecting seed germination. Thus, if atrazine exposure impacted these riparian parameters, water quality within the Chesapeake Bay watershed could be affected.

Because woody plants are typically not sensitive to environmentally-relevant atrazine concentrations (MRID 4687040001), effects on shading and structural diversity (height classes) of vegetation are not expected. The potential for effects is limited to herbaceous (non-woody) plants, which are not generally associated with shading or considered to represent vegetation of higher height classes. The most sensitive riparian quality criteria are expected to be plant diversity, vegetation cover, and buffer width because the more sensitive plants (young, herbaceous plants) are expected to important in maintaining these parameters. A reduction in the quality in any of these parameters may have the potential to reduce water quality and thus adversely affect the assessed listed species.

The riparian health criteria described in Fleming *et al.* (2001; Table 5.15) and the characteristics associated with effective vegetative buffer strips suggest that healthy riparian zones would be less sensitive to the impacts of atrazine runoff than poor riparian zones. Riparian zones rich in species diversity and woody species may contain sensitive species; however, they would also be less likely to consist of a high proportion of very sensitive plants. Wider buffers have greater potential to reduce atrazine residues over a larger area, resulting in lower levels. In addition, trees and woody plants in a healthy riparian area would act to filter spray drift (Koch *et al.* 2003) and push spray drift plumes over the riparian zone (Davis *et al.* 1994) thus reducing exposure to herbaceous plants, which tend to be more sensitive to atrazine. Thus, high quality riparian zones would be expected to be less sensitive to atrazine's effects than riparian zones that are narrow, low in species diversity, and comprised of young herbaceous plants or unvegetated areas. Therefore, the available data suggest that riparian zones comprised largely of herbaceous plants and grasses would likely be most sensitive to atrazine effects. Bare ground riparian areas could also be adversely affected by prevention of new growth of grass which can be an important component of riparian vegetation for maintaining water quality.

Effects from atrazine are more likely to occur in reaches abutting sparsely vegetated riparian zones because these are the areas where sediment loading to surface water and the potential for significant deposition is expected to be highest. However, high quality habitat for the assessed species and high quality spawning habitat of the shortnose sturgeon are more likely to occur in areas that have not previously been affected by sedimentation.

Cropping to the edge of surface water bodies is expected to result in the greatest level of sedimentation in adjacent water bodies because no riparian vegetation is present to reduce the amount of sediment reaching the water. However, the lack of riparian vegetation in these areas precludes atrazine-induced effects to such vegetation. Therefore, the use of atrazine on fields without riparian vegetation draining into the Chesapeake Bay and its source waters is not expected to significantly affect erosion from fields and subsequent sediment loading into the waters.

5.2.5.6. Agricultural Practices and Sedimentation

In row crop agriculture, land and soil management practices have been identified as having a large effect on erosion (Green *et al.* 2003, Tebrügge and Düring 1999). The practices identified as erosion reducing, some of which employ herbicide use, are consistent with recent U.S. government policies encouraging soil conservation (Uri and Lewis, 1998). Also, current atrazine labels mandate a 66 foot set back for streams. These setbacks are expected to result in lower loading from application sites to riparian areas; however, the reduction cannot be quantified.

In preparing soil for crops, seeding, and controlling pests, a number of different practices may be employed that have a large effect on erosion levels and, presumably on subsequent sediment loading to receiving water bodies. Those practices that disturb the soil are correlated to a greater extent with increased erosion; conversely, management practices that do not disturb the soil result in lowered erosion levels. For example, the method of tilling is strongly correlated with erosion levels (Shiptalo and Edwards 1998). No-till and chisel plow practices result in relatively low disruption of the soil and are associated with significantly reduced erosion levels. These two methods of tillage are commonly referred to as conservation tillage, based on their ability to preserve topsoil. Combining conservation tillage methods with the use of "cover crops" (not removing the crop residue after harvest to reduce the surface area of soil directly exposed the impact of rain drops) has been shown by numerous researchers to be an effective means of soil conservation, resulting in a significant reduction in erosion under a wide range of conditions (e.g. Williams *et al.* 2000, Jacinthe *et al.* 2004). An integral part of many soil conservation plans includes the use of herbicides (Mickelson *et al.* 2001, Kelly *et al.* 1996). Some soil conservation scenarios require greater use of herbicides relative to conventional tillage to control weeds that would be managed as a result of plowing (Kelly *et al.* 1996). Atrazine may be used as part of the soil conservation methods that reduce erosion or in more traditional farming methods, which may increase erosion due to the inherent nature of conventional tillage as opposed to any direct cause-effect relationship to atrazine use. Thus, atrazine use may be associated with relatively high or low sediment loadings resulting from upland agricultural erosion.

Another key factor in the evaluation of potential risks from use of atrazine to riparian habitats is that a number of sediment reducing strategies are currently in place for the Chesapeake Bay and its source waters. For example, Maryland (Maryland Department of Natural Resources, 2004) and Virginia (Commonwealth of Virginia, 2005) have sediment reduction strategies for the Chesapeake Bay Watershed including its tributaries as part of the Chesapeake Bay 2000 agreement (Chesapeake Bay Program, 2000; http://www.chesapeakebay.net/agreement.htm). These strategies include implementation of BMPs to reduce loading of nutrients, chemicals, and sediment into the Chesapeake Bay and its source waters. In some of the locations of the assessed species (for example, in the coastal zone or in critical areas), a sediment control and water quality (SCWQ) plan must be submitted to the state for agricultural land adjacent to tributaries of the Chesapeake Bay. These plans can include a number of BMPs such as tillage practices, land retirement, cover crops, tree planting, and riparian buffers. However, any number or combination of BMPs may be included in the SCWQ plan, and a quantitative relationship between the presence of these BMPs in combination with each other and reduction in sediment or nutrient loading or reductions in pesticide loading to riparian areas has not been established. It would be anticipated that atrazine use would have negligible impact on sediment loading in areas where these state adopted BMPs are implemented.

5.2.6. Potential for Atrazine to Affect the Assessed Species via Effects on Riparian Vegetation in the Chesapeake Bay Watershed

It is difficult to estimate the magnitude of potential impacts of atrazine use on riparian habitat and the magnitude of potential effects on stream water quality from such impacts as they relate to survival or reproduction of the assessed species. The level of exposure and any resulting magnitude of effect on riparian vegetation are expected to be highly variable and dependent on many factors. The extent of runoff and/or drift into stream corridor areas is affected by the distance the field is offset from the stream, local geography, weather conditions, and quality of the riparian buffer itself. The sensitivity of the riparian vegetation is dependent on the susceptibility of the plant species present to atrazine and composition of the riparian zone (e.g. vegetation density, species richness, height of vegetation, width of riparian area).

Quantification of risk to the assessed species is precluded by the following factors:

- A quantitative relationship between factors such as temperature fluctuation and increased sedimentation and survival or reproduction of the assessed species is not known;
- Relationship between distance of soil input into the stream and sediment deposition in areas critical to survival and reproduction of the assessed species is not known;
- Riparian areas are highly variable in their composition and location with respect to atrazine use; therefore, their sensitivity to potential damage is also variable.
- Locations of shortnose sturgeon spawning habitat within the Chesapeake Bay watershed are not known;

In addition, even if plant community structure was quantifiably correlated with riparian function, it may not be possible to discern the effects of atrazine on species composition separate from other agricultural actions or determine if atrazine is a significant factor in altering community structure. Plant community composition in agricultural field margins is likely to be modified by many agricultural management practices. Driving on and mowing of field margins and off-target movement fertilizer and herbicides are all likely to cause changes in plant community structure of riparian areas adjacent to agricultural fields (Jobin *et al.* 1997, Kleijn and Snoeijing 1997, Schippers and Joenje 2002). Although herbicides are commonly identified as a contributing factor to changes in plant communities adjacent to agricultural fields, some studies identify fertilizer use as the most important factor affecting plant community structure near agricultural fields (e.g. Schippers and Joenje 2002) and community structure is expected to be affected by a number of other factors (de Blois *et al.* 2002). In addition, urbanization and development are also critical factors that may affect stream quality. Thus, the effect of atrazine on riparian community structure would be expected to be one influence complicated by a myriad of other factors. Although the data do not allow for a quantitative estimation of risk from potential riparian habitat alteration, a qualitative discussion is presented below.

As previously discussed, the potential for atrazine to affect the six assessed listed species via effects to riparian vegetation depends on the potential exposure to and extent of sensitive (herbaceous/grassy) riparian zones and the importance of sensitive riparian zones to water quality in the Chesapeake Bay. As of 2004, there were approximately 44,507 acres of grassy riparian buffers in the Chesapeake Bay watershed (Sweeney, 2006), which represents approximately 0.1% of the total land and 0.4% of the agricultural land in the Chesapeake Bay watershed (land areas obtained from www.chesapeakebay.net/wspv31). There are a total of 185,500 miles of riparian forest buffers in the Chesapeake Bay (www.chesapeakebay.net)/wspv31), which corresponds to approximately 1,539,000 acres of forested riparian areas in the Chesapeake Bay watershed.[7] Therefore, the acreage of grassy riparian areas is approximately 2.8% of the forested riparian land (44,500 / 1.5 million = 0.028). Given that forested and grassy riparian areas represent only a fraction of the total riparian areas in the Chesapeake Bay watershed, the area of grassy riparian zones relative to all riparian areas in the Chesapeake Bay is likely considerably less than 2.8%. In addition, only a fraction of the

[7] The Chesapeake Bay Program (http://www.chesapeakebay.net/wspv31/) reports forested riparian buffer miles that are at least 100 feet in width and those that are less than 100 feet in width. This calculation was performed using a weighted average forested riparian buffer width. Riparian forest miles with >100 feet in width were assigned a width of 100 feet; riparian forest miles with a width <100 feet were assigned a width of 50 feet.

grassy/riparian areas are expected to be adjacent to cropland labeled for atrazine use, which would further diminish the extent of potential impacts of atrazine on grassy riparian buffers and resulting impacts to water quality in the Chesapeake Bay. Most (70%) of the grassy riparian buffers in the Chesapeake Bay is on Maryland's Eastern Shore (Sweeney, 2006). However, no potential suitable spawning habitat of shortnose sturgeon has been located in major rivers on Maryland's Eastern shore (U.S. EPA, 2003b), and Maryland's Eastern shore is not expected to be a critical feeding area for the sea turtles, which are transient throughout the Bay and tend to prefer the higher salinities of the Virginia portion of the Bay (http://www.chesapeakebay.net/info/seaturtle.cfm).

The low acreage of grassy, herbaceous riparian areas in the Chesapeake Bay watershed sensitive to atrazine exposure suggests that potential impacts of atrazine to these riparian areas and resulting effects on sedimentation in the Chesapeake Bay as a whole are expected to be minimal. This does not imply that grassy riparian buffers are ineffective in reducing nutrient or sediment loading, but rather that the acreage of land currently devoted to grassy riparian buffers is sufficiently low such that potential impacts of atrazine to sensitive riparian buffers are not expected to result in a measurable effect to the assessed species that reside in the main stem of the Chesapeake Bay, major rivers, or river mouths (shortnose sturgeon and sea turtles). For these reasons, potential impacts of atrazine to sensitive riparian areas are not likely to adversely affect shortnose sturgeon or sea turtles. This determination is based on insignificance of the effects because atrazine effects to grassy, herbaceous riparian vegetation in the Chesapeake Bay, major rivers, or river mouths cannot be meaningfully measured or detected in the context of a level of effect where "take" of a single short-nosed sturgeon or assessed sea turtle would occur.

However, it is possible that localized areas may exist where sensitive riparian areas are important with respect to soil retention and sediment loading prevention, particularly in small headwater streams. The only species included in this assessment located in small headwater streams is the dwarf wedgemussel. Therefore, additional analyses were conducted to allow for an evaluation of potential effects to the dwarf wedgemussel via impacts to riparian areas. The potential for atrazine to affect riparian areas of the dwarf wedgemussel was evaluated by assessing land use in the local watersheds of the known dwarf wedgemussel populations. If local land use data suggests a potential for atrazine exposure to affect the riparian areas to an extent that water quality may be affected, further evaluation of the potential sensitivity of the type of riparian area (if any) present around the known streams of dwarf wedgemussels is conducted. If land cover is consistent with atrazine use and riparian areas surrounding the known locations of the dwarf wedgemussel are expected to be sensitive to atrazine, then a "likely to adversely effect" determination could be made.

Land use within the watershed of known dwarf wedgemussel locations was evaluated to determine the possible extent of riparian area potentially exposed to and affected by atrazine. An example map used for this analysis, the Po River watershed, is in Figure 5-2 below. Figure 5-2 is one example of the analyses performed; similar maps were created for all known dwarf wedgemussel populations (Appendix I) except for McIntosh Run and Nanjemoy Creek because land use has been previously evaluated for these watersheds by U.S. FWS (1997). In addition, land use maps were not created for watersheds on Maryland's Eastern shore because agriculture and cropland are clearly a predominant land cover (Appendix I) in these watersheds based on data obtained from the Chesapeake Bay Program (http://www.chesapeakebay.net/wspv31/) and USDA (http://www.ams.usda.gov/statesummaries/). Therefore, it was assumed that atrazine exposure to riparian areas of these dwarf wedgemussel populations could be significant. Results of these analyses are summarized in Table 5.19 and are presented in greater detail in Appendix I. Land use data utilized to create the maps were obtained from the Regional Earth Science Applications Center (RESAC) of the University of Maryland and is available on-line at http://www.geog.umd.edu/resac/outgoing/.

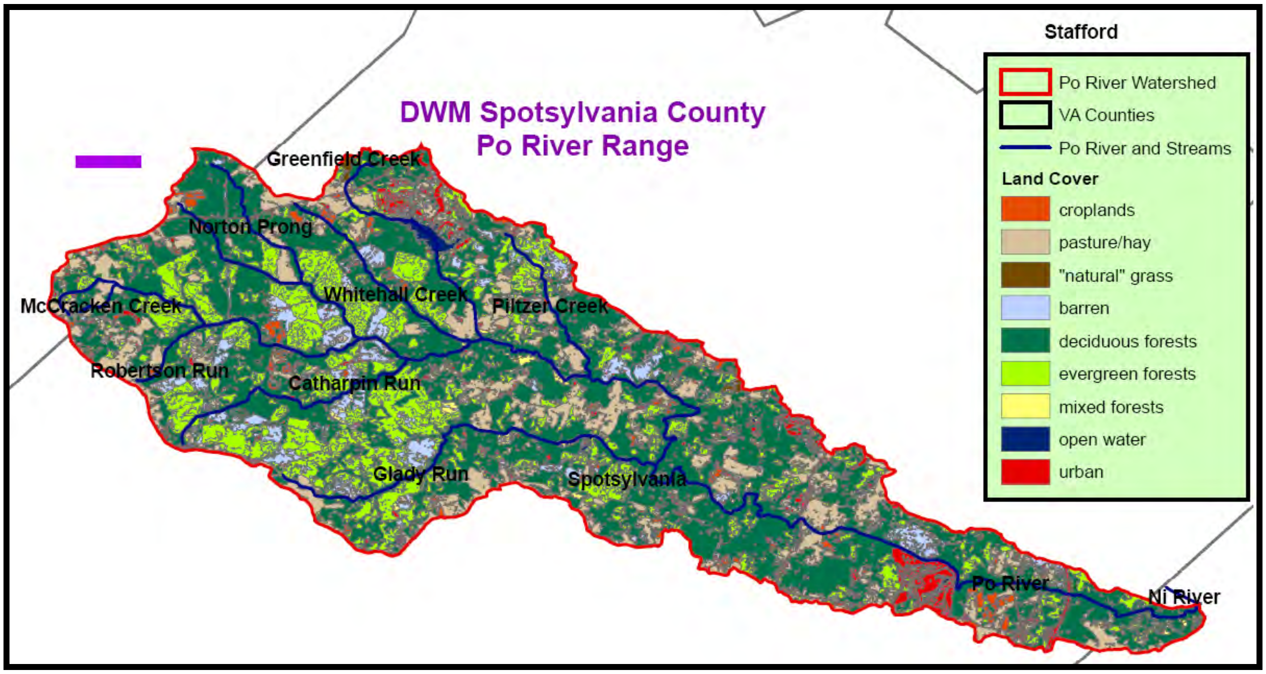


Figure 5-2. Map of Land Cover Data in the Po River Watershed.
The map in Figure 5-2 illustrates that land cover in the Po River watershed is mainly forested cover with some pastureland and urban areas. The area of cropland in the watershed is minimal, particularly areas in close proximity to the Po River. Atrazine use on forestry in Virginia is negligible (VA DOF, 2004), and pastureland is not a labeled use (U.S. EPA, 2006c). Therefore, atrazine exposure to riparian areas of the Po River is expected to be minimal, and the significance of any potential effects to the dwarf wedgemussel resulting in effects to riparian areas is expected to be low such that a take is not anticipated. For these reasons, atrazine is not likely to adversely affect dwarf wedgemussels in the Po River via effects to riparian areas. Comparable analyses were performed for other dwarf wedgemussel populations. Results of these analyses are summarized in Table 5.19 and are presented in greater detail in Appendix I. DWM = dwarf wedgemussel.

Cropland within the watersheds was further characterized using data from USDA (http://www.ams.usda.gov/statesummaries/) at the county level and from the Chesapeake Bay program (http://www.chesapeakebay.net/wspv31/) at the sub-watershed level. If the predominant land cover surrounding waters of specific dwarf wedgemussel habitats was found to be inconsistent with land cover on which atrazine would be expected to be applied, then a determination that atrazine will "not likely adversely affect" the dwarf wedgemussel population within the watershed could be made.

In the Po River example above (Figure 5-2), land cover within the watershed is mainly forest cover with some pastureland and urban areas; 77% of land cover forest, open water, wetland, or barren land (http://www.chesapeakebay.net/wspv31/). The extent of cropland in close proximity to the Po River is minimal (Figure 5-2), and only a small proportion of cropland (approximately 20%) in Spotsylvania County was harvested for corn from 1987 to 2002 (no sorghum was harvested in Spotsylvania County from 1987 to 2002; http://www.ams.usda.gov/statesummaries/). Atrazine use in forestry operations in Virginia is minimal (VA DOF, 2004). Together, these data suggest that the extent of riparian areas in the Po River watershed expected to be exposed to and affected by atrazine is minimal, and the significance of any resulting potential effects to water quality in the Po River and effects to the dwarf wedgemussel resulting from such effects is expected to be insignificant (as defined in Section 2.1) such that a take is not anticipated. For these reasons, it was concluded that atrazine is not likely to adversely affect dwarf wedgemussels in the Po River.

Similar analyses were performed for other dwarf wedgemussel populations. Results of these analyses are summarized in Table 5.19 and presented in greater detail in Appendix I. As summarized in Table 5.19, land use data surrounding riparian areas of waters inhabited by the dwarf wedgemussel in Virginia and two populations in Maryland suggest that riparian area exposure to atrazine is expected to be minimal. These populations include Aquia Creek, South Anna River, Po River, Carter Run, Nanjemoy Creek, and McIntosh Run. These data would also suggest that the water concentrations of atrazine estimated using PRZM/EXAMS modeling, which assumes that 87% to 100% of the watershed is cropped, over-estimate potential atrazine exposures to these populations of the dwarf wedgemussels.

As previously discussed, atrazine use in forestry is considered negligible in the Chesapeake Bay watershed (Powers, 2006; VA DOF, 2004; Muir, 2006; USDA, 2004; Wagner *et al.*, 2004, Pannill, 2006). However, even if atrazine was used in forestry operations, increased sedimentation from potential effects of atrazine on riparian areas may not occur. Intensive forest management practices, particularly road building, harvesting and mechanical site preparation, result in the greatest increases in erosion from forest sites. The available studies on the impact of mechanical versus chemical (i.e., herbicide) site-preparation for forestry demonstrate that use of mechanical site preparation methods result in 20 to 400% more sediment than observed on paired sites which are prepared with herbicides (Michael *et al.*, 2000). Therefore, even if atrazine is used in forestry operations near dwarf wedgemussel locations, its use may or may not be associated with increased sedimentation.

As discussed above, land use data suggest that exposure to riparian areas of some dwarf wedgemussel populations is expected to be minimal. However, land use surrounding Long Marsh Ditch, Mason's Branch, Aquia Creek, and tributaries of the Corsica River and Southeast Creek is predominantly agriculture (including cropland, http://www.chesapeakebay.net/wspv31/). Each of these locations are along Maryland's Eastern shore. Presence of large acreage of cropland does not necessarily imply that atrazine use would be expected to affect dwarf wedgemussels via effects to riparian areas. For atrazine to affect water quality via impacts on riparian vegetation, riparian areas would need to be present and to be comprised predominantly of grassy or herbaceous vegetation. Therefore, a qualitative analysis of the riparian areas along waters inhabited by dwarf wedgemussels where cropland is predominant land cover using aerial photography images obtained from Google Earth (Version 4.0, available at http://earth.google.com/). These analyses were not conducted for the dwarf wedgemussel populations where land use data suggests that atrazine exposure to riparian areas is expected to be minimal. An analysis of the riparian area of Norwich Creek is presented below as an example. Analyses conducted for riparian areas of other known dwarf wedgemussel populations in Maryland's Eastern shore are presented in Appendix I and summarized in Table 5.19.

Norwich Creek is part of the Tuckahoe drainage system. Land use in the watershed is predominantly (73%) cropland (U.S. FWS, 1997). The area of Norwich creek with dwarf wedgemussel habitat is surrounded by a 50 meter forested riparian zone (U.S. FWS, 1997). In addition, upstream locations are surrounded by predominantly forested riparian areas on both sides of the stream bank (Figures 5-3 and 5-4). Atrazine is not expected to affect forested riparian areas based on the low sensitivity of woody plants to atrazine. Therefore, riparian areas of Norwich Creek are not expected to be affected by atrazine to an extent that would be anticipated to have significant impacts on the dwarf wedgemussel.

Figure 5-3. Example of a Riparian Area of Norwich Creek.
The area in the photograph was the subject of analyses in U.S. FWS, 1997.

Figure 5-4. Example of Riparian Area Upstream of the Norwich Creek Site (U.S. FWS, 1997). Presented in Figure 5-3.

5.2.6. Summary of Conclusions: Potential of Atrazine to Affect the Six Listed Species via Impacts on Riparian Habitat.

Conclusions of the potential for atrazine to affect the six assessed listed species from potential terrestrial plant and riparian habitat effects are shown in Figure 5-5 and Table 5.19 below. The best available data suggests that sedimentation from agricultural land could have a negative impact on some dwarf wedgemussel populations; however, based either on land use or the type of riparian areas surrounding the known habitats of the dwarf wedgemussel (cropped or forested), atrazine use is expected to have an insignificant adverse impact on dwarf wedgemussels via effects to riparian vegetation. However, if the composition of riparian areas in the Chesapeake Bay and its tributaries changes over time or if land use patterns change over time such that atrazine use increases considerably, then this conclusion would need to be reevaluated.

Table 5.19. Conclusions for the Potential of Atrazine to Affect Specific Dwarf Wedgemussel Populations

Site	Basis for Effects Determination	Effects Determination
Dwarf Wedgemussel Habitats Expected to Have Minimal Exposure to Atrazine		
Aquia Creek, Stafford County, VA	The Aquia Creek watershed is 88 square miles with 81% of land cover forest, open water, wetland, or barren land and 13% agriculture (http://www.chesapeakebay.net/wspv31/). An analysis of the land cover surrounding the Aquia Creek watershed (Figure I-4, Appendix I) indicates forestland is the predominant land cover with minimal cropland in close proximity to the creek. Data from USDA indicate that 1% of the land cover in Stafford County (1500 of the 173,000 acres) was harvested for corn and sorghum (Attachment 1), and forestry is a rare use in Virginia (VA DOF, 2004). This analysis suggests that the extent of riparian areas of the Aquia Creek watershed that may be subject to atrazine exposure is minimal. Therefore, potential effects to riparian areas and resulting potential effects to dwarf wedgemussels are expected to constitute an insignificant effect. [d]	May effect, but not likely to adversely affect
South Anna River, Louisa County, VA	An analysis of the land use surrounding the South Anna River watershed (Figure I-3, Appendix I) indicates that land use is predominantly forest and pastureland, with minimal cropland or residential land cover. Data from USDA indicate that 1% and 4% of the land cover in Louisa and Hanover counties, respectively, was harvested for corn or sorghum (Attachment 1 of Appendix I). In addition, row crops constitute approximately 3% of land cover in Louisa county[b], and approximately 4% of land cover is residential.[b] Atrazine use in forestry operations is minimal in Virginia (VA DOF, 2004), and pastureland is not a currently labeled use (U.S. EPA, 2006). This analysis suggests that the extent of riparian areas of the South Anna River that may be subject to atrazine exposure is minimal. Therefore, potential effects to riparian areas and resulting potential effects to dwarf wedgemussels are expected to constitute an insignificant effect in the South Anna River.[d]	May effect, but not likely to adversely affect

Table 5.19. Conclusions for the Potential of Atrazine to Affect Specific Dwarf Wedgemussel Populations		
Site	**Basis for Effects Determination**	**Effects Determination**
Po River, Spotsylvania County, VA	The predominant land cover surrounding the Po River is forest land; 77% of land cover is forest, open water, wetland, or barren land.[a] Approximately 2% of the land cover in Spotsylvania county (4,300 of the 257,000 acres) was harvested for commodities labeled for atrazine uses (corn or sorghum; 2002 data, Attachment 1), and minimal cropland surrounds the Po River in Spotsylvania County (Figure I-1, of Appendix I). Atrazine use on forestry, the predominant land cover in the Po River watershed, in Virginia is rare (VA DOF, 2004), and atrazine is not labeled for use on pastures (U.S. EPA, 2006a). This analysis suggests that the extent of riparian areas of the Po River that may be subject to atrazine exposure is minimal. Consequently, the significance of any potential effects to the dwarf wedgemussel resulting in effects to riparian areas is expected to be low such that a take is not anticipated.[d] For these reasons, atrazine is not likely to adversely affect dwarf wedgemussels in the Po River via effects to riparian areas.	May effect, but not likely to adversely affect
Carter Run, Fauquier County, VA	The Carter Run watershed is approximately 56 square miles (36,000 acres). The predominant land cover in the watershed is forest (63%) and pasture land (34%). Cropland constitutes 1.5% of the land cover in the Carter Run watershed. Figure I-4 (Appendix I) illustrates land cover data in the watershed. These data were presented in the Bacteria TMDL for Carter Run, Fauquier County, Virginia (January, 2005). Atrazine use in forestry operations is minimal (VA DOF, 2004), and pasture land is not a labeled use for atrazine (U.S. EPA, 2006). Therefore, this analysis suggests that the extent of riparian areas of Carter Run that may be subject to atrazine exposure is minimal. Therefore, potential effects to riparian areas and resulting potential effects to dwarf wedgemussels are expected to constitute an insignificant effect.[d]	May effect, but not likely to adversely affect
Rappahannock River, Spotsylvania and Stafford County	Figure I-2 (Appendix I) indicates that land cover near the Rappahannock River in Spotsylvania and Stafford Counties is predominantly forested with minimal cropland. Approximately 2% of the land cover in Spotsylvania County and approximately 1% of the land cover in Stafford County were harvested for commodities labeled for atrazine use (corn or sorghum, 2002 data; Attachment 1 of Appendix I). Atrazine use in forestry operations is minimal in Virginia (VA DOF, 2004), and atrazine is not labeled for use on pastures (U.S. EPA, 2006a). This analysis suggests that the extent of riparian areas of the Rappahannock River watershed that may be subject to atrazine exposure is minimal, and the significance of any potential effects to the dwarf wedgemussel resulting in effects to riparian areas from atrazine is expected to be low such that a take is not anticipated.[d] For these reasons, atrazine is not likely to adversely affect dwarf wedgemussels in the Rappahannock River via effects to riparian areas.	May effect, but not likely to adversely affect

Table 5.19. Conclusions for the Potential of Atrazine to Affect Specific Dwarf Wedgemussel Populations		
Site	**Basis for Effects Determination**	**Effects Determination**
Nanjemoy Creek, Charles County, MD	Land cover data for the Nanjemoy Creek watershed was evaluated and presented in U.S. FWS (1997). The Nanjemoy Creek watershed is a predominantly forested watershed with 90% forested area within 100 meters of streams in the watershed. In addition, approximately 13% of land cover in the Nanjemoy Creek watershed is agriculture,[a] and approximately 2% of land in Charles County was harvested for corn or sorghum (Attachment 1 of Appendix I). Together, these data suggest that the extent of potential atrazine exposure to riparian areas of Nanjemoy Creek is minimal and that the types of riparian areas in the Nanjemoy Creek watershed (forestland, typically 100 meters or more) are not expected to be sensitive to atrazine exposure based on the low sensitivity of woody plants to atrazine (Wall et al., 2006; MRID 4687040001). Therefore, potential effects to riparian areas and resulting potential effects to dwarf wedgemussels are expected to constitute an insignificant effect.	May effect, but not likely to adversely affect
McIntosh Run, St. Mary's County, MD	Land cover data for the McIntosh Run watershed was evaluated and presented in U.S. FWS (1997). The McIntosh Run watershed is largely forested; 85% of the area within 100 meters of streams in the watershed is forested and 9% is cropland. In addition, 4% of land cover was harvested for corn or sorghum[c] (Attachment 1, appendix I). Bank vegetation is dominated by mature- and sapling-aged trees. [c] Together, these data suggest that the extent of potential exposure to riparian areas of McIntosh Run is minimal and that the types of riparian areas surrounding Nanjemoy Creek (forestland, typically 100 meters or more) are not expected to be sensitive to atrazine based on the low sensitivity of woody plants (Wall et al., 2006; MRID 4687040001). Therefore, potential effects to riparian areas and resulting potential effects to dwarf wedgemussels are expected to constitute an insignificant effect. [d]	May effect, but not likely to adversely affect
Dwarf wedgemussel Habitats with Predominant Agriculture Landcover in the Watershed		
Longmarsh Ditch; Mason Branch, MD	Riparian area of Longmarsh Ditch is cropped to the edge of the stream bank (Figure I-11, Appendix I); riparian buffer is absent (see Appendix I for photograph). Riparian area of Mason Branch, however, is primarily forested with some areas cropped to the streambank where Longmarsh Ditch becomes Mason Branch (Appendix I). Neither of these types of riparian areas are expected to be sensitive to atrazine. Based on the predominance of cropland in the watershed, atrazine exposure to the riparian areas is expected; however, based on the low anticipated sensitivity of wooded or cropped riparian areas to atrazine, potential effects to the riparian area are expected to constitute and insignificant effect to the dwarf wedgemussel.[d]	May effect, but not likely to adversely affect
Browns Branch, Granny Finley, and tributaries of Southeast Creek and Corsica River, MD	Land cover surrounding these habitats is primarily agriculture; therefore, atrazine exposure to these riparian areas may occur. However, aerial photography of these waters indicates the presence of wooded riparian buffers on both sides of the streambanks (Appendix I). Atrazine is not expected to detrimentally impact primarily forested riparian areas based on the low sensitivity of atrazine on woody plants (Wall et al., 2006; MRID 4687040001). Therefore, potential effects to dwarf wedgemussels resulting from potential impacts to riparian areas at these locations is expected to constitute an insignificant effect.[d]	May effect, but not likely to adversely affect
Norwich Creek, Talbot County, MD	Norwich Creek is part of the Tuckahoe drainage system. Land use in the watershed is predominantly (72%) agriculture (U.S. FWS, 1997). The area of Norwich creek with dwarf wedgemussel habitat is surrounded by a 50 meter forested riparian zone (U.S. FWS, 1997; Figure I-6a of Appendix I). In addition, upstream locations are surrounded by predominantly forested	May effect, but not likely to adversely affect

Table 5.19. Conclusions for the Potential of Atrazine to Affect Specific Dwarf Wedgemussel Populations		
Site	**Basis for Effects Determination**	**Effects Determination**
	riparian areas on both sides of the stream bank (Figure I-6b of Appendix I). Atrazine is not expected to detrimentally impact primarily forested riparian areas based on the low sensitivity of atrazine on woody plants (Wall et al., 2006; MRID 4687040001). Therefore, riparian areas of Norwich Creek are not expected to be affected by atrazine use to an extent that would be anticipated to have significant [d] impacts on the dwarf wedgemussel.	

[a] http://www.chesapeakebay.net/wspv31

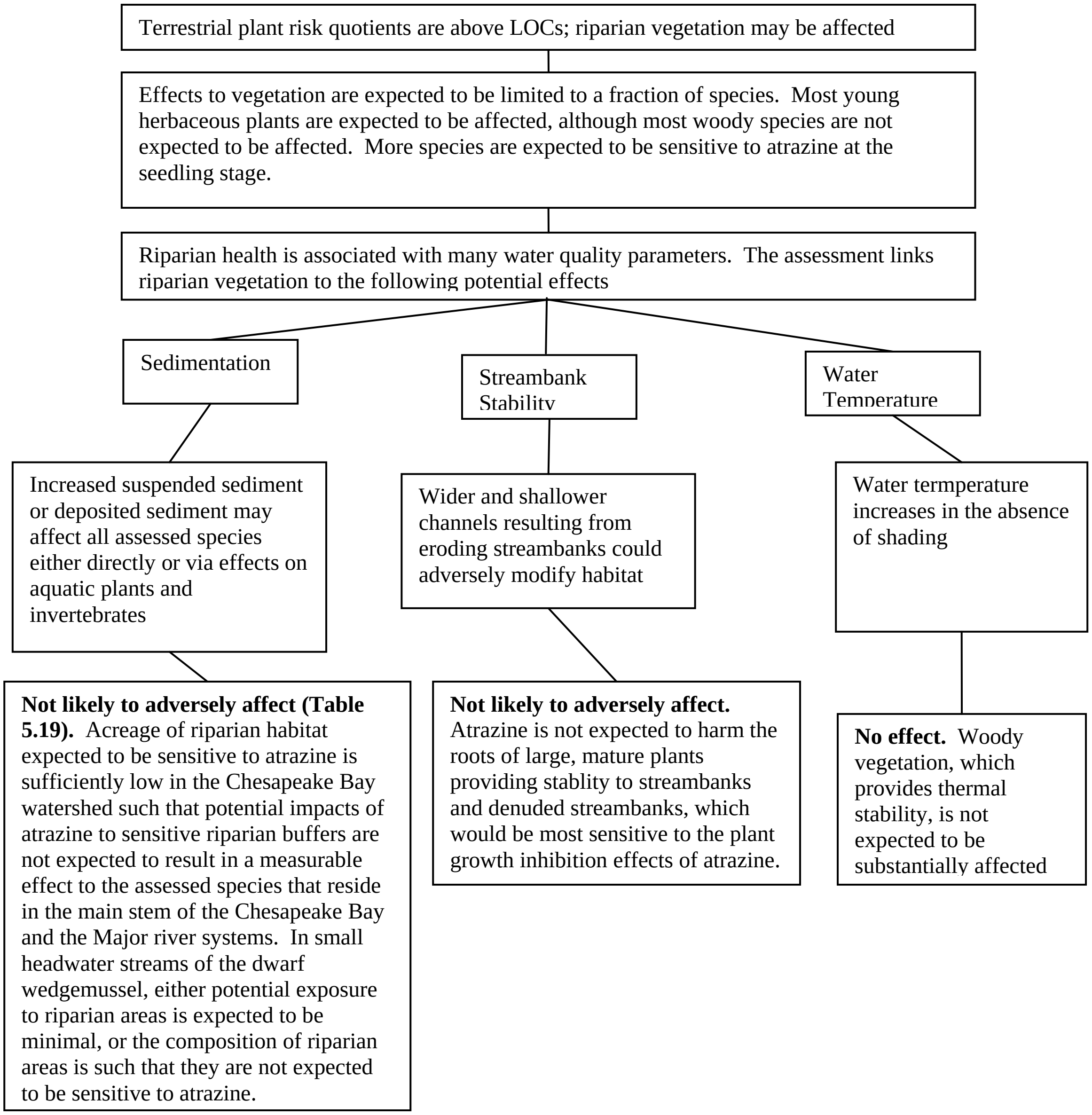
[b] http://fisher.lib.virginia.edu/collections/gis/nlcd/browse_county.html

[c] http://www.ams.usda.gov/statesummaries/)

[d] Significance of Effect: Insignificant effects are those that cannot be meaningfully measured, detected, or evaluated in the context of a level of effect where take occurs for even a single individual

[e] U.S. FWS. 1997. Characterization of Endangered Dwarf Wedgemussel (*Alasmidonta heterodon*) habitats in Maryland. CBFO-C97-01. January, 1997.

Figure 5-5. Summary of the Potential of Atrazine to Affect the Six Assessed Listed Species via Riparian Habitat Effects.



6.0 Uncertainties

A number of uncertainties are inherent in ecological risk assessment, which are discussed in detail in U.S. EPA (2004). Principle uncertainties in this risk assessment are discussed below. Additional uncertainties were discussed in Sections 2 through 5.

6.1. Exposure Assessment Uncertainties

6.1.1. PRZM Modeling Inputs and Predicted Aquatic Concentrations

Overall, the uncertainties inherent in the exposure assessment tend to result in over estimation of exposures. This is apparent when comparing modeling results with monitoring data. In general, the monitoring data should be considered a lower bound on exposure, while modeling represents an upper bound. Factors influencing the over-estimation of exposure include the assumption of no dilution or flow within the receiving water body. In addition, the impact of setbacks on runoff estimates have not been quantified while acknowledging that these buffers, especially well-vegetated buffers, are likely to result in significant reduction in runoff loading of atrazine.

In general, the simplifying assumptions used in this assessment appear from the characterization above to be reasonable especially in light of the analysis completed and the available monitoring data. There are also a number of assumptions that tend to result in over-estimation that cannot be quantified, but can be qualitatively described. For instance, modeling in this assessment for each use site assumes (with the exception of the right of way scenario) that the entire 10-hectare watershed is taken up by the respective use pattern. The assessment assumes that all applications have occurred concurrently on the same day at the exact same application rate. This is unlikely to occur in reality but is a reasonable assumption in lieu of actual data. In addition, the use of the standard water body assumes no flow through and thus the longer-term average concentrations presented above are likely reasonable approximations of headwater streams and water bodies but are also likely over-estimates of what is expected in lower reaches of the tributaries and within the Chesapeake Bay itself.

In general, buffer restrictions are an effective means of reducing movement of pesticides via drift to non-target aquatic resources. Effectiveness of a spray drift buffer can be evaluated quantitatively using AgDrift, which estimates the percentage of drift that is expected from a given buffer distance. This buffer-specific drift fraction is then used in PRZM/EXAMS in lieu of the default spray drift percentages that assume no buffer to quantitatively evaluate the effectiveness of a given buffer distance on off-site drift loadings. Currently, atrazine labels specify setback (or buffer) distances between applications and surface water bodies. These distances were integrated into this assessment using AgDrift to estimate distance specific spray drift values as substitute for the standard edge-of-field assumptions.

Unlike spray drift, there are currently no models that evaluate the effectiveness of a vegetative buffer on runoff and loadings. The effectiveness of vegetative buffers is highly dependent on the condition of the buffer. For example, a well-established, healthy vegetative buffer can be a very effective means of reducing runoff and erosion from agricultural fields. Alternatively, a buffer of poor vegetative quality or a buffer that is channelized can be ineffective at reducing loadings. Until such time that a quantitative method for estimate the effect of vegetative buffers of various conditions on pesticide loadings, it can only be stated that aquatic exposure predictions are likely to overestimate exposure where healthy vegetative buffers exist and likely do not overestimate exposure where poorly developed, channelized, or bare buffers exist.

In general, the linked PRZM/EXAMS model produces estimated aquatic concentrations that are exceeded once within a ten-year period. PRZM is a process or "simulation" model that calculates what happens to a pesticide in a farmer's field on a day-to-day basis. It considers factors, such as rainfall and plant transpiration of water, as well as how and when the pesticide is applied. It has two major components: hydrology and chemical transport. Water movement is simulated by the use of generalized soil parameters, including field capacity, wilting point, and saturation water content. The chemical transport component can simulate pesticide application on the soil or on the plant foliage. Dissolved, adsorbed, and vapor-phase concentrations in the soil are estimated by simultaneously considering the processes of pesticide uptake by plants, surface runoff, erosion, decay, volatilization, foliar wash-off, advection, dispersion, and retardation.

Uncertainties surrounding each of the individual components named above add to the overall uncertainty of the modeled concentrations. Additionally, model inputs from the environmental fate degradation studies are chosen to represent the upper confidence bound on the mean, values that are not expected to be exceeded in the open environment 90 percent of the time. Mobility input values are chosen to be representative of conditions in the open environment. The natural variation in soils adds to the uncertainty of modeled values. Factors such as application date, crop emergence date, and canopy cover can also affect estimated concentrations, adding to the uncertainty of modeled values. Factors within the ambient environment such as soil temperatures, sunlight intensity, antecedent soil moisture, and surface water temperatures can cause actual aquatic concentrations to differ for the modeled values.

Additionally, the rate at which atrazine is applied, the percent of a watershed that is cropped, and the percent of crops in that watershed that was actually treated with atrazine may be lower than the default assumption of the maximum allowable application rate being used, the entire crop being treated, and the default estimate of the area within a watershed planted with agricultural crops. The geometry of a watershed, and limited meteorological data sets also add to the uncertainty of estimated aquatic concentrations.

There is significant uncertainty with the quantitative use of the predicted EEC generated from these alternative scenarios. The standard water body and the Index Reservoir has been developed and vetted through a public peer review process. Both were developed with a specific range of exposure settings in mind. For the EXAMS static water body the 1 hectare body is intended to represent highly vulnerable water bodies, streams, creeks and rivers in headwater areas adjacent to agricultural fields. The Index Reservoir was developed to represent a small highly vulnerable drinking water reservoir. Neither water body was intended to represent larger, faster flowing water bodies. Therefore, the use of these two water bodies to represent flowing streams, creeks and rivers is intended only to provide a sense of the impact of flow on the modeled EEC for characterizing what those EEC represent and is not intended to provide the means to better characterize the exposure and potential risks.

6.1.2. Monitoring Data

The monitoring data used in this risk assessment were not collected for the purpose of supporting an ecological risk assessment; therefore they are not likely to be representative of peak atrazine concentrations.

6.1.3. Exposure to Degradates

Some degradates of atrazine are common to other triazine herbicides such as simazine or propazine. Therefore, exposure to degradates could be higher because some of the degradates are common to other triazines that may co-occur in the action area. However, given the low magnitude of risk of the degradates, this uncertainty is not expected to impact the conclusions of this assessment.

6.1.4. Use Characterization

Corn is expected to be the predominant atrazine use in the Chesapeake Bay watershed. However, atrazine may also be used on conifers and softwoods, and there is considerable forestland in some areas of the Chesapeake Bay watershed. Exposure to the assessed listed species from atrazine use in forestry operations was considered negligible because a total of 24 pounds of atrazine was used in the Commonwealth of Virginia by Virginia's forestry community in 2003 (VA DOF, 2004). USDA (2004) indicates that use of atrazine in coniferous evergreen operations is also low. In addition, atrazine is not used in the maintenance of State forestland in Maryland (MD DNR, 2006) and is not used in National Forestland (http://www.fs.fed.us/foresthealth/publications/pesticide/pur/reports.htm). The Maryland Department of Natural Resources also indicated that atrazine use in tree farms is uncommon in Maryland (MD DNR, 2006; Pannel, 2006). In addition, atrazine is applied only during the first year of tree growth. Based on the extended duration between planting and harvest for trees, only a small proportion of forestland is expected to be less than 1 year and, thus, treated with atrazine. For these reasons, potential risk from atrazine use on trees is expected to be less than risks based on agricultural crops described in this assessment.

Aquatic EECs from forestry (PRZM conifers scenario) were lower than those for sorghum and were, therefore, not used in RQ calculations. Therefore, the assumption that atrazine use in forestry is negligible would not impact aquatic EECs or RQs. However, the assumption of negligible atrazine use in forestry was important in the evaluation of land use surrounding dwarf wedgemussel populations, which was used to evaluate potential effects to riparian areas of dwarf wedgemussel locations, particularly in Virginia. If agricultural practices in forestry change such that atrazine use increases dramatically, then risks to Virginia dwarf wedgemussel populations would need to be re-evaluated.

6.1.5. Long-range Transport of Volatilized Atrazine

The environmental fate and monitoring data suggest that long range transport of volatilized atrazine is a possible route of exposure for the listed species. However, given the magnitude of documented atrazine concentrations in rainfall at or below available surface water and groundwater monitoring data (as well as modeled estimates for surface water), and the lack of modeling tools to predict the impact of long range transport of atrazine, the extent of the action area is defined by the transport processes of runoff and spray drift for the purposes of this assessment.

6.2. Effects Assessment Uncertainties

6.2.1. Age Class and Sensitivity of Effects Thresholds

It is generally recognized that test organism age may have a significant impact on the observed sensitivity to a toxicant. The acute toxicity data for fish are collected on juvenile fish between 0.1 and 5 grams. Aquatic invertebrate acute testing is performed on recommended immature age classes (e.g., first instar for daphnids, second instar for amphipods, stoneflies, mayflies, and third instar for midges).

Testing of juveniles may overestimate toxicity at older age classes for pesticidal active ingredients, such as atrazine, that act directly (without metabolic transformation) because younger age classes may not have the enzymatic systems associated with detoxifying xenobiotics. In so far as the available toxicity data may provide ranges of sensitivity information with respect to age class, this assessment uses the most sensitive life-stage information as measures of effect for surrogate aquatic animals. Nonetheless, no data on glochidial stage mussels were available; therefore, data on juvenile bivalves were used for RQ calculations used to estimate risk to mussels. It is unknown if glochidial stage mussels are expected to be more, less, or equivalent in sensitivity to atrazine as the juvenile mussels that were tested in the available studies.

6.2.2. Use of Surrogate Species Effects Data

Guideline toxicity tests are not available for turtles or freshwater mussels; therefore, surrogate species were used as outlined in U.S. EPA (2004). Therefore, birds were used as a surrogate for reptiles and saltwater mussels were used as a surrogate for the freshwater dwarf wedgemussel. The available open literature information on atrazine toxicity to reptiles was insufficient to allow for a direct comparison of the surrogate species to the assessed species. Extrapolating the risk conclusions from the surrogate tested species to the assessed species may either underestimate or overestimate potential risks. However, as described in Section 4, use of birds as a surrogate for sea turtles was considered conservative. Efforts are made to select the organisms most likely to be affected by the type of compound and usage pattern; however, there is an inherent uncertainty in extrapolating across phyla. LOCs are intentionally set low, and conservative estimates are made in the screening level risk assessment to account for these uncertainties.

Aquatic invertebrates were used as surrogates for jellyfish, which is the primary dietary item of leatherback turtles. It is unknown if the available toxicity data are representative of the sensitivity of jellyfish to atrazine. If jellyfish are particularly sensitive to atrazine, then jellyfish availability could be affected by its use. However, leatherback turtles are highly pelagic and reside in the main stem of the Chesapeake Bay. At atrazine concentrations expected in the main stem of the Bay (low µg/L range), none of the surrogate aquatic invertebrate species tested are likely to be adversely affected to such an extent that a "take" as defined in Section 2.1 is expected for turtles resulting from a reduction in food supply. Also, data from the Chesapeake Bay Monitoring Program suggests that jellyfish populations have remained stable in the main channel of Chesapeake Bay since 1984, where this highly pelagic species is most likely to be found (Figure 6-1). The presence of a stable jellyfish population does not necessarily indicate that atrazine have not impacted jellyfish numbers prior to 1984 or that atrazine may not affect seasonal fluctuations of jellyfish numbers. However, these data support the conclusion that atrazine does not appear to be affecting jellyfish numbers in the Chesapeake Bay.

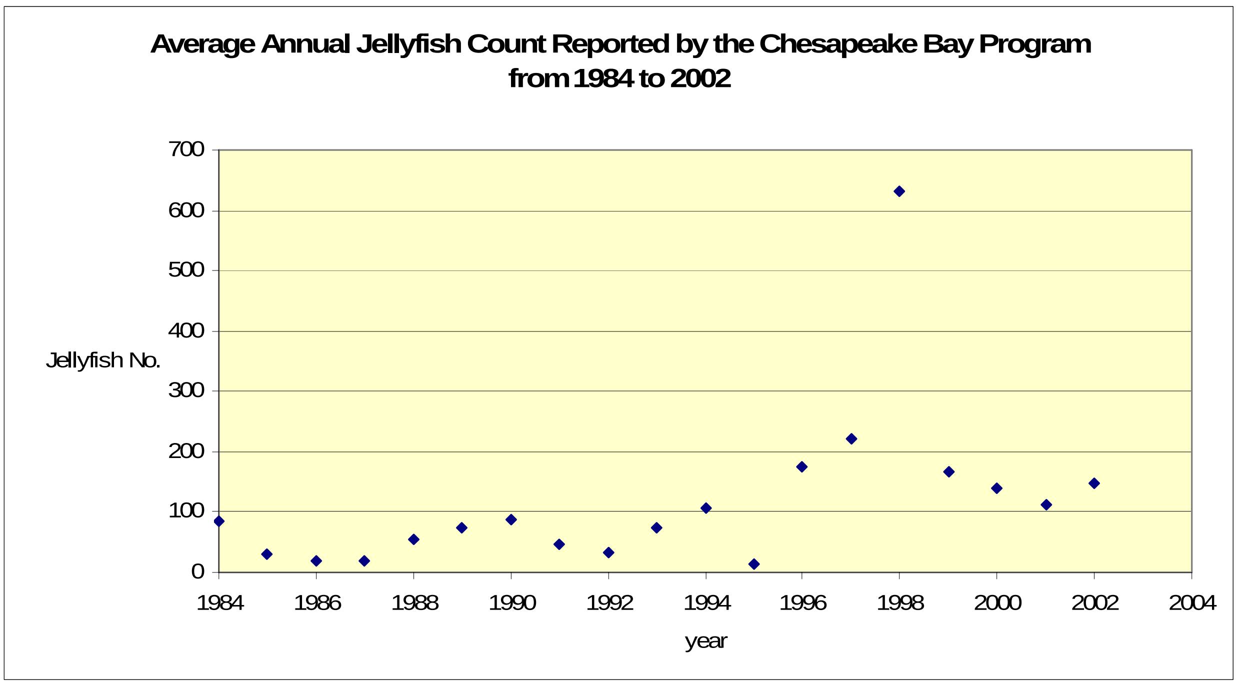


Figure 6-1. Jellyfish Monitoring Data in the Chesapeake Bay
Values represent average number of jellyfish across all monitoring stations that reported jellyfish numbers >0. Figures representing all data (count and biovolume across all monitoring stations) are in Appendix H). Data obtained from (http://www.chesapeakebay.net/baybio.htm).

6.2.3. Use of the Lowest Aquatic Invertebrate Toxicity Value to Estimate Risk to Potential Food Items

Several of the aquatic invertebrates (e.g., midge, copepod, daphnid) showed a wide range of sensitivity within and between species of the same genus (2 orders of magnitude). Therefore, acute RQs based on the most sensitive toxicity endpoint for aquatic invertebrates may represent an under- or over-estimation of potential direct risks to freshwater invertebrates and indirect effects to the assessed species via a reduction in available food.

6.2.4. Absence of a chronic study in the most acutely sensitive marine/estuarine invertebrate

Acute toxicity data suggest that the copepod was the most sensitive species in acute studies. However, the copepod is not used as a surrogate for direct effects to any of the assessed species. In addition, the copepod was not a surrogate food item of any of the assessed species. Therefore, potential impacts of this uncertainty to the conclusions of this assessment are expected to be minimal. However, one study in open literature was located that allows for some characterization of potential chronic effects of atrazine on the copepod. Forget-Leray *et al.* (2004) reported results from a 96-hour, 10-day, and 30-day exposure study. An acute 96-hour LC_{50} of 125 µg/L was reported for the copepod *E. affinis* nauplii. A 10-day NOAEC of 25 µg/L was reported; increased incidences of mortality were observed at 49 µg/L. In addition, delayed maturity (time from nauplius to adult before molting) was observed at 25 µg/L in the 30-day exposure study. This study is discussed in further detail in Appendix A. Although there are uncertainties associated with the study that may limit its utility in ecological risk assessment, including reporting deficiencies and use of an unacceptable solvent, these data suggest that the copepod may represent the most sensitive species tested in available chronic studies. Potential impacts of this uncertainty on this risk assessment are discussed in Section 5; however, use of a NOAEC of 25 µg/L in place of the NOAEC of 60 µg/L used in risk estimation would not be expected to alter the conclusions of this assessment.

6.2.5. Extrapolation of Long-term Environmental Effects from Short-term Laboratory Tests

The influence of length of exposure and concurrent environmental stressors (e.g., urban expansion, habitat modification, decreased quantity and quality of water, predators, etc.) to the assessed species may affect the species response to atrazine. The most probably effect of these types of uncertainty is that the effect is underestimated. Timing, peak concentration, and duration of exposure are critical in terms of evaluating effects, and these factors will vary both temporally and spatially within the action area. Overall, the effect of this variability may result in either an overestimation or underestimation of risk.

6.2.6. Use of Threshold Concentrations as Community-Level Endpoints

For the purposes of this endangered species assessment, threshold concentrations are used to predict potential indirect effects (via aquatic plant community structural change) to the assessed species. The conceptual aquatic ecosystem model used to develop the threshold concentrations is intended to simulate the ecological production dynamics in a 2nd or 3rd order Midwestern stream; however, the model has been correlated to the micro- and mesocosm studies, which were derived from a wide range of experimental studies (i.e., jar studies to large enclosures in lentic and lotic systems), that represent the best available information for atrazine-related community-level endpoints.

Although it is not possible to determine how well the responses observed in the micro- and mesocosm studies reflect the Chesapeake Bay community, available microcosm and mesocosm data and laboratory studies (Appendix A) do not indicate that estuarine systems are more sensitive than the freshwater systems on which the threshold concentrations were based. In addition, the available laboratory saltwater plant studies do not suggest increased sensitivity compared with freshwater aquatic plant species. Given that threshold concentrations were derived based on the best available information from available community-level data for atrazine, these values are intended to be protective of the aquatic community. Additional uncertainties associated with use of the screening thresholds to estimate community-level effects are discussed in Section B.8 of Appendix B.

6.2.7. Sediment Loading from Riparian Effects

No standard methods are available for assessing the effects from increased sedimentation that is a potential effect of damaging riparian vegetation.

6.2.8. Exposure to Pesticide Mixtures

This assessment considered only the single active ingredient of atrazine. However, the assessed species and their environments may be exposed to multiple pesticides simultaneously. Interactions of other toxic agents with atrazine could result in additive effects, synergistic effects or antagonistic effects. Conceptually, the combined effect of the mixture is equal to the sum of the effects of each stressor (1 + 1 = 2) for additive toxicity. Synergistic effects occur when the combined effect of the mixture is greater than the sum of each stressor (1 + 1 >2), and antagonistic effects occur when the combined effect of the mixture is less than the sum of each stressor (1 + 1 <2).

The available data suggest that pesticide mixtures involving atrazine may produce either synergistic, additive, or antagonistic effects. Mixtures that have been studied include atrazine with insecticides such as organophosphates and carbamates or with herbicides including alachlor and metolachlor. Additive or synergistic effects have been reported in several taxa including fish, amphibians, invertebrates, and plants.

As previously discussed, evaluation of pesticide mixtures is beyond the scope of this assessment because of the myriad of factors that cannot be quantified based on the available data. Those factors include identification of other possible co-contaminants and their concentrations, differences in the pattern and duration of exposure among contaminants, and the differential effects of other physical/chemical characteristics of the receiving waters (e.g. organic matter present in sediment and suspended water). Evaluation of factors that could influence additivity/synergism is beyond the scope of this assessment and is beyond the capabilities of the available data to allow for an evaluation. However, it is acknowledged that not considering mixtures could over- or under-estimate risks depending on the type of interaction and factors discussed above.

6.2.9. Sublethal Effects

The assessment endpoints used in ecological risk assessment include potential effects on survival, growth, and reproduction of the assessed species and organisms on which the species depend for survival. A number of studies were located that evaluated potential sublethal effects to fish from exposure to atrazine. Many of these studies reported toxicity values that were less sensitive than the submitted studies, and were not considered for use in risk estimation. However, several fish studies were located in the open literature that reported effects on endpoints other than survival, growth, or reproduction at concentrations that were considerably lower than the most sensitive endpoint from submitted studies.

Reported sublethal effects including changes in hormone levels, behavioral effects, kidney pathology, gill physiology, and potential olfaction effects have been observed at concentrations lower than 65 µg/L, the most sensitive fish life-cycle NOAEC (see Appendix A and Section 4.1.2.). These studies were not considered appropriate for risk estimation in place of the life cycle studies because quantitative relationships between these sublethal effects and the ability of fish to survive, grow, and reproduce has not been established. The magnitude of the reported sublethal effect associated with reduced survival or reproduction has not been established; therefore it is not possible to quantitatively link sublethal effects to the selected assessment endpoints for this ESA. In addition, in the fish life-cycle studies, no effects were observed to survival, reproduction, and/or growth at levels associated with the sublethal effects. Also, there were limitations to the studies that reported sublethal effects that preclude their quantitative use in risk assessment (see Appendix A and Section 4.2.1). Nonetheless, if future studies establish a quantitative link between the reported sublethal effects and fish survival, growth, or reproduction, the conclusions with respect to potential effects to fish may need to be revisited.

Upon evaluation of the available studies, however, the most sensitive NOAEC from the submitted life-cycle studies was considered to be the most appropriate chronic endpoint for use in risk assessment. In the life-cycle study design, fish are exposed to atrazine from one stage of the life cycle to at least the same stage of the next generation (e.g. egg to egg). Therefore, exposure occurs during the most sensitive life stages and during the entire reproduction cycle. Four life cycle studies have been submitted in support of atrazine registration. Species tested include brook trout, bluegill sunfish, and fathead minnows. The most sensitive NOAEC from these studies was 65 µg/L.

6.3. Assumptions Associated with the Acute LOCs

The risk characterization section of this endangered species assessment includes an evaluation of the potential for individual effects. The individual effects probability associated with the acute RQ is based on the mean estimate of the slope and an assumption of a probit dose response relationship for the effects study corresponding to the taxonomic group for which the LOCs are exceeded. These slopes from surrogate species could over- or under-estimate potential risks.

7.0 Conclusions

Conclusions of this assessment are summarized in Table 7.1. The best available data suggest that atrazine will either have no effect or is not likely to adversely affect any of the assessed species either by direct toxic effects or by indirect effects resulting from effects to aquatic or terrestrial plants or aquatic animals.

Table 7.1. Summary of Effects Determinations For Six Listed Species			
Assessment Endpoint	**Species**	**Effects Determination**	**Basis for Determination**
Direct effects to listed species (Section 5.1)	All six assessed species	No Effect	No acute or chronic LOCs for endangered species are exceeded.
Indirect effects to listed species via reduction of <u>aquatic animals</u> as food supply (Section 5.2.2.)	Shortnose sturgeon, loggerhead turtle, Kemp's ridley turtle, green turtle, leatherback turtle	Not likely to adversely affect	Acute LOCs are exceeded for some animals that are food items of the assessed species. However, the low magnitude of potential effects on any one species, the low number of dietary species potentially affected (indicated by LOC exceedances) relative to the number potentially consumed by the assessed species, and the conservative nature of the EECs used to derive RQs for organisms in flowing water systems suggests that the potential effects to the food supply of the assessed species constitutes an insignificant effect.[a]
	Dwarf wedgemussel	No effect	No acute or chronic LOCs are exceeded.
Indirect effects to listed species via reduction of <u>aquatic plants</u> as food items or primary productivity (Section 5.2.4.)	All six assessed species	Not likely to adversely affect	No known obligate relationship between the assessed species and any single aquatic plant species exists, and short-term and long-term atrazine concentrations were estimated to be lower than established thresholds for community-level effects to aquatic vegetation.
Indirect effects to listed species via direct effects on riparian areas required to maintain acceptable water quality and spawning habitat (Section 5.2.5.)	Shortnose sturgeon and each of the four assessed sea turtles	Not likely to adversely affect	Acreage of riparian habitat expected to be sensitive to atrazine is sufficiently low in the Chesapeake Bay watershed such that potential impacts of atrazine to sensitive riparian buffers are not expected to result in a measurable effect to the assessed species that reside in the main stem of the Chesapeake Bay and the Major river systems. Therefore, potential effects to riparian areas from use of atrazine are expected to constitute an insignificant effect [a].
	Dwarf wedgemussel	Not likely to adversely affect	Landcover data surrounding watersheds of dwarf wedgemussel habitats suggest that riparian area exposure to atrazine is expected to be minimal and/or that the predominant riparian area adjacent to waters of dwarf wedgemussel habitats is not expected to be sensitive to atrazine. Therefore, potential effects to the dwarf wedgemussel from effects to riparian areas are expected to constitute an insignificant effect.[a]

[a] <u>Significance of Effect</u>: Insignificant effects are those that cannot be meaningfully measured, detected, or evaluated in the context of a level of effect where take occurs for even a single individual

8.0 References

Allan, JD. (1995). Stream ecology: structure and function of running waters. Chapman and Hall, NY.

Armstrong, D. E., C. Chester and R. F. Harris 1967. Atrazine hydrolysis in soil. Soil Sci. Soc. Amer. Proc. 31:61-66.

BEAD. 2003. Assessment of Potential Mitigation Measures for Atrazine, 2003, Biological and Economic Analysis Division (BEAD), Office of Pesticide Programs, U.S. Environmental Protection Agency.

Baker, D. B., K. A. Kieger, R. P. Richards and J. W. Kramer. 1985. Effects of intensive agricultural land use on regional water quality in northwestern Ohio, p. 201-207. *In*: US EPA. Perspectives on nonpoint source pollution, proceedings of a national conference, Kansas City, MO. EPA-440/5-85-001. XI – 75

Bartell, S.M., K.R. Campbell, C.M. Lovelock, S.K. Nair, and J.L. Shaw. 2000. Characterizing aquatic ecological risk from pesticides using a diquat dibromide case study III. Ecological Process Models. Environ. Toxicol. Chem. 19(5):1441-1453.

Bartell, S.M., G. Lefebvre, G. aminski, M. Carreau, and K.R. Campbell. 1999. An ecosystem model for assessing ecological risks in Quebec rivers, lakes, and reservoirs. Ecol. Model. 124:43-67.

Bennett H.H, (1939). Soil Conservation. New York, New York, 993 pp.

Byles, R.A. (1988) The behavior and ecology of sea turtles *Caretta caretta* and *Lepidochelys kempi*, in Chesapeake Bay. PhD Dissertation. College of William and Mary, Williamsburg, VA.

Capel, P. D., M. Lin and P. J. Wotzka. 1994. Pesticides in rains in Minnesota 1991-1993: An interim report. Minnesota Dept. Agr., Agronomy Serv. Div., p. 26. (Additional pages on 1994 data).

CBP (Chesapeake Bay Program). 2006. Data obtained from http://www.chesapeakebay.net/wspv31/. Accessed in August, 2006.

CDMS search, 2006, https://premier.cdms.net/webapls/

Cloern, J.E.. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. Continental Shelf Research 7(11-12): 1367-1381.

Colby, S.R., E.R. Hill, N.E. Humberg, L.M. Kitchen, R.G. Lym, W.J. McAvoy, R. Prasad. (1989) The Herbicide Handbook of the Weed Science Society of America. 6th Ed. Champaign Illinois.

Curran, W., Professor of Weed Science, Pennsylvania State University, Personal Communication with Nicole Zinn, May 17, 2006.

Davies, P. E., L. S. J. Cook and J. L. Barton. 1994. Triazine herbicide contamination of Tasmanian streams: Sources, concentrations and effects on biota. Aust. J. Mar. Freshwater Res. 45:209-226. (MRID 45020003).

Davis B. N. K., M. J. Brown, A. J. Frost, T. J. Yates, R. A. Plant. (1994). The Effects of Hedges on Spray Deposition and on the Biological Impact of Pesticide Spray Drift. Ecotoxicology and Environmental Safety. 27(3):281-293.

DeAngelis, D.L., S.M. Bartell, and A.L. Brenkert. 1989. Effects of nutrient recycling and food-chain length on resilience. Amer. Nat. 134(5):778-805.

de Blois S., G. Domon, A. Bouchard (2002). Factors affecting plant species distribution in hedgerows of southern Quebec. Biological Conservation 105(3): 355-367.

Dodd, C.K., Jr. (1988) Synopsis of the biological data on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758). Fish and Wildlife Service Biological Report 88(14). 110pp.

Duron, M. 1978. Contribution a letude de la biologie de Dermochelys coriacea (Linne) dans les pertuis Charentais. These, Doct. Biol. Anim., Univ. Bordeaux. 112p. Cited in NOAA Fisheries (http://swr.nmfs.noaa.gov/PIR/feis/Appendix%20C.PDF)

Ernst, C.H., Lovich, J.E. and Barbour R.W. (1994) Turtles of the United States and Canada. Smithsonian Institution Press, Washington, D.C.

Everest, F.H., R.L. Beschta, J.C. Scrivener, K.V. Koski, J.R. Sedell, C.J. Cederholm. (1987). Fine sediments and salmonid production: a paradox. p. 98-142. In E.O. Salo and T.W. Cundy [ed.] Proceedings of the Symposium on Streamside Management: Forestry and Fishery Interactions. University of Washington, Seattle, WA.

Fish and Wildlife Health Program. 2004. Final Report Submitted to National Marine Fisheries Service. Contract #: 40EMNP200722.

Fleming, W., D. Galt, J. Holechek. (2001). Ten steps to evaluate rangeland riparian health. Rangelands 23(6):22-27.

Forget-Leray, J., Landriau, I., Minier, C., and Leboulenger, F. (2005). Impact of Endocrine Toxicants on Survival, Development, and Reproduction of the Estuarine Copepod Eurytemora affinis (Poppe). *Ecotoxicol.Environ.Saf.* 60: 288-294. EcoReference No.:

80951

Foreman, W.T., Majewski, M.S., Goolsby, D.A., Wiebe, F.W., and Coupe, R.H. 2000. Pesticides in the atmosphere of the Mississippi River Valley, Part II – air. Sci. Tot. Environ. 248. 135-146.

Frank, R., H. E. Braun, M. V. Holdrinet, G. J. Sirons and B. D. Ripley. 1982. Agriculture and water quality in the Canadian Great Lakes Basin. V. Pesticide use in 11 agricultural watersheds and presence in stream water 1975-1977. J. Environ. Qual. 11:497-505.

Frank, R. and G. J. Sirons. 1979. Atrazine: Its use in corn production and its loss to stream waters in southern Ontario, 1975-1977. Sci. Total Environ. 12:223-239.

Frank, R., G. J. Sirons, R. L. Thomas and K. McMillan. 1979. Triazine residues in suspended solids (1974-1976) and water (1977) from the mouths of Canadian streams flowing into the Great Lakes. J. Great Lakes Res. 5:131-138. XI – 79

Giddings, Jeffrey M., Todd A. Anderson, Lenwood W. Hall, Jr., Ronald J. Kendall, R. Peter Richards, Keith R. Solomon, and W. Martin Williams, "Aquatic Ecological Risk Assessment of Atrazine – A Tiered Probabilistic Approach, A Report of an Expert Panel," Novartis Crop Protection, Inc., Greensboro, NC, Section 5.3.1.5, page 147, June 23, 2000.

Gilbert, C.R. 1989. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic Bight)--Atlantic and shortnose sturgeons. U.S. Fish Wildl.Serv. Biol. Rep. 82(11.122). U.S. Army Corps of Engineers TR EL-82-4. 28 pp.

Glenn, S., Associate Professor, University of Maryland College Park, Personal communication with Nicole Zinn, May 15, 2006.

Hagood, S., Professor, Virginia Polytechnic Institute and State University, Personal communication with Nicole Zinn, May 16, 2006.

Hall, J. K., M. Pawlus and E. R. Higgins. 1972. Losses of atrazine in runoff water and soil sediment. J. Environ. Qual. 1:172-176.

Hörmann, W. D., J. C. Tournayre and H. Egli. 1979. Triazine herbicides residues in central European streams. Pest. Monit. J. 13:128-131.

Jacinthe P. -A., R. Lal, L.B. Owens, D. L. Hothem (2004). Transport of labile carbon in runoff as affected by land use and rainfall characteristics. Soil and Tillage Research 77(2):111-123.

Jobin, B., C. Boutin, J.L. DesGranges. (1997). Effects of agricultural practices on the flora of hedgerows and woodland edges in southern Quebec. Can J Plant Sci 77:293-299.

Kadoum, A. M. and D. E. Mock. 1978. Herbicide and insecticide residues in tailwater pits: Water and pit bottom soil from irrigated corn and sorghum fields. J. Agric. Food Chem. 26:45-50.

Kaul, M., T. Kiely, and A. Grube, 2005. Triazine pesticides usage data and maps for cumulative risk assessment, D317992. Unpublished EPA report.

Kaul, M. and J. Carter, 2005, Use Sites for Blanco, Hays, and Travis Counties in Texas for Atrazine, Simazine, Carbaryl, Metolachlor, Diazinon, and Prometon (D322226).

Kenneth Rygwelski, 2002, personal communication regarding materials submitted for inclusion in the first draft of the Lake Michigan Lake Wide Management Plan, LaMP 2002

Keinath, J.A., Musick, J.A. and R.A. Byles. (1987) Aspects of the biology of Virginia's sea turtles: 1979-1986. Virginia J. Sci. 38: 329-336.

Kimmel, T. L. (2004) Sea Turtle Tagging and Health Assessment Study in Maryland's Chesapeake Bay and Summary of Sea Turtle Strandings in Maryland, 1991 to 2003. Maryland Department of Natural Resources, Cooperative Oxford Laboratory

Kimmel, Tricia. (2006) Maryland Department of Natural Resources. Personal Communication.

Kimmel, T.L., Driscoll, C. P. and J. R. Brush (2006) A Five Year Summary of Sea Turtle Tagging Data From an In-Water Study in the Maryland Portion of the Chesapeake Bay: Possible Evidence of Site Fidelity. Maryland Department of Natural Resources, Cooperative Oxford Laboratory, Oxford, Maryland, USA

Kleijn, D., G.I. Snoeijing. (1997). Field boundary vegetation and the effects of agrochemical drift: botanical change caused by low levels of herbicide and fertilizer. Journal of Applied Ecology 34: 1413-1425.

Koch H., P. Weisser, M. Landfried. (2003). Effect of drift potential on drift exposure in terrestrial habitats. Nachrichtenbl. Deut. Pflanzenschutzd. 55(9): S. 181-188.

Lutcavage, M., P.L. Lutz. 1986. Metabolic Rate and Food Energy Requirements of the Leatherback Sea Turtle, Dermochelys coriacea. Copeia 1986 (3): 796-798. Cited in NOAA Fisheries (http://swr.nmfs.noaa.gov/PIR/feis/Appendix%20C.PDF)

Lutz, P.L. and Musick, J.A., *et al.* (1997) The Biology of Sea Turtles. CRC Press, LLC. Washington, D.C.

Majewski, M.S., Foreman, W.T., and Goolsby, D.A. 2000. Pesticides in the atmosphere of the Mississippi River Valley, Part I – rain. Sci. Tot. Enviorn. 248. 201-212.

Mansfield, Kate. (2006) Virginia Institute of Marine Sciences. Personal Communication.

Maryland Department of Natural Resources. 2006. Personal Communication with James M. McCann, Zoologist on July 25, 2006.

Moody, J. A. and D. A. Goolsby. 1993. Spatial variability of triazine herbicides in the lower Mississippi River. Environ. Sci. Technol. 27:2120-2126.

Moore, A. and Lower, N. (2001). The Impact of Two Pesticides on Olfactory-Mediated Endocrine Function in Mature Male Atlantic Salmon (Salmo salar L.) Parr. *Comp.Biochem.Physiol.B* 129: 269-276. EcoReference No.: 67727

Muir, D. C. G., J. Y. Yoo and B. E. Baker. 1978. Residues of atrazine and n-deethylated atrazine in water from five agricultural watersheds in Quebec. Arch. Environ. Contam. Toxicol. 7:221-235.

Muir, M. Department of Natural Resources. 2006. Personal Communication. August 16, 2006.

Musick, J.A. (1979) Loggerhead. Proceedings of the Symposium on Endangered and Threatened Plants and Animals of Virginia.

Nelson R.L., M.L. McHenry, W.S. Platts. (1991). Mining, Chap 12 in Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats, Meehan, WR, ed. American Fisheries Society, Bethesda, MD.

NMFS (National Marine Fisheries Service and U.S. Fish and Wildlife Service). 1991. Recovery Plan for U.S. Population of Atlantic Green Turtle. National Marine Fisheries Service, Washington, D.C.

NMFS (National Marine Fisheries Service and U.S. Fish and Wildlife Service). 1991. Recovery Plan for U.S. Population of Loggerhead Turtle. National Marine Fisheries Service, Washington, D.C.

NMFS (National Marine Fisheries Service) and U.S. Fish and Wildlife Service. 1992. Recovery plan for the Kemp's ridley sea turtle (Lepidochelys kempii). National Marine Fisheries Service, St. Petersburg, Florida.

NMFS (National Marine Fisheries Service). 1998. Recovery Plan for the Shortnose Sturgeon (*Acipenser brevirostrum*). Prepared by the Shortnose Sturgeon Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland. 104 pages.

NMFS. 2004. Endangered species act biological opinion. April 19, 20004.

NOS, CCEHBR, Cooperative Oxford Laboratory, Oxford, Maryland, USA. Poster presented at the 26th Annual Symposium on Sea Turtle Conservation and Biology in Greece.

Ortiz R.M., R.M. Patterson, C.E. Wade, and F.M. Byers. 2000. Effects of actue fresh water exposure on water flux rates and osmotic responses in Kemp's ridley sea turtles (Lepidochelys kempi). Comp biochem Physiol. Part A 127: 81-87.

Pannill, 2006. Maryland Department of Natural Resources Forest Service. 2006. Personal communication on August 29, 2006.

Powers, K. Maryland Department of Natural Resources. 2006. Personal communication on August 14, 2006.

Richard, J. J., G. A. Junk, M. J. Avery, N. L. Nehring, J. S. Fritz and H. J. Svec. 1975. Residues in water. Pest. Monit. J. 9:117-123.

Roberts, G. C., G. J. Sirons, R. Frank and H. E. Collins. 1979. Triazine residues in a watershed in southwestern Ontario, Canada (1973-1975). J. Great Lakes Res. 5:246-255.

Rosgen, D.L. (1996) Applied Fluvial Geomorphology. Wildland Hydrology, Pagosa Springs, CO.

Schulz, A., F. Wengenmayer and H. M. Goodman. 1990. Genetic engineering of herbicide resistance in higher plants. Plant Sci. 9:1-15.

Schippers P., W. Joenje (2002). Modelling the effect of fertiliser, mowing, disturbance and width on the biodiversity of plant communities of field boundaries. Agriculture, Ecosystems & Environment 93(1-3):351-365.

Seney, E. E. (2003) Historical Diet Analysis of Loggerhead (Caretta caretta) and Kemp's ridley (Lepidochelys kempi) Sea Turtles in Virginia. A Thesis Presented to the Faculty of the School of Marine Science, The College of William and Mary in Virginia.

Sweeny, 2006. Personal communication with Jeff Sweeny of the Chesapeake Bay Tributaries Strategy Program, July 14th, 2006.

Tierney D, Christensen B, Mattan C, and Dando C. 1999. Atrazine voluntary monitoring program analytical results 1993 – 1998. December 13, 1999. Submitted by Novartis.

U.S. Department of Agriculture (USDA), Natural Resources Conservations Service (NRCS), 2000. Conservation Buffers to Reduce Pesticide Losses. Natural Resources Conservation Service. Fort Worth, Texas. 21pp.

USDA. 2004. Agricultural Chemical Usage. 2003 Nursery and Floriculture Summary. September, 2004. Available on-line at http://usda.mannlib.cornell.edu/usda/current/AgriChemUsNursFlo/AgriChemUsNursFlo-09-15-2004.pdf

U.S. EPA. 1993. Wildlife Exposure Factors Handbook. EPA/600?R-93/187a. Office of Research and Development, Washington DC.

U.S. EPA. 1998. Framework for Ecological Risk Assessment. EPA/630/R-95/002F April 1998. Published on May 14, 1998, Federal Register 63(93):26846-26924.

U.S. EPA. 2003a. Interim Reregistration Eligibility Decision (IRED) for Atrazine, Case No. 0062, January 2003, EPA/OPP, Web address: http://www.epa.gov/oppsrrd1/REDs/atrazine_ired.pdf

U.S. EPA. 2003b. Biological Evaluation for the Issuance of Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries. April 25, 2003. Region III, Chesapeake Bay Program Office.

U.S. EPA. 2003c. Ambient aquatic life water quality criteria for atrazine - revised draft. EPA-822-R-03-023. October, 2003.

U.S. EPA. 2003d. White paper on potential developmental effects of atrazine on amphibians. May 29, 2003. Office of Pesticide Programs, Washington D.C. Available at http://www.epa.gov/scipoly/sap.

U.S. EPA. 2003e. Atrazine MOA Ecological Subgroup: Recommendations for aquatic community Level of Concern (LOC) and method to apply LOC(s) to monitoring data. Subgroup members: Juan Gonzalez-Valero (Syngenta), Douglas Urban (OPP/EPA), Russell Erickson (ORD/EPA), Alan Hosmer (Syngenta). Final Report Issued on October 22, 2003.

U.S. EPA. 2004. Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs. Office of Prevention, Pesticides, and Toxic Substances. Office of Pesticides Programs. Washington, D.C. January 23, 2004.

U.S. EPA. 2006a. Risks of Atrazine Use to Federally Listed Endangered Barton Springs Salamanders (*Eurycea sosorum*). Pesticide Effects Determination. Office of Pesticide Programs, Environmental Fate and Effects Division. August 22, 2006.

U.S. EPA. 2006b. Cumulative Risk Assessment for the Chlorinated Triazines. Office of Pesticides Programs. EPA-HQ-OPP-2005-0481. Washington, D.C. March 28, 2006.

U.S. EPA. 2006c. Memorandum from Special Review and Reregistration Division to Environmental Fate and Effects Division: Errata Sheet for Label Changes Summary Table in the January 2003 Atrazine IRED. June 12, 2006.

U.S. Fish and Wildlife Service and National Marine Fisheries Service. 1998. Endangered Species Consultation Handbook: Procedures for Conducting Consultation and Conference Activities Under Section 7 of the Endangered Species Act. Final Draft. March 1998.

U.S. FWS. 1993. Dwarf wedgemussel (alasmidonta heterodon) recovery plan. February 08, 1993.

U.S. FWS. 2006. http://www.fws.gov/midwest/mussel/current_threats.html. Accessed July 6, 2006.

USFWS/NMFS. 2004a. 50 CFR Part 402. Joint Counterpart Endangered Species Act Section 7 Consultation Regulations; Final Rule. FR 47732-47762.

USFWS/NMFS. 2004b. Letter from USFWS/NMFS to U.S. EPA Office of Prevention, Pesticides, and Toxic Substances. January 26, 2004. (http://www.fws.gov/endangered/consultations/pesticides/evaluation.pdf).

Virginia Department of Forestry. 2004. Operational Forestry Herbicide Use. Virginia Forestry Herbicide User Group. Chesapeake Bay & Non-Chesapeake Bay Watershed Counties. Available on-line at http://www.dof.virginia.gov/mgt/herbicide-03-use.shtml.

Virginia Department of Game and Inland Fisheries (VA DGIF). 2006. DWM_locations_dist1783. Vector digital data. Acquired August 01, 2006.

Waldron, A. C. 1974. Pesticide movement from cropland into Lake Eire. US EPA, Athens, GA, EPA-660/2-74-03.

Wallace P, C.L. Williams, F.V. Paladino, S.J. Morreale, R.T. Lindstrom, and J.R. Spotila. 2005. Bioenergetics and diving activity of internesting leatherback turtles Dermochelys coriacea at Parque Nacional Marino Las Baulas, Costa Rico. J. Exp. Biol. 208: 3873 – 3884.

Wauchope, R. D. 1978. The pesticide content of surface water draining from agricultural fields – a review. J. Environ. Qual. 7:459-472.

Wieser, C. M. and T. Gross. 2002. Determination of potential effects of 20 day exposure of atrazine on endocrine function in adult largemouth bass (Micropterus salmoides). Prepared by University of Florida, Wildlife Reproductive Toxicology Laboratory, Gainesville, FL, Wildlife No. NOVA98.02e; submitted by Syngenta Crop Protection, Inc., Greensboro, NC. (MRID No. 45622304).

Weissing F.J., J. Huisman. (1994). Growth and Competition in a Light Gradient. Journal of Theoretical Biology 168(3):323-336.

Williams, W. M., Harbourt, C. M., Matella, M. K., Ball, M. H., and Trask, J. R., 2004. Atrazine Ecological Flowing Water Chemical Monitoring Study in Vulnerable Watersheds Interim Report: Watershed Selection Process. Waterborne Environmental, Inc. (WEI). WEI Report No. WEI 936.32. pp 57.

Wolke, R.E., Brooks, D.R. and A George. (1982) Spirorchidiasis in loggerhead sea turtles (Caretta caretta). Pathology Journal of Wildlife Diseases 18(2): pp. 175-185. Wildlife Disease Association.

Wu, T. L., D. L. Correl and H. E. H. Remenapp. 1983. Herbicide runoff from experimental watersheds. J. Environ. Qual. 12:330-336. XI - 92

Wu, T. L. 1981. Atrazine residues in estuarine water and the aerial deposition of atrazine into Rhode River, Maryland. Water, Air, Soil Poll. 15:173-184.

Zinn, N. and A. Jones, 2006, Refined Usage Data for Atrazine in Alabama, Maryland, Pennsylvania, and Virginia, D328786.

Zug, G.R., and G.H. Balazs. (1985) Skeletochronological age estimates for Hawaiian green turtles. Marine Turtle Newsl. (33):9-10.