GEORGIA DOT RESEARCH PROJECT 18-06

FINAL REPORT

REVIEW OF SPECIAL PROVISIONS AND OTHER CONDITIONS PLACED ON GDOT PROJECTS FOR IMPERILED SPECIES PROTECTION

VOLUME I

Georgia Department of Transportation

OFFICE OF PERFORMANCE-BASED MANAGEMENT AND RESEARCH

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Georgia has numerous protected freshwater species, which means that the Georgia Department of Transportation (GDOT) must frequently consult with federal and state agencies to identify measures to avoid, minimize and mitigate impacts to imperiled aquatic organisms. Some of these measures, such as restrictions on in-water work during the reproductive season, impose substantial costs on GDOT projects, but their efficacy has not been thoroughly evaluated. The current system also provides limited flexibility. The research team has developed a system for assessing the impact of road construction projects on imperiled freshwater species that accounts for project characteristics, site characteristics, and species sensitivity. Called the “Total Effect Score” (TES), it is based on a comprehensive assessment of the tolerances and traits of 111 freshwater species and a thorough review of the literature on the efficacy of construction and post-construction BMPs. It employs a risk-based system to assess construction-phase effects and post-construction effects over a 50-year time horizon, making it possible to identify tradeoffs among alternative management practices. Additionally, the research team developed a template for a programmatic agreement (PA) that uses the TES as the basis for a streamlined system for evaluating projects. The PA is intended to cover both informal and formal consultation under a single system, which should reduce consultation time and increase predictability. To support the adoption of the PA, the research team conducted a biological assessment of all species. Adoption of the PA and the TES system should provide substantial cost savings for GDOT while improving outcomes for federally and state protected freshwater species.

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GDOT Research Project 18-06

Final Report

REVIEW OF SPECIAL PROVISIONS AND OTHER CONDITIONS PLACED ON GDOT PROJECTS FOR IMPERILED AQUATIC SPECIES PROTECTION VOLUME I

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The contents of this report reflect the views of the authors who are responsible for the facts and the accuracy of the data presented herein. The contents do not necessarily reflect the official views or policies of the Georgia Department of Transportation or the Federal Highway Administration. This report does not constitute a standard, specification, or regulation.

SI* (MODERN METRIC) CONVERSION FACTORS

APPROXIMATE CONVERSIONS TO SI UNITS

Symbol	When You Know	Multiply By	To Find	Symbol
		LENGTH		
in	inches	25.4	millimeters	mm
ft	feet	0.305	meters	m
yd	yards	0.914	meters	m
mi	miles	1.61	kilometers	km
		AREA		
in^2	square inches	645.2	square millimeters	mm^2
ft^2	square feet	0.093	square meters	m^2
yd^2	square yard	0.836	square meters	m^2
ac	acres	0.405	hectares	ha
mi^2	square miles	2.59	square kilometers	km^2
		VOLUME		
fl oz	fluid ounces	29.57	milliliters	mL
gal	gallons	3.785	liters	L
ft^3	cubic feet	0.028	cubic meters	m^3
yd^3	cubic yards	0.765	cubic meters	m^3
		NOTE: volumes greater than 1000 L shall be shown in m^3		
		MASS		
oz	ounces	28.35	grams	g
lb	pounds	0.454	kilograms	kg
T	short tons (2000 lb)	0.907	megagrams (or "metric ton")	Mg (or "t")
		TEMPERATURE (exact degrees)		
^oF	Fahrenheit	5 (F-32)/9 or (F-32)/1.8	Celsius	^oC
		ILLUMINATION		
fc	foot-candles	10.76	lux	lx
fl	foot-Lamberts	3.426	candela/m^2	cd/m^2
		FORCE and PRESSURE or STRESS		
lbf	poundforce	4.45	newtons	N
lbf/in^2	poundforce per square inch	6.89	kilopascals	kPa

APPROXIMATE CONVERSIONS FROM SI UNITS

Symbol	When You Know	Multiply By	To Find	Symbol
		LENGTH		
mm	millimeters	0.039	inches	in
m	meters	3.28	feet	ft
m	meters	1.09	yards	yd
km	kilometers	0.621	miles	mi
		AREA		
mm^2	square millimeters	0.0016	square inches	in^2
m^2	square meters	10.764	square feet	ft^2
m^2	square meters	1.195	square yards	yd^2
ha	hectares	2.47	acres	ac
km^2	square kilometers	0.386	square miles	mi^2
		VOLUME		
mL	milliliters	0.034	fluid ounces	fl oz
L	liters	0.264	gallons	gal
m^3	cubic meters	35.314	cubic feet	ft^3
m^3	cubic meters	1.307	cubic yards	yd^3
		MASS		
g	grams	0.035	ounces	oz
kg	kilograms	2.202	pounds	lb
Mg (or "t")	megagrams (or "metric ton")	1.103	short tons (2000 lb)	T
		TEMPERATURE (exact degrees)		
^oC	Celsius	1.8C+32	Fahrenheit	^oF
		ILLUMINATION		
lx	lux	0.0929	foot-candles	fc
cd/m^2	candela/m^2	0.2919	foot-Lamberts	fl
		FORCE and PRESSURE or STRESS		
N	newtons	0.225	poundforce	lbf
kPa	kilopascals	0.145	poundforce per square inch	lbf/in^2

* SI is the symbol for the International System of Units. Appropriate rounding should be made to comply with Section 4 of ASTM E380. (Revised March 2003)

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PREFACE/EXECUTIVE SUMMARY

Georgia has well over a hundred protected freshwater species, which means that the Georgia Department of Transportation (GDOT) must frequently consult with federal and state agencies to identify measures to avoid, minimize and mitigate impacts to imperiled aquatic organisms. Some of these measures, such as restrictions on in-water work during the reproductive season, impose substantial costs on GDOT projects. There is a need for an assessment of the efficacy of these and other potential measures, an assessment of the sensitivities of the various imperiled taxa, and a system to provide the flexibility for GDOT to employ the most effective measures for a given project, location and species.

To meet this need, the research team has developed a system for assessing the impact of road construction projects on imperiled freshwater species that accounts for project characteristics, site characteristics, and species sensitivity. Called the "Total Effect Score" (TES), it is based on a comprehensive assessment of the tolerances and traits of 111 freshwater species and a thorough review of the literature on the efficacy of construction and post-construction best management practices. It employs an innovative, risk-based system to assess both direct and indirect construction-phase effects and post-construction effects over a 50-year time horizon, making it possible to identify tradeoffs among alternative management practices. For example, the system allows the user to compare the benefit of timing restrictions versus improved stormwater management practices, providing a great deal of flexibility to identify the most appropriate and cost-effective management tools. The system is implemented with a user-friendly Excel tool designed to use readily available inputs and provide outputs in the form needed to support existing systems.

Additionally, the research team developed a template for a programmatic agreement (PA) that uses the TES as the basis for a streamlined system for evaluating GDOT projects. The programmatic agreement is intended to cover both informal and formal consultation under a single system, which should substantially reduce consultation time and increase predictability. To support the adoption of the PA, the research team has also conducted a biological assessment of all 111 species, which was reviewed by a panel of 13 external experts. The actual PA and supporting biological opinion will need to be drafted by the US Fish and Wildlife Service, in cooperation with GDOT and other state and federal agencies, but the research team has supplied all of the essential information for preparing the official documents. The research team believes that adoption of the PA and the TES system will provide substantial cost savings for GDOT while improving outcomes for federally and state protected freshwater species.

This volume is the first in a series. The other volumes in the series are *Review of Special Provisions and Other Conditions Placed on GDOT Projects for Imperiled Species Protection Volumes II-IV.*

CHAPTER 1. INTRODUCTION

The Southeastern United States is a global hotspot of aquatic biodiversity, including hundreds of species of fish, mussels, crayfish, snails, amphibians, herpetofauna and insects that are considered imperiled. Georgia alone has over 100 species that are listed as protected under federal or state law. Georgia also has a high per-capita investment in road and bridge infrastructure, with dozens of large projects under development at any one time, and many of these have the potential to affect protected aquatic species. As a consequence, the Georgia Department of Transportation (GDOT) spends considerable time and resources consulting with federal and state agencies to avoid, minimize and mitigate impacts to imperiled aquatic organisms.

As part of the consultation process, US Fish and Wildlife Service (USFWS) and Georgia Department of Natural Resources (DNR) may recommend special provisions, which in turn may be incorporated into requirements for contractors by GDOT. Such provisions may include (among others) limits on in-water use of equipment, sedimentation minimization practices, noise restrictions, and timing restrictions that prohibit work below the water surface during the spawning season of protected species. In early consultation the USFWS may also make design recommendations (e.g., use of stormwater BMPs, use of bridges rather than culverts). While these measures are based on sound science and are presumed to be necessary for minimizing the impact to sensitive species, there has been no comprehensive examination of the benefits of these practices to the different species. It is possible that some provide relatively low benefit, and it is possible that other more beneficial actions have been overlooked. A comprehensive review of these practices is therefore warranted, particularly since some measures can impose substantial additional costs on construction.

For example, the majority of the conditions are associated with minimizing the impacts of the actual construction process by avoiding work during the spawning season and by minimizing sedimentation. However, many fish species can recover rapidly from occasional short-term, localized impacts, as long as there are healthy populations nearby. Conversely, improperly designed infrastructure may create long-term stressors; for example, upslope highway runoff directed to the stream without proper treatment can degrade water quality for decades. This suggests the possibility that some timing restrictions could be relaxed if enhanced stormwater measures are employed and extensive safeguards are taken to minimize sedimentation. However, this must be considered on a species-by-species basis. Timing restrictions are essential if a project affects the only known spawning habitat for an isolated population of a species.

The overarching goal of this project is to increase the efficiency of GDOT consultation and management for imperiled organisms while providing protection to species at a level equal to or higher than current practice. In the scoping phase (April-September 2017) the research team reviewed special conditions, assessed the need for additional data collection, established methodologies for the main project phase, and estimated the cost benefit of the full project. The findings are summarized below.

- The research team found the current system of consultation to develop special conditions could be greatly streamlined with the development of a programmatic agreement (PA) for aquatic species.
- The research team developed a preliminary decision tree method to assess whether timing restrictions were needed for each of 131 species, in each watershed (10-digit hydrologic unit code or HUC10) in Georgia.
- The research team determined that additional biological field data collection was not required to meet the objectives of the main project phase and was not feasible within a reasonable budget.
- The team decided that the project should be focused on freshwater aquatic species. The research team consulted with NOAA representatives on whether to include Atlantic sturgeon (*Acipenser oxyrinchus*) but found that this species was already the focus of a dedicated programmatic agreement under development.
- The team determined that construction-phase special conditions could be divided into 16 categories, and recommended conducting a literature review of the effectiveness of each category, and of individual practices within categories where warranted, in the main phase of the project.
- The team noted that effective management of post-construction stormwater runoff is critical to protect aquatic species. Existing literature already shows strong evidence of negative effects of stormwater discharges from large areas of impervious surfaces, but limited effects of direct drainage from bridge decks alone. The research team proposed to synthesize this literature to assess stormwater impacts.

- The team also recommended that for practical purposes, the main project should be limited to special provisions and conditions recommended by USFWS and DNR under the auspices of the US Endangered Species Act and the Fish and Wildlife Coordination Act, but not the US Army Corps of Engineers under the US Clean Water Act.

Based on the results of the scoping phase, the research team proposed six tasks for the main project phase:

- Task 1. Review each aquatic species to determine whether timing provisions are warranted.
- Task 2. Review all special provisions governing construction activities.
- Task 3. Review stormwater standards and best management practices.
- Task 4. Develop recommendations for a programmatic agreement for aquatic species.
- Task 5. Write a biological assessment to serve as the basis for a biological opinion associated with the proposed programmatic agreement.
- Task 6. Prepare final report.

This document constitutes the final report of the main project phase, and reports on the results of tasks 1-5.

The research team deviated from the planned tasks in one key respect: the team expanded Task 1 from a narrow focus on timing restrictions into the development of a comprehensive, flexible system for identifying the optimal mix of avoidance, minimization and mitigation measures for protected species. This system involves the calculation of a "Total Effect Score" that is the sum of detrimental effects on an imperiled species from a project or sub-project. These include direct effects of in-water construction, effects of sedimentation from erosion of disturbed soil, and post-construction effects of stormwater runoff. The system allows for tradeoffs among the use of timing restrictions, increased erosion and sedimentation control, and enhanced stormwater management, as long as the total effect score is kept under a project-specific maximum. The Total Effect Score (TES) and its components were developed in close collaboration with personnel from GDOT, USFWS, and DNR, with input from the Federal Highway Administration.

The research team also made some modifications to the species included in the review and biological assessment, again in consultation with partner agencies (Table 1). First, the research team confined the list to currently listed federal and state species and excluded federally petitioned species (unless also state-listed), even though the authors initially planned to include petitioned species in the scoping phase. The exception is the lake sturgeon (*Acipenser fulvescens*), which is currently under consideration for federal protection and is included here. The research team also excluded species presumed to be extirpated by USFWS and DNR, with the exception of the Suwannee moccasinshell, *Medionidus walkeri*¸ which was included at the request of USFWS. The research team omitted four cave species (*Eurycea wallacei*, Georgia blind salamander; *Gyrinophilus palleucus*, Tennessee cave salamander; *Cambarus cryptodytes*, Dougherty Plain cave crayfish; and *Typhlichthys subterraneus*, southern cavefish), since their cave habitat generally lacks a direct connection to streams and rivers, rendering the TES approach inappropriate. Furthermore, these species are sufficiently rare and sensitive that any projects that could potentially affect them should undergo individual consultation. The research team also omitted the two plant species (*Hymenocallis coronaria*, shoals spiderlily, and *Sagittaria secundifolia*, Kral's water plantain), both because they're handled differently under the ESA and because the TES approach isn't appropriate for their biology. The research team added the rayed creekshell, *Strophitus radiatus*, which is a state-threatened species that was accidentally omitted from the scoping phase report.

The research team made several other nominal changes to the species list for reasons of systematics and nomenclature. The research team considered the Apalachicola alligator snapping turtle, sometimes known as *Macrochelys apalachicolae,* to be a subspecies of *M. temminckii*, following USFWS and DNR practice, even though some authorities consider it to be a separate species and it was listed as such in the scoping report. The scientific name of the Apalachicola floater was corrected from *Anodonta heardi* to *Utterbackiana heardi*, to reflect recent taxonomic research. Similarly, the Alabama creekmussel was corrected from *Strophitus connasaugaensis* to *Pseudodontoideus connasaugaensis*. Finally, the research team refers to *Noturus munitus* as the Coosa madtom (*Noturus* sp. cf. *N. munitus*) rather than frecklebelly madtom, because it is treated as a separate species in the scientific literature and by the DNR, even though it has not been officially described.

The remainder of this document is organized into six chapters, followed by three appendices. Chapter 2 provides an overview of the total effect score (revised Task 1). Chapter 3 provides details of the sediment effect score and incorporates a review of special provisions for construction activities (Task 2). Chapter 4 provides details of the post-construction effect score and includes a review of stormwater management and best management practices (Task 3). Chapter 5 provides an annotated template for a programmatic agreement that would implement the TES and cover GDOT projects that affect the species listed in this report (Task 4). Chapter 6 describes the effects of road construction activities on aquatic organisms and serves as the basis for the individual biological assessments, which are provided in Appendix C. Chapter 7 is a brief concluding section.

The work here represents the collective contributions of more than 40 people. In addition to the authors, contributors to the biological assessments included Zachary Butler, Carmen Candal, Kyle Connelly, David Lee Haskins, Max Kleinhans, Nicole Pontzer, Laura Rack, Robert Ratajcak, Shishir Rao, Edward Stowe, and Carol Yang. External expert reviewers of the biological assessments included Brett Albanese, Kyle Barrett, Darold Batzer, Kristen Cecala, William Ensign, Mary Freeman, Bernard Kuhajda, Paula Marcinek, Ani Popp, Matthew Rowe, James Stoeckel, Chris Taylor, and Jason Wisniewski. Agency partners who reviewed drafts, tested tools, and provided expert advice included Katy Allen, Federal Highway Administration, Mike Garner, GDOT, Jeffrey Garnett, GDOT, Chris Goodson, GDOT, Peter Maholland, USFWS, Brad McManus, GDOT, Paula Marcinek, GDNR, Eric Prowell, USFWS, and Carrie Straight, USFWS. Finally, Sarah Buckleitner performed copy editing and the final layout of this document. The research team is indebted to all of these individuals for their contributions to the success of this project.

Table 1. Species covered in this report.

Group	Genus	Species	Common name
amphibian	*Cryptobranchus*	*alleganiensis*	Eastern hellbender
amphibian	*Lithobates*	*capito*	gopher frog
amphibian	*Amphiuma*	*pholeter*	one-toed amphiuma
amphibian	*Notophthalmus*	*perstriatus*	striped newt
crayfish	*Cambarus*	*speciosus*	beautiful crayfish
crayfish	*Cambarus*	*unestami*	blackbarred crayfish
crayfish	*Distocambarus*	*devexus*	Broad River burrowing crayfish
crayfish	*Cambarus*	*howardi*	Chattahoochee crayfish
crayfish	*Cambarus*	*scotti*	Chattooga River crayfish
crayfish	*Cambarus*	*extraneus*	Chickamauga crayfish
crayfish	*Cambarus*	*cymatilis*	Conasauga blue burrower
crayfish	*Cambarus*	*coosawattae*	Coosawattee crayfish
crayfish	*Cambarus*	*doughertyensis*	Dougherty burrowing crayfish
crayfish	*Cambarus*	*fasciatus*	Etowah crayfish
crayfish	*Procambarus*	*verrucosus*	grainy crayfish
crayfish	*Cambarus*	*parrishi*	Hiwassee headwaters crayfish
crayfish	*Cambarus*	*strigosus*	lean crayfish
crayfish	*Cambarus*	*georgiae*	Little Tennessee crayfish
crayfish	*Procambarus*	*gibbus*	Muckalee crayfish
crayfish	*Cambarus*	*truncatus*	Oconee burrowing crayfish
crayfish	*Cambarus*	*harti*	Piedmont blue burrower
crayfish	*Procambarus*	*versutus*	sly crayfish
crayfish	*Cambarus*	*englishi*	Tallapoosa crayfish
dragonfly	*Gomphus*	*consanguis*	Cherokee clubtail
dragonfly	*Ophiogomphus*	*edmundo*	Edmund's snaketail
dragonfly	*Cordulegaster*	*sayi*	Say's spiketail
fish	*Alosa*	*alabamae*	Alabama shad
fish	*Cyprinella*	*xaenura*	Altamaha shiner
fish	*Percina*	*antesella*	amber darter
fish	*Enneacanthus*	*chaetodon*	blackbanded sunfish
fish	*Etheostoma*	*duryi*	blackside snubnose / black darter
fish	*Erimystax*	*insignis*	blotched chub
fish	*Cyprinella*	*caerulea*	blue shiner
fish	*Elassoma*	*okatie*	bluebarred pygmy sunfish
fish	*Lucania*	*goodei*	bluefin killifish
fish	*Pteronotropis*	*welaka*	bluenose shiner
fish	*Cyprinella*	*callitaenia*	bluestripe shiner
fish	*Percina*	*kusha*	bridled darter
fish	*Pteronotropis*	*euryzonus*	broadstripe shiner
fish	*Notropis*	*asperifrons*	burrhead shiner

fish	*Etheostoma*	*scotti*	Cherokee darter
fish	*Etheostoma*	*ditrema*	coldwater darter
fish	*Percina*	*jenkinsi*	Conasauga logperch
fish	*Macrhybopsis*	*etnieri*	Coosa chub
fish	*Noturus*	*sp. cf. N. munitus*	Coosa madtom
fish	*Percina*	*sciera*	dusky darter
fish	*Etheostoma*	*etowahae*	Etowah darter
fish	*Phenacobius*	*crassilabrum*	fatlips minnow
fish	*Hemitremia*	*flammea*	flame chub
fish	*Percina*	*lenticula*	freckled darter
fish	*Percina*	*aurolineata*	goldline darter
fish	*Etheostoma*	*parvipinne*	goldstripe darter
fish	*Etheostoma*	*chlorobranchium*	greenfin darter
fish	*Percina*	*crypta*	Halloween darter
fish	*Notropis*	*hypsilepis*	highscale shiner
fish	*Etheostoma*	*brevirostrum*	holiday darter
fish	*Acipenser*	*fulvescens*	lake sturgeon
fish	*Hybopsis*	*lineapunctata*	lined chub
fish	*Etheostoma*	*chuckwachatte*	lipstick darter
fish	*Noturus*	*eleutherus*	mountain madtom
fish	*Percina*	*smithvanizi*	muscadine darter
fish	*Fundulus*	*catenatus*	northern studfish
fish	*Ichthyomyzon*	*bdellium*	Ohio lamprey
fish	*Percina*	*squamata*	olive darter
fish	*Notropis*	*ariommus*	popeye shiner
fish	*Moxostoma*	*robustum*	robust redhorse
fish	*Etheostoma*	*rupestre*	rock darter
fish	*Notropis*	*scepticus*	sandbar shiner
fish	*Moxostoma*	sp. 2	sicklefin redhorse
fish	*Notropis*	*photogenis*	silver shiner
fish	*Percina*	*tanasi*	snail darter
fish	*Ameiurus*	*serracanthus*	spotted bullhead
fish	*Phenacobius*	*uranops*	stargazing minnow
fish	*Fundulus*	*bifax*	stippled studfish
fish	*Micropterus*	*notius*	Suwannee bass
fish	*Etheostoma*	*tallapoosae*	Tallapoosa darter
fish	*Percina*	*aurantiaca*	tangerine darter
fish	*Chrosomus*	*tennesseensis*	Tennessee dace
fish	*Etheostoma*	*trisella*	trispot darter
fish	*Etheostoma*	*vulneratum*	wounded darter
mussel	*Pseudodontoideus*	*connasaugaensis*	Alabama creekmussel
mussel	*Medionidus*	*acutissimus*	Alabama moccasinshell
mussel	*Elliptio*	*arca*	Alabama spike

mussel	*Alasmidonta*	*arcula*	Altamaha arcmussel
mussel	*Elliptio*	*spinosa*	Altamaha spinymussel
mussel	*Utterbackiana*	*heardi*	Apalachicola floater
mussel	*Fusconaia*	*masoni*	Atlantic pigtoe
mussel	*Medionidus*	*parvulus*	Coosa moccasinshell
mussel	*Elliptio*	*arctata*	delicate spike
mussel	*Amblema*	*neislerii*	fat threeridge
mussel	*Hamiota*	*altilis*	finelined pocketbook
mussel	*Pleurobema*	*hanleyianum*	Georgia pigtoe
mussel	*Medionidus*	*penicillatus*	Gulf moccasinshell
mussel	*Elliptio*	*purpurella*	inflated spike
mussel	*Pleurobema*	*pyriforme*	oval pigtoe
mussel	*Elliptoideus*	*sloatianus*	purple bankclimber
mussel	*Strophitus*	*radiatus*	Rayed Creekshell
mussel	*Ptychobranchus*	*foremanianus*	rayed kidneyshell
mussel	*Toxolasma*	*pullus*	Savannah lilliput
mussel	*Hamiota*	*subangulata*	shinyrayed pocketbook
mussel	*Pleurobema*	*decisum*	Southern clubshell
mussel	*Alasmidonta*	*triangulata*	southern elktoe
mussel	*Pleurobema*	*georgianum*	Southern pigtoe
mussel	*Medionidus*	*walkeri*	Suwannee moccasinshell
snail	*Leptoxis*	*foremani*	interrupted rocksnail
turtle	*Graptemys*	*pulchra*	Alabama map turtle
turtle	*Macrochelys*	*temminckii*	Apalachicola alligator snapping turtle
turtle	*Graptemys*	*barbouri*	Barbour's map turtle
turtle	*Glyptemys*	*muhlenbergii*	bog turtle
turtle	*Graptemys*	*geographica*	Northern map turtle
turtle	*Macrochelys*	*suwanniensis*	Suwanee alligator snapping turtle

CHAPTER 2. TOTAL EFFECT SCORE CALCULATION

The Total Effect Score (TES) is a metric of the aggregate effect of a GDOT project on an imperiled aquatic organism over a 50-year time horizon. It is intended to provide a flexible approach to meeting the avoidance, minimization and mitigation requirements under the Endangered Species Act, so GDOT can employ the most effective measures with greatest cost efficiency for a given species and project.

The basic procedure is:

1. Calculate the TES for a species based on project characteristics, site characteristics, species characteristics, and measures employed.
2. Compare the TES with the maximum effect score (MES). The maximum effect score is calculated individually for each project and each species.
3. If TES> MES, adjust the measures employed and recalculate TES. Repeat this until TES < MES for each species in the project area. If this is not possible for one or more species for a project, then the programmatic agreement cannot be used for that project.

The TES must be calculated separately for each species. If a project affects multiple stream reaches, the TES must be calculated separately for each reach, and for each species in each reach.

There are five components to the Effect Score:

1. Direct Impact Effect Score (DES). This is an estimate of the potential effect of physical in-stream construction activities on the local population. It can be greatly reduced by timing restrictions on in-water work.
2. Sediment Effect Score (SES). This is an estimate of the probability of a large sedimentation event and the effect of such an event. It is reduced by use of more advanced sedimentation AMMs.
3. Other Construction Effect Score (OCES). This incorporates construction-phase risks to species not captured in the above effect scores, such as hydraulic fluid leaks, or effects of herbicides. Since AMMs for managing these effects are required on all projects, this score is small and cannot be adjusted.
4. Post-Construction Effect Score (PCES). This is an estimate of the effect of stormwater runoff from impervious surfaces associated with the project over a 50-year time horizon. It is reduced by use of stormwater management BMPs.
5. Reconnection Effect Score (RES). This is only included in projects that reconnect species habitat through culvert removal or replacement. It requires a project-specific estimate of the area of reconnected habitat.

Each of these is described in detail in subsequent sections.

Figure 1 shows the how the different effect scores are calculated.

Species-level traits are shown in green in the figure. They are considered constants that are invariant to project location and characteristics. Once assigned, they will not change unless new scientific information becomes available, and all parties agree to update the PA accordingly. Site or project characteristics that are not adjustable are shown as grey in the figure. Adjustable control measures are shown as blue. Calculated values are shown as purple in the diagram, with intermediate calculations shown as light purple, and the effect scores shown as dark purple.

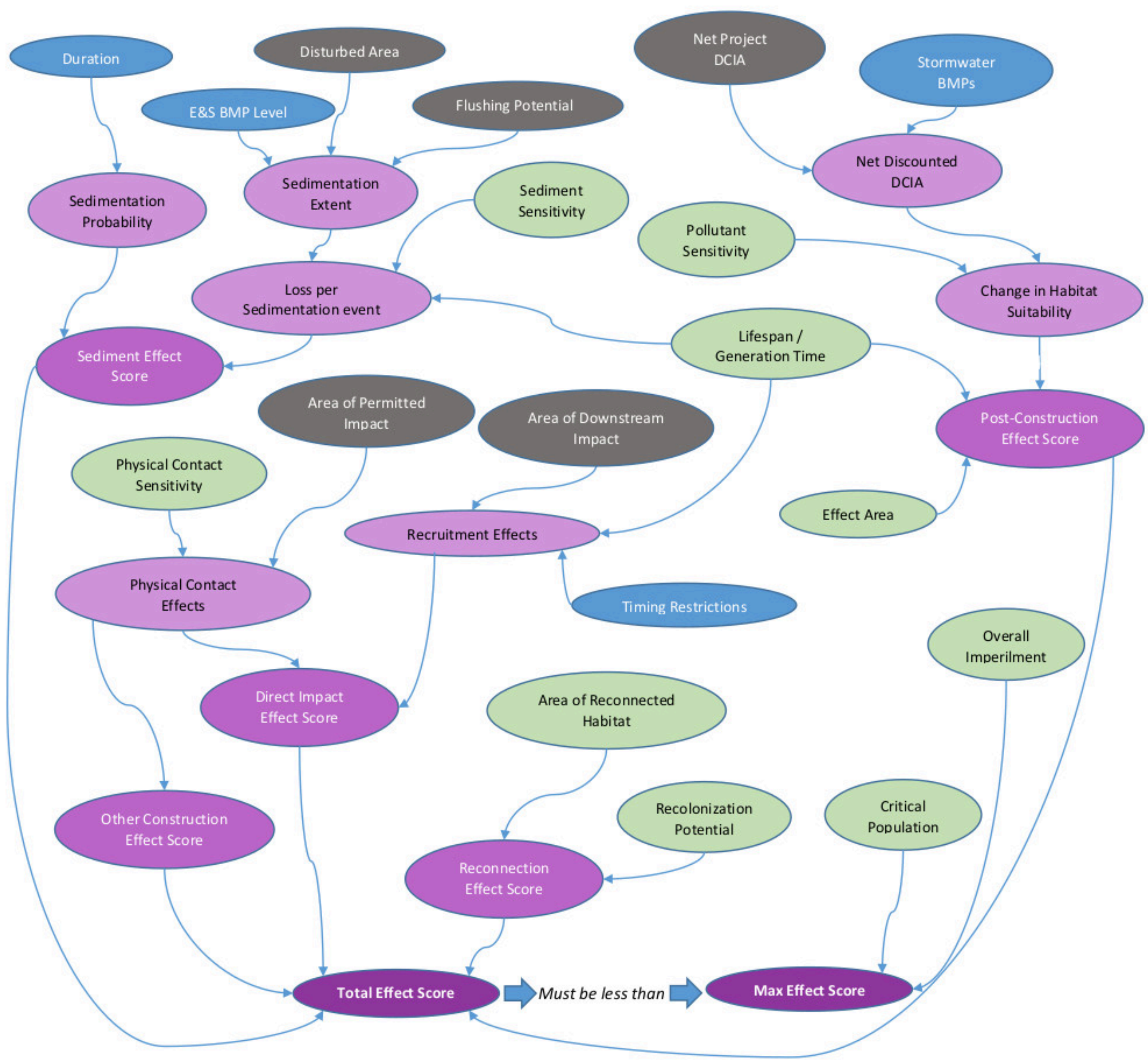

Figure 1. Diagram. Illustrating the calculation of Total Effect Score for a species

DIRECT IMPACT EFFECT SCORE

The Direct Impact Effect Score (DES) has three sub-components: (a) physical contact, which covers the effect of heavy equipment or falling debris directly harming organisms, (b) recruitment effects, which covers the failure of a year-class when work is conducted in water during the spawning/recruitment season, and (c) avoidable habitat destruction, which accounts for long-term loss of habitat due to channel modifications (concrete, stream straightening) that are not essential to the project. The calculation is:

DES = Physical contact effects + recruitment effects + avoidable habitat destruction

Physical contact effects = Area of permitted impact * physical contact sensitivity * migratory species timing restrictions * duration

Recruitment effects = Area of downstream impact * (1/lifespan) * sediment sensitivity * reproductive seasons * no local reproduction * disturbance rate

Avoidable habitat destruction = Lost habitat * (1/lifespan) * 50 years

Area of permitted impact is the area of the stream bed directly affected by falling debris, in-water use of heavy equipment, and other activities that have a direct, physical, destructive but short-term and localized effect on biota. It corresponds to the area directly beneath a bridge and other structures that extend over the water during the construction phase. It is the area of impacts permitted under the corresponding Clean Water Act Section 404 permit, or the area directly beneath a bridge and any other structures that extend over the water, whichever is larger.

Physical contact sensitivity is a function of a species' ability to avoid threats such as heavy equipment and falling debris. It is a species-level trait that can be either "high" or "medium".

- High sensitivity = a value of 0.75. This applies to organisms that have limited mobility and are not normally relocated in advance of projects. Examples include crayfish and dragonflies.
- Medium sensitivity = a value of 0.5. This applies to fish, which are mobile and can avoid some impacts, and mussels, which are normally relocated. The research team assumes that about 25% of mussels are not detected during the relocation process.

Migratory species timing restrictions. For migratory species that only pass through the region, if in-water work is not conducted during the migratory season, then physical contact effects are zero.

Mean lifespan is a species-level trait established from the literature. The research team assumes that when work is conducted in water during the spawning/recruitment season, there will be no successful reproduction within the action area, leading to the loss of a year class of recruits. The value of 1/lifespan represents the loss of a year class that would have been recruited into a reproductively active population of adults at carrying capacity. That is, when a population is at its equilibrium level, the number of recruits into the population will be equal to the natural mortality of adults. For habitat destruction, mean lifespan is used to calculate the number of generations of individuals lost over a 50-year period.

Duration. The number of years of active work on the project. It is never less than one but may be fractional if greater than one.

Disturbance rate is the frequency of events that cause a year-class failure. This is currently set at 0.5.

Area of downstream impact is the area of the stream potentially subjected to detrimental effects on recruitment when work is carried out during the spawning/recruitment season for a species. It includes the area of permitted impact and areas extending a modest distance downstream. The area of downstream impact is 3 times the area of permitted impact, or an area equivalent to the width of the stream multiplied by a length of 10 times the width of the stream (whichever is larger). For example, a 5 ft wide stream would have an area of downstream impact of at least 250 ft^2.

Sediment sensitivity is the same as for the sediment effect score, described in the next section.

Reproductive Seasons is the number of reproductive seasons over which work occurs. It may be reduced by timing restrictions.

Timing restrictions. Timing restrictions are a prohibition of in-water work during the reproductive season of a species that is present and uses the stream channel for reproduction.

No local reproduction. If a species does not use the stream channel for reproduction, this has a value of zero, which means that recruitment effects are zero. This applies to species that use terrestrial habitats, migratory species that pass through the area but do not spawn there, and species that use specialized habitats not present locally (e.g., an obligate spring-spawning species, for a project in a stream or river).

Lost habitat is always a custom number that represents the area of the stream bed rendered unsuitable as habitat for the species due to long-term modification (assumed to be 50 years). Only the habitat loss that is avoidable is included. “Avoidable” means there are feasible alternatives with smaller amounts of habitat loss. This value will often be zero.

SEDIMENT EFFECT SCORE

Most special provisions applied to GDOT construction projects are sediment-control best management practices, reflecting the fact that sedimentation has negative effects on the recruitment and survival of many imperiled freshwater organisms. This is addressed with a probabilistic approach, in which the research team first estimates the chance of a major failure of erosion control, and then multiply this by the expected loss of recruitment in the current or succeeding spawning season. A minimum level of erosion and sedimentation best management practices (E&S BMPs) is required for all projects. In many cases, it’s possible to add additional redundancy or employ more protective versions of E&S BMPs, which can reduce the probability and severity (extent) of events.

SES = Sedimentation Probability * Loss per Sedimentation Event

Sedimentation Probability = p_{event}* duration

Loss per Sedimentation Event = Sedimentation extent * Species sediment sensitivity * (1/lifespan) * No local reproduction

Sedimentation extent = (Soil loss/area) * disturbed area * (1/threshold depth) * Flushing potential * E&S BMP level

Sedimentation Probability is the expected number of events with a specified probability representing the likelihood of a major failure of erosion control that results in a large slug of sediment to the stream or river that buries the natural bed material in at least 2 cm of fine sediment (clay, silt, sand). The research team used a 2-year, 24-hour storm as a design event since it is typically capable of producing enough run off to generate large quantities of sediment without enhanced E&S control on construction sites. The annual exceedance probability of this event (50%) is multiplied by the project duration to determine the expected number of events. For example, for a 1-year project, this is 0.5, and for a 2.5-year project this is 1.25 (2.5*0.5). The research team noted that they could, with similar results, assume a rarer but larger event that would result in greater sedimentation extent.

$\boldsymbol{P_{event}}$ is the annual exceedance probability of the design event. This is set at 0.5.

Duration is the project duration in years. It is never less than 1 but may be fractional if greater than one.

Sedimentation extent is based on an estimate of soil loss in a major failure, spread over the surface of the stream to a biologically relevant threshold depth (2 cm). The extent is modified by the level of BMPs and the flushing potential of the stream (e.g. slope and drainage area).

Soil loss/area is an order of magnitude estimate of event soil loss based on the Revised Universal Soil Loss Equation (RUSLE). Soil loss is calculated on a per-area basis, and it is proportional to terrain characteristics and the predominant soil type. Calculated soil loss (mass/area) is multiplied by the project scale and a conversion factor to obtain sediment volume. Required inputs will be selected from a drop-down menu and include soil type, a characteristic slope of the terrain, and a length describing the longest path parallel to the slope before the gradient decreases.

Project Scale in this case means approximate extent of disturbed area, which will be directly input as the estimated area of the limits of disturbance.

E&S BMP level. The above sediment extent is based on the standard (minimum required) level of E&S BMPs. Higher E&S BMP levels reduce the extent:

- Standard: 0.6
- Advanced: 0.5
- Advanced-High: 0.4
- High: 0.3
- Very High: 0.2

The E&S BMP level is based on a framework that determines if applied BMPs are sufficient to achieve the high or very high categories. The framework is intended to provide a high degree of flexibility so that BMPs can be appropriately matched to site conditions and is described in detail in the BMP Level Determination section of the report.

Rationale: The research team assumes that an event equaling or exceeding the design storm event results in failure of standard E&S control practices. This general assumption is supported by evidence from the literature documenting a wide range and, in some circumstances, low efficiencies of E&S practices under normal conditions. Failure of common sediment barriers has been observed in an experimental test simulating a 2-year, 24-hour storm event (Whitman et al., 2018; 2019). The field capacity of E&S devices is highly uncertain due to the time-varying efficiencies from repetitive and cumulative loading between maintenance. This evidence coupled with substantial precedent in stormwater practice supports the use of a 2-year event as a reasonable design value.

However, restricting particular sediment generating activities, enhancing the level of source prevention (i.e. preventing erosion at the source), increasing storage requirements for intercepting practices (e.g. silt fence and sediment basins), and enhancing monitoring and maintenance can increase the efficiency of E&S control. The method for classifying E&S BMP level accounts for various levels of activity restriction, source prevention, interception, and monitoring and maintenance to account for the various efficiencies achieved by these four types of practices. Activity restriction has the highest efficiency followed by source prevention, interception, and monitoring and maintenance. The classification evaluates various combinations of these types, and the individual scores are based on a combination of empirical evidence in the literature and professional judgment.

Flushing potential is based on an index of specific stream power (i.e. stream power per unit bed width) within a reach, or the power available to do geomorphic work within a reach. Within the SES calculation, the extent of a sedimentation event is modified by flushing potential to account for reaches that might store large amounts of sediment for longer periods of time compared to reaches that might mobilize the sediment downstream relatively quickly. The research team used a stream power index that scales drainage area by slope (Flores et al., 2006) to delineate energy classifications and define flushing potential.

- $S > 0.025$ & $SA^{0.4} > 0.206$ (very high energy): 0.5
- $S > 0.025$ & $SA^{0.4} \leq 0.206$ (medium – high energy): 0.75
- $S < 0.025$ & $SA^{0.4} > 0.055$ (low – medium energy): 0.8
- $S < 0.025$ & $SA^{0.4} \leq 0.055$ (low energy): 1

Rationale: Scaling drainage area by slope ($SA^{0.4}$) has been derived as an index of specific stream power that was successful at delineating streams with different energy classifications (Flores et al., 2006). Thus, sediment extent is modified by this classification, which the research team terms "flushing potential," to account for variation among reaches to store or transport deposited sediment from and E&S control failure.

Specific stream power has been useful in a number of applications to estimate sediment transport and predict zones of erosion and deposition (Lammers and Bledsoe, 2018; Sholtes et al., 2018).

The classification system applied by Flores et al. (2006) generally delineates reaches as transport-limited, transitioning between transport-limited and supply-limited, and supply-limited, reflecting differences in energy dissipation and relative transport capacity (Montgomery and Buffington, 1997; Montgomery and MacDonal, 2002). While the research team does not intend to draw these distinctions for the rivers in Georgia, this system provides a useful means for discriminating among reaches of varying specific stream power. Consequently, it can be expected that lower energy reaches will store sediment in pools and along channel margins even after high flow events, such as annual peak floods (Rathburn and Wohl, 2002; Wohl and Cendrelli, 2000), and have a greater propensity to retain larger amounts of sediment in a reach for periods of time biologically relevant for aquatic organisms. Evidence of this is supported by a modeling experiment that simulated sediment transport in pool-riffle reach with a high-resolution 2-D hydrodynamic model showing fine sediment from pulsed and chronic events remained within the reach over year after the event (Maturana et al., 2013). In contrast, higher energy reaches have a higher capacity to mobilize fine sediment across a wider range of flow events compared to low energy reaches.

The research team recognizes that sediment dynamics will be highly variable in space and time among locations and even within a single channel reach. Ultimately, the flushing potential and the duration of a sediment impact within a reach will depend on multiple complex factors such as the ambient sediment supply, sequence of streamflow, relative size distribution of the input sediment disturbance relative to the parent material, and magnitude of the disturbance. Patterns of erosion and deposition are difficult to predict without resource intensive data collection and modeling efforts. However, the calculation proposed classifies a reach by the power available to transport sediment downstream, and the general capacity of its anticipated form to transport or store sediment.

Loss per sedimentation event assumes the loss of a year class due to sediment smothering eggs, filling interstitial spawning cavities, and otherwise degrading spawning habitat. Its calculation is analogous to that of recruitment effects for the DES. It is multiplied by 1/lifespan and the extent of sedimentation (see below), and then adjusted based on species sensitivity.

The value of **1/lifespan** represents the loss of a year class recruited into a population at carrying capacity. Lifespan is a species-level trait established from the literature.

Species sediment sensitivity is a species-level trait. The focus is on spawning susceptibility to sedimentation events. The examples here are illustrative and based on spawning mode, but additional factors are used to assign species sediment sensitivity, including published assessments of sediment sensitivity and length of spawning period (longer spawning periods are assumed to be less sensitive).

- High sensitivity = 1.0.
- Medium sensitivity = 0.66.
- Low sensitivity = 0.33.

These are essentially reduction factors for species of medium and low sensitivity, such that year class mortality is 66% (medium sensitivity) and 33% (low sensitivity), in the event of a sedimentation event.

No local reproduction. If a species does not use the stream channel for reproduction, this has a value of zero, which means that the sediment effect score is zero. This applies to species that use terrestrial habitats, migratory species that pass through the area but do not spawn there, and species that use specialized habitats not present locally (e.g., an obligate spring-spawning species, for a project in a stream or river).

OTHER CONSTRUCTION EFFECT SCORE

This score captures other construction-phase risks that require AMMs. These include measures for handling hydraulic fluid, restrictions on use of herbicides, and other construction-phase activities that are not related to erosion and sedimentation control. At this time, all such AMMs are proposed to be mandatory, although some include flexibility in how they can be met. There is little literature available for estimating this value. It is currently set at 0.5 * physical contact score. Although the mechanisms of effect differ, this approach scales the effect to project size and allows for higher impacts on species that have reduced mobility to avoid spills.

OCES = 0.5 * physical contact score

POST-CONSTRUCTION EFFECT SCORE

The post-construction effects of a project are those associated with runoff from the roadway, and are calculated for an assumed 50-year lifespan of structures. Based on the scientific literature (REFS), the research team assumed that the main mechanism by which runoff from impervious surfaces affects aquatic biota is via toxicity from contaminants such as heavy metals and hydrocarbons that derive mainly from automobile traffic. Impervious surfaces that are directly connected to streams via the stormwater drainage network have a much stronger effect on aquatic biota than disconnected impervious surfaces (REFS). Consequently, directly connected impervious area (DCIA) has been shown to be a better predictor of sensitive aquatic organism occurrence than simple impervious area. The research team used the net increase in DCIA as the basis for the post-construction effect score, allowing DCIA to be reduced by stormwater BMPs that lead to disconnection (i.e., that reduce runoff volume via infiltration), as well as those that improve water quality without reducing volumes. If a project involves retrofitting of BMPs to treat existing runoff, it may produce a negative post-construction effect score, indicating a positive net impact to aquatic biota.

PCES = Change in habitat suitability * (1/lifespan) * Effect area * 50 years

Change in habitat suitability = Net discounted DCIA * (1/ 1747 acres) * Pollutant sensitivity

Net discounted DCIA = Net project DCIA – DCIA reduction by BMPs

Post-construction effect score is based on the annual change in organismal abundance in the effect area. It assumes that species abundance in a reach is proportional to habitat suitability and the area of the reach. Thus, a change in habitat suitability (occurrence probability) can be converted into an index of decline in the number of organisms. The research team assumed that this decline affects every generation of organisms for the next 50 years; e.g., a species with a mean lifespan of 5 years would lose a portion of 10 generations by the decline in habitat suitability.

The value of 1/**lifespan** is used to estimate the number of generations over the 50-year period. Lifespan is a species-level trait established from the literature.

Effect area is a flat value of 30,000 m^2 or 7.41 acres, which is based on a 1.5km area of influence for impervious cover (see below) and a 20m wide mid-sized stream. The research team does not adjust for stream size because the amount of contaminant loading is fixed, not a function of stream size.

Change in habitat suitability is a function of the net change in DCIA expressed as a percentage of the area of a 1.5km radius circle: $7.07*10^6$ m^2, or 1747 acres. This is based on the pollutant sensitivity relationships of Wenger et al. (2008) for species in the Etowah River basin, which found that DCIA in a 1.5km radius best explained species occurrences. This percentage in all cases will be very small.

Pollutant sensitivity is a species-level trait based on the scientific literature and, where available, empirical relationships between species occurrence and directly connected impervious area. The values below indicated the decline in % occurrence with a 1% increase in DCIA. The "extremely intolerant" values are based on *Cyprinella trichroistia,* the most sensitive species analyzed by Wenger et al. (2008), which approaches an occurrence probability of zero at 3% DCIA. The "tolerant" value reflects a weak relationship with DCIA, and the intermediate categories span the range of sensitivities between these.

- Extremely intolerant = 40
- Very intolerant = 25
- Somewhat intolerant = 15
- Moderate/Somewhat tolerant = 7.5
- Tolerant = 2.5

These values are linear slopes. Wenger et al. (2008) reported logistic regression results, which are nonlinear relationships between occurrence probability and DCIA, but the relationships are close to linear except when occurrence approaches zero or one.

Net discounted DCIA is the effective change in DCIA associated with the project (**net project DCIA**), after accounting for the effect of stormwater BMPs treating new or existing impervious cover. The calculations steps are:

1. Determine the existing and proposed IA for the project.
2. Determine the number and type of existing and/or new BMPs used, and calculate the amount of IA draining to (managed by) each BMP.
3. Estimate a Toxicity Reduction Value (TR%) for stormwater BMPs as a function of design storm depth using an Excel spreadsheet.
4. Calculate the new and existing discounted DCIA. The equations below use TR% to calculate a Net Discounted DCIA for each project to quantify a stormwater effect based on level to which new IA is treated with BMPs. The total effect can be further reduced by electing to retrofit existing, unmanaged IA on the project site with stormwater BMPs.

   New Discounted $DCIA_1$ = [New IA_1*(1-TR_1% for BMP_1)*(1-TR_2 for BMP_2)*…(1-TR_n for BMP_n)]

   If existing IA is treated with new BMPs, the following credit is calculated:

   Existing Discounted $DCIA_1$= [Existing IA_1*((TR_1% for BMP_1)+(1-TR_1% for BMP_1)(TR_2% for BMP_2))*…((TR_n% for BMP_n)+(1-TR_n%)*(TR_{n+1}))]

Treatment of existing IA is optional and can include retrofitting BMPs where controls do not currently exist, or installing additional BMPs to augment current management of existing IA.

5. Calculate the Net Discounted DCIA for the entire project by subtracting the Existing Discounted DCIA (if any) from the New Discounted DCIA.

Net Discounted $DCIA_{SITE} = \sum^{n}_{I=1}$ New Discounted $DCIA_i$ - $\sum^{n}_{I=1}$ Existing Discounted $DCIA_i$

RECONNECTION EFFECT SCORE

This score is zero unless a project involves culvert removal and replacement that increases connectivity between populations of a species. It is based primarily on amount of reconnected habitat and secondarily on the recolonization potential of the species.

RES = Area of reconnected habitat * recolonization potential

Area of reconnected habitat is a category based on the proportional extension of habitat for the species. This must be determined on a case-specific basis in consultation with USFWS and/or DNR. In some cases, the increased habitat could be calculated for a subspecies or ESU rather than the full species.

- Small: <1% increased habitat = .25
- Medium: 1-5% increased habitat = 0.5
- Large: >5% increased habitat = 1.0

Recolonization potential is a species-level trait that indicates the species propensity to move into new habitat. This is a modifier on the area of reconnected habitat with relatively little variability among categories, since organisms could be translocated if necessary.

- High = 1.0
- Medium = 0.9
- Low = 0.8

MAXIMUM EFFECT SCORE

The maximum effect score is the effect score limit: the effect score must be less than this value. It is based on the component effect scores re-calculated with the following settings: (1) no recruitment effect (i.e., as if timing restrictions were in place), (2) Habitat destruction that is unavaoidable, (3) SES calculated with the "advanced" level of BMPs (level 2/5), (4) OCES, (5) PCES calculated based on actual net DCIA addition multiplied by .65, reflecting a typical level of stormwater control under current conditions.

$MaxES = (\text{phys contact score} + SES_{adv} + OCES + PCES_{std}) * \text{Overall imperilment} * \text{Critical population}$

The **phys contact score** and the **OCES** are calculated as for the TES. These impacts are considered to be unavoidable, as they have already been minimized by required AMMs.

The **SES_{adv}** effectively sets a baseline expectation of sedimentation control that is slightly better than standard, reflecting the AMMs traditionally required when federally protected aquatic species are present.

The **$PCES_{std}$** similarly sets a baseline expectation of stormwater management that corresponds to the research team's best estimate of the toxicity reduction associated with a typical level of stormwater control typically used when imperiled aquatic species are present. It assumes a 35% reduction in toxicity by stormwater BMPs, which is equivalent to typical BMPs built for a 0.5" design storm.

Overall imperilment is a species-level trait. For species that are critically endangered across their range this is set at 0.8.

Critical population adjusts the MaxES up or down if the USFWS judges the local population to be more or less important for the recovery of the species than is typical. The adjustment can be:

- High- .8
- Above average- .9
- Below average- 1.1

CHAPTER 3. SEDIMENT EFFECT SCORE DETAILS

Development of the sediment effect score involved reviewing current GDOT standard practices related to construction phase AMMs, special provisions previously implemented on GDOT construction projects, AMMs from additional states' PBAs and DOT manuals, and relevant literature regarding construction phase AMMs. The research team then made recommendations to previous special provisions applied on projects across the state resulting in a suite of recommended construction phase AMMs. A method was developed for calculating species level risk to sedimentation based on local project characteristics and implemented AMMs. The following sections of this chapter outline that procedure in greater detail.

GDOT STANDARD PRACTICE AND SPECIAL PROVISIONS

Example special provisions applied to past GDOT projects with imperiled aquatic organisms were evaluated based on the level of protection they provide relative standard practice, additional states' AMMs, and potential improvements identified in the literature review. There were 34 example special provision documents, 27 of which were provided by the USFWS. The other 7 were obtained from an Internet search. These documents collectively covered 62 aquatic species, 29 Georgia counties, and 51 individual avoidance and minimization measures. The research team assumed this sample of special provisions is representative of measures used to reduce construction-related impacts on imperiled aquatic organisms in Georgia. The research team only considers freshwater aquatic organisms in the analysis.

Special provisions intended to reduce impacts from sediment pollution were the most commonly applied class of provisions in the documents (Table 2). However, the most commonly applied individual special provision was the restriction of pesticide and herbicide use within 200’ of streams (Table 3). The next most frequent individual special provision was the prohibition of staging and equipment maintenance areas within 200’ of streams. The large number of sediment special provisions compared to the other types highlights the variation among the sediment-related special provisions and the need to account for various species sensitivities and site conditions.

Each special provision was categorized by the impact type it was intended to reduce and the type of reduction measure that it represents (e.g., activity restriction versus intercepting practice). The research team quantified the number of instances each special provision was used in the 34 documents. Where available, information on project type was included in the analysis since some special provisions pertained to unique situations. However, project information was not available for all the examples, and professional judgment was used to determine applicability in these scenarios.

A comparison of individual special provisions with other states’ AMMs, standard practice, and the results of the literature review are provided in table format in Appendix A.

Table 2. Frequency of application of special provisions classified by impact type.

Special Provision Impact Type	Number of Individual Provisions	Total Number of Applications
Sediment	33	312
Contaminants	4	68
Physical Contact	4	28
Altered Hydrology/Connectivity	7	23
Noise	3	10

Table 3. Primary terminology of the 5 most commonly applied individual special provisions among the 34 example GDOT special provision documents.

Special Provision	Count	Impact Type	Reduction Type
The Contractor shall not use pesticides or herbicides, within 200 feet	30	Contaminants	Activity Restriction
Equipment staging areas and equipment maintenance areas (particularly for oil changes) shall be located at least 200 feet from stream banks	29	Contaminants, Sediment	Source Prevention
Stockpiled materials shall be placed at least 200 feet away from the banks	25	Sediment	Source Prevention
Instream Timing Restriction	23	Sediment	Activity Restriction
The Contractor shall notify the Project Engineer immediately in the event of an erosion control failure that allows discharge of sediment into Stream	19	Sediment	Monitoring

LITERATURE REVIEW OF CONSTRUCTION PHASE AMMS

Additional States' AMMs

The research team reviewed Programmatic Biological Agreements (PBAs) from multiple state departments of transportation and other agencies around the country to identify relevant AMMs that could be compared to practices used by GDOT. The PBAs came from nine states and four agencies: Federal Highways Administration (FHWA), Florida (FLDOT), Maine (MDOT), Minnesota (MinnDOT), North Carolina (NCDOT), North Dakota (NDDOT), Nebraska (NDOR), National Marine Fisheries Service (NMFS), Oregon (ODOT), Pennsylvania (PennDOT), Tennessee Valley Authority (TVA), U.S. Army Corps of Engineers (USACE), and Washington (WSDOT). The purpose of the review was to identify potential improvements and recommendations for GDOT construction related AMMs. The research team also sought to identify limitations in previous PBAs, such as limited scope or project applicability.

The research team identified and assessed 494 unique AMMs in the various PBAs. Each AMM was assigned a potential impact it would mitigate and the routine GDOT project types to which it would apply (Appendix B). This allowed for direct comparisons with GDOT standard practices and special provisions. After extensive evaluation, the research team concluded that GDOT's mitigation practices were up to par compared to the rest of the country, but that there were relevant AMMs recommended or mandated by other states and agencies that improved on GDOT standard practices and special provisions. These findings are summarized in the sections below with the relevant states and agencies cited.

Cofferdams: Multiple AMMs identified improved ways to mitigate the impact of cofferdams on biota. MDOT, FLDOT, and NMFS had AMMs focused on reducing the amount of fine sediment that was released when a cofferdam was removed. The practices included de-watering techniques and restrictions on using silt/turbidity curtains with cofferdams.

Site disturbance: Most of the PBAs highlighted the importance of leaving the vegetated rootstock intact on-site to prevent excess erosion. They also placed restrictions on grading or clearing streambanks and riparian buffers. Site planning and phasing practices, like those from FLDOT, ensured that disturbed areas would be kept to a minimum throughout a project.

Stabilization: Maine, North Carolina, and Washington had AMMs that called for stabilization of any disturbed soil, even temporary storage piles on-site. Their practices included using tarps, mulch, and coir fiber mats to decrease the amount of sediment that was eroded during storm events.

Maintenance and monitoring: The majority of the PBAs reviewed called for increased maintenance and monitoring of mitigation practices. This was an important theme that highlighted how the efficiency of erosion control measures and other BMPs was dependent on proper upkeep. These measures sought to prevent catastrophic failures that would lead to large sedimentation events.

Other AMMs: Most of the states and agencies included practices that mitigated the impacts of contaminants, altered hydrology/connectivity, physical contact, and noise. Heavy machinery and equipment were restricted from use in water bodies (MDOT) to prevent biota from being crushed or in direct contact with harmful heavy metals and hydrocarbons. Secondary containment measures (MinnDOT) were recommended to prevent construction debris, spills, and leaks from reaching streams. Restrictions were placed on the use of liquid concrete, pesticides, herbicides, and treated materials around riparian zones and water bodies by most of the PBAs. For noise, most of the PBAs had AMMs focused on the use of vibratory hammers, pile cushions, and containment structures during hydrodemolition/in-water blasting.

A comparison of other states' AMMs with individual special provisions applied on past GDOT projects with imperiled aquatic organisms is shown in Appendix C.

LITERATURE REVIEW

The research team reviewed a broad literature on aquatic impacts and mitigation measures for construction-related activities. This included peer-reviewed literature as well as reports and publications that were highly relevant and of sufficient quality, including reports from other state DOTs and the Transportation Research Board. In general, there was a wealth of information on sediment impact and mitigation measures ranging from field evaluations, full-scale experimental tests, and fine-scale laboratory studies. However, there was generally much less information on the effectiveness of mitigation measures for other impact types. For instance, the research team found almost no studies documenting the efficacy of cofferdams to mitigate sediment impacts from construction-related activities. Where there were few studies available to evaluate the effectiveness of certain special provisions, the research team relied primarily on other states'

AMMs (prior section) for guidance. For organizational purposes, a full outline of the literature review is included in Appendix A. Key findings include:

Monitoring and maintenance: Improper installation and lack of maintenance are common occurrences that greatly reduce the efficacy of erosion and sediment control BMPs. Consequently, enhanced levels of monitoring and maintenance can improve BMP efficacy by identifying deficiencies and correcting them prior to erosion generating rain events.

Sediment barrier BMPs: Silt fence is one of the most commonly applied sediment barriers, but efficacy is sensitive to soil type. The presence of smaller grained soil types generally reduces BMP efficacy. In other words, silt fence performs better when retaining sand compared to silt or clay. Techniques have been developed to mitigate this issue, improve the integrity of the installation and reduce failure likelihood. Silt fence backed with baled straw is not recommended because it is prone to failure from overtopping, undercutting, and structural instability since it reduces the flow-through rate and causes excess ponding upslope.

Site disturbance: Modifying construction phasing and sequencing to reduce the site footprint can result in cost savings and decrease the magnitude and likelihood of erosion on a site.

Stabilization: The use of polyacrylamide (PAM) in conjunction with stabilization measures, such as mulching or vegetation, can reduce the amount of soil erosion. Erosion control mats, particularly those made of jute or coconut fiber are more effective at reducing erosion than mulch. However, when applied at the appropriate coverage rate, mulch is effective at reducing erosion compared to bare soil.

Noise: The use of scare charges as a mitigation measure for blasting operations can cause more harm than good, as it increases noise impacts on aquatic organisms.

RECOMMENDATIONS

A full list of special provision recommendations is provided in Appendix A. Some key recommendations include:

- Projects that result in disturbed soil area and are exempt from the requirements of the General NPDES Permit No. GAR100002 according to Part I.C.1.c-d of the Permit should, at a minimum, follow the guidelines in Part IV. D.
- Increasing the frequency of BMP inspection and maintenance schedule will increase BMP efficacy, reduce deficiencies, and reduce the risk associated with sediment impacts.
- The use of enhanced containment devices for heavy equipment and staging areas along with the development of a specification for staging areas will reduce the risk of sediment and contaminant impacts while allowing for a potential reduction in the 200' stream buffer listed in the existing special provision.
- Modifications and additions to sediment barrier practices are recommended to increase their effectiveness based on soil type (i.e. sand, silt, clay).

- The research team recommends developing a checklist or other mechanism that serves as a reminder to contractors of critical erosion control standard specifications immediately prior to construction.

In Appendix A (located in Volume II) the research team listed the primary terminology of each special provision, the number of documents in which it was identified (out of 34 documents reviewed), the impact it reduces (e.g. sediment), the mitigation type (e.g. source control, interception, etc.), and the project types where it was applied. Information on project type was not available for all special provisions. The special provisions are organized by the impact type they are intended to mitigate. The impact types are sediment, contaminants, physical contact, altered hydrology/connectivity, and noise. Based on the findings of the evaluation, the research team developed a suite of 48 AMMs recommended for inclusion as special provisions in the PBA. The list of recommended AMMs is included in Appendix A, located in Volume II. An outline and justification of special provisions that are likely candidates for incorporation into standard practice is also included in Appendix A.

SEDIMENT EFFECT SCORE

Calculation

Most special provisions applied to GDOT construction projects are sediment-control best management practices, reflecting the fact that sedimentation has negative effects on the recruitment and survival of many imperiled freshwater organisms. The research team address this with a probabilistic approach, in which the research team first estimates the chance of a major failure of erosion control, and then multiply this by the expected loss of recruitment in the current or succeeding spawning season. For this purpose, a rainfall-runoff event causing enough soil erosion to result in significant instream sedimentation defines a major failure. A minimum level of erosion and sedimentation best management practices (E&S BMPs) is required for all projects. In many cases it's possible to add redundancy or employ more protective versions of E&S BMPs, which can reduce the probability and severity (extent) of events. Further, streams with different geomorphic characteristics (e.g. drainage area, slope, etc.) will have varying capacities to flush deposited sediment. Consequently, the research team defines multiple E&S BMP levels and flushing potentials to modify the sedimentation severity. The sediment effect score (SES) is calculated as:

$$SES = Sedimentation\ probability \times Loss\ per\ sedimentation\ event$$

Sedimentation Probability

Sedimentation probability is the expected number of events with a specified probability representing the likelihood of a major failure of erosion control that results in a large slug of sediment to the stream or river that buries the natural bed material in at least 2 cm of fine sediment (clay, silt, sand). The research team uses a 2-year, 24-hour storm as a design event since it is typically capable of producing enough runoff to generate large quantities of sediment without enhanced E&S control on construction sites. The annual exceedance probability of this event (50%) is multiplied by the project duration to determine the expected number of events. For example, for a 1-year project, this is 0.5, and for a 2.5-year project this is 1.25 (i.e. 2.5*0.5). The research team could, with similar results, assume a rarer but larger event that would result in greater sedimentation extent.

$$Sedimentation\, probability = \; p_{event} \; \times duration$$

P_{event} is the annual exceedance probability of the design event. This is set at 0.5. *Duration* is the project duration in years. It is never less than 1 but may be fractional if greater than one.

Sedimentation Probability Rationale

Failure of common sediment barriers has been observed in an experimental test simulating a 2-year, 24-hour storm event (Whitman et al., 2018; 2019), and field investigations have identified BMP failures at events less than the 2-year, 24-hour storm (Barret et al. 1995). Further, evidence from the literature coupled with substantial precedent in stormwater practice supports the use of a 2-year event as a reasonable design value.

Loss Per Sedimentation Event

The loss per sedimentation event is calculated as:

$$\begin{aligned} &Loss\ per\ sedimentation\ event \\ &\qquad = Sedimentation\ extent\ \times Species\ sediment\ sensitivity\ \times (1/lifespan) \end{aligned}$$

Loss per sedimentation event assumes the loss of a year class due to sediment smothering eggs, filling interstitial spawning cavities, and otherwise degrading spawning habitat. Its calculation is analogous to that of recruitment effects for the DES. *Sediment extent* is multiplied by *1/lifespan* and then adjusted based on species sensitivity. The value of 1/lifespan represents the loss of a year class recruited into a population at carrying capacity. *Lifespan* is a species-level trait established from the literature.

Species sediment sensitivity is a species-level trait. The focus is on spawning susceptibility to sedimentation events. The examples in Table 4 are illustrative and based on spawning mode, but additional factors are used to assign species sediment sensitivity, including published assessments of sediment sensitivity and length of spawning period (longer spawning periods are assumed to be less sensitive).

Table 4. Species level sediment sensitivities implemented in the SES calculation.

Sensitivity	SES Factor	Example
High	1.0	species that rely on coarse gravel with minimal fine sediment for spawning
Medium	0.66	cavity spawners that may find habitat reduced but not eliminated by a sedimentation event
Low	0.33	species that attach eggs to sides of boulders

These are essentially reduction factors for species of medium and low sensitivity, such that year class mortality is 66% (medium sensitivity) and 33% (low sensitivity), in the event of a sedimentation event.

Sedimentation extent is based on an estimate of soil loss in a major failure, spread over the surface of the stream to a biologically relevant threshold depth (2 cm). The extent is modified by the level of BMPs and the flushing potential of the stream (e.g. slope and drainage area).

$$\begin{aligned} &Sedimentation\ extent \\ &\qquad = (soil\ loss/area) \times disturbed\ area \times (1/theshold\ depth) \\ &\qquad \times Flushing\ potential\ \times E\&S\ BMP\ level \end{aligned}$$

Soil loss/area is an order of magnitude estimate of event soil loss based on the Revised Universal Soil Loss Equation (RUSLE). Soil loss is calculated on a per area basis, and it is proportional to terrain characteristics and the predominant soil type. Calculated soil loss (mass/area) is multiplied by the project scale and a conversion factor to obtain sediment volume. The RUSLE is implemented in a spreadsheet tool. Required inputs can be selected from a drop-down menu, and include soil type, a characteristic slope of the terrain, and a length describing the longest path parallel to the slope before the gradient decreases.

Project Scale in this case means approximate extent of disturbed area, which will be directly input as the estimated area of the limits of disturbance.

The *E&S BMP level* modifies the sedimentation extent based on the level of protection provided by E&S BMPs at a project (Table 5). There are five categories of protection: standard (minimum required), advanced, advanced – high, high, and very high which reduce the sediment extent by 40-80%.

Table 5. E&S BMP levels implemented in the SES calculation.

BMP Level	Design Storm Efficiency	SES Factor
Standard	40%	0.6
Advanced	50%	0.5
Advanced - High	60%	0.4
High	70%	0.3
Very High	80%	0.2

The *E&S BMP level* is based on a framework that determines if applied BMPs are sufficient to achieve categories above the standard. The framework is intended to provide a high degree of flexibility so that BMPs can be appropriately matched to site conditions, and is described in detail in the BMP Level Determination section of the report.

Flushing potential is based on an index of specific stream power (i.e. stream power per unit bed width) within a reach, or the power available to do geomorphic work within a reach. Within the SES calculation, the extent of a sedimentation event is modified by flushing potential to account for reaches that might store large amounts of sediment for longer periods of time compared to reaches that might mobilize the sediment downstream relatively quickly. The research team uses a stream power index that scales drainage area by slope (Flores et al. 2006) to delineate energy classifications and define flushing potential (Table 6).

Table 6. Flushing potential implemented in the SES calculation.

Flushing Potential Energy Classification	Longitudinal Stream Slope (ft/ft)	Stream Power Index ($SA^{0.4}$)	SES Factor
Very high	> 0.025	> 0.302	0.5
Medium – high	> 0.025	≤ 0.302	0.75
Low – medium	< 0.025	> 0.081	0.8
Low	< 0.025	≤ 0.081	1
Note: S is longitudinal stream slope (ft/ft), and A is upstream drainage area (mi^2)			

E&S BMP level Rationale

The research team assumes that an event equaling or exceeding the design storm event results in partial failure of standard E&S control practices. The BMP efficiency during a 2-year, 24-hour design event will be highly variable due to many site-specific factors. On average, this efficiency will be reduced compared to more common rain events, but it will not be zero (Shueler et al 2014; U.S. EPA 2009). This general assumption is supported by empirical evidence from the literature documenting a wide range of BMP efficiencies that generally decreases with increasing storm event magnitude (Barrett et al. 1995; Chapman et al. 2014; Edwards et al. 2016, Kalainesan et al. 2009; Line 2007; Line and White 2001). The research team evaluated case studies that document BMP efficiency, type, and storm magnitude to quantify a distribution of BMP efficiencies and inform the BMP level factors in Table 5.

Restricting particular sediment generating activities, enhancing the level of source prevention (i.e. preventing erosion at the source), increasing requirements for intercepting practices (e.g. silt fence and sediment basins), and enhancing monitoring and maintenance can increase the efficiency of E&S control. The method for classifying E&S BMP level accounts for various levels of activity restriction, source prevention, interception, and monitoring and maintenance to account for the various efficiencies achieved by these four types of practices. Activity restriction has the highest efficiency followed by source prevention, interception, and monitoring and maintenance. The classification evaluates various combinations of these types, and the BMP level determination is based on a combination of empirical evidence and professional judgment. The classification framework asks a series of questions in a decision tree that links construction activities with applicable BMPs. The decision tree avoids an arbitrary scoring mechanism and provides transparency in the determination of a specific E&S BMP level. The full details of the classification framework are available in Appendix A, Volume II.

Flushing Potential Rationale

Scaling drainage area by slope ($SA^{0.4}$) has been derived as an index of specific stream power that was successful at delineating streams with different energy classifications (Flores et al. 2006). Thus, sediment extent is modified by this classification, which the research team terms “flushing potential”, to account for variation among reaches to store or transport deposited sediment from and E&S control failure.

Specific stream power has been useful in a number of applications to estimate sediment transport and predict zones of erosion and deposition (Lammer and Bledsoe 2018; Sholtes et al. 2018). The classification system applied by Flores et al. (2006) generally delineates reaches as transport-limited, transitioning between transport-limited and supply-limited, and supply-limited, reflecting differences in energy dissipation and relative transport capacity (Montgomery and Buffington 1998; Montgomery and MacDonal, 2002). While the research team does not intend to draw these distinctions for the rivers in Georgia, this system provides a useful means for discriminating among reaches of varying specific stream power. Consequently, it can be expected that lower energy reaches will store sediment in pools and along channel margins even after high flow events, such as annual peak floods (Rathburn and Wohl 2002; Wohl and Cendrelli 2000), and have a greater propensity to retain larger amounts of sediment in a reach for periods of time biologically relevant for aquatic organisms. Evidence of this is supported by a modeling experiment that simulated sediment transport in pool-riffle reach with a high-resolution

2-D hydrodynamic model showing fine sediment from pulsed and chronic events remained within the reach over year after the event (Maturana et al. 2013). In contrast, higher energy reaches have a higher capacity to mobilize fine sediment across a wider range of flow events compared to low energy reaches.

The research team recognizes that sediment dynamics will be highly variable in space and time among locations and even within a single channel reach. Ultimately, the flushing potential and the duration of a sediment impact within a reach will depend on multiple complex factors such as the ambient sediment supply, sequence of streamflow, relative size distribution of the input sediment disturbance relative to the parent material, and magnitude of the disturbance. Patterns of erosion and deposition are difficult to predict without resource intensive data collection and modeling efforts. However, the calculation the research team proposes classifying a reach by the power available to transport sediment downstream, and the general capacity of its anticipated form to transport or store sediment.

Required Inputs

The sediment effect score requires several inputs. The aim was to keep them as simple as possible and maintain a level of detail sufficient to describe the general site characteristics relevant to construction-related impacts on aquatic biota. The inputs generally account for the wide range of conditions that might be encountered across Georgia. For example, sediment impacts from a bridge replacement project at a small headwater tributary might have vastly different impacts on aquatic biota than a bridge replacement project at a river with a large drainage area and high dilution factor.

The required inputs are implemented in a simple format via spreadsheet tool (Figure 2) allowing users to select values and categories from dropdown menus. Calculations and decision trees implemented in the effect score calculation is automated and occur as a background process. The sediment effect score is calculated on a species by species basis, but multiple streams / rivers or site locations on a project might require the summation of multiple sediment effect score values to obtain the project total.

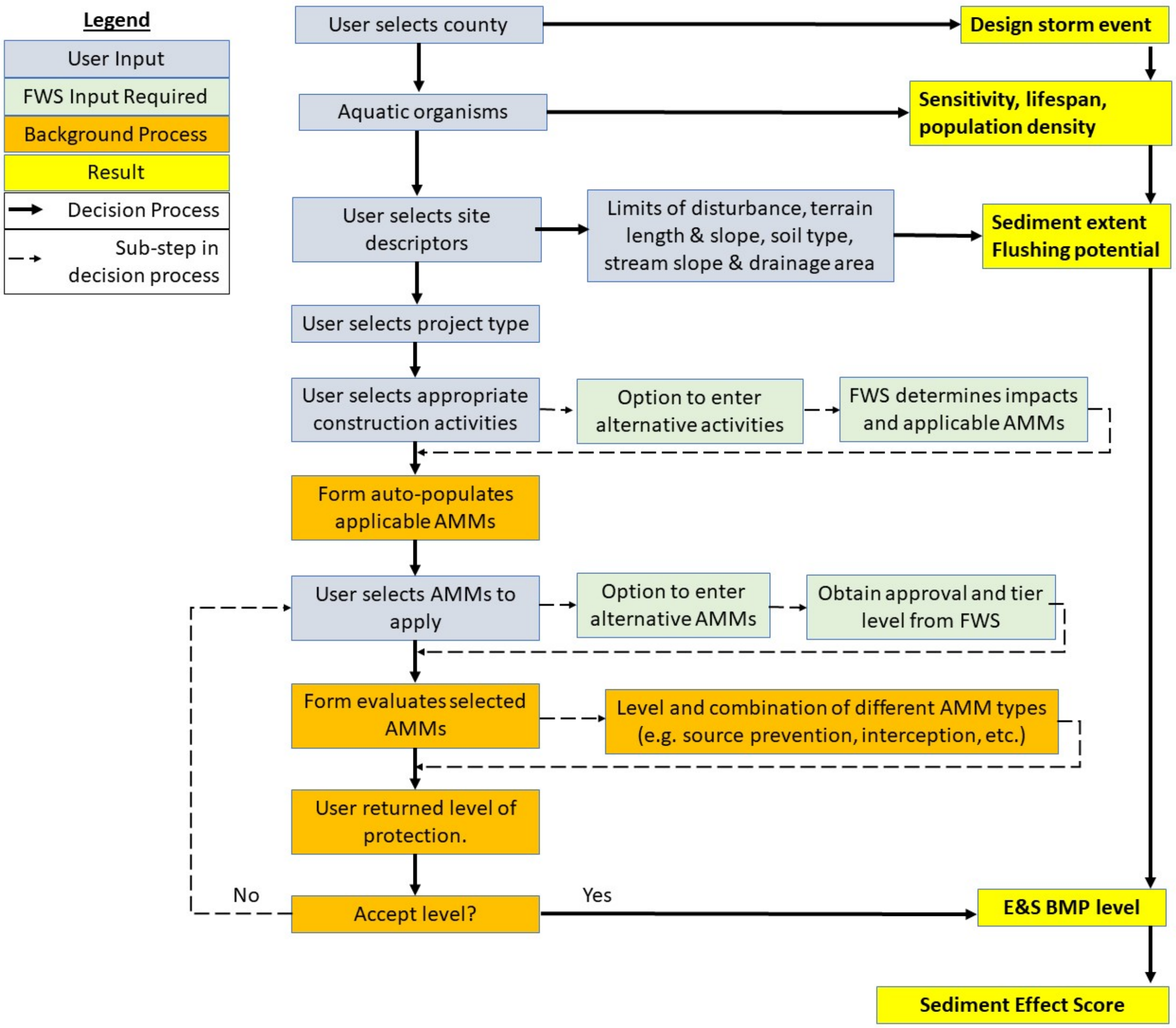

Figure 2. Diagram. Framework for achieving the sediment effect score.

The inputs in Table 7 represent those required to estimate construction phase sediment impacts from a project. These inputs pertain to a single stream / river that a disturbed project area drains to. A project might require multiple sets of inputs (e.g. slope 1, slope 2, length 1, length 2, etc.) depending on the extents and orientation of the limits of disturbance (LOD) relative to adjacent streams and rivers. For instance, two different sets of inputs would be required for a project containing two streams where different portions of the LOD drained to each stream. In addition to the inputs in Table 7, the anticipated construction activities and AMMs must be identified.

Table 7. Site level construction phase input data required to calculate the sediment effect score.

Required Input	Description
County	The county that the project occurs in. If the set of inputs pertains to an area that spans more than one county, select the county that contains the largest portion of area for that set of inputs.
Estimated project duration (years)	The estimated duration of initial disturbance to final stabilization (e.g. permanent grassing, matting, and mulching complete) for a project area. Estimates may be in fractions of years, but 1 is the minimum (e.g. 1.5-years).
Limits of disturbance (acres)	For each stream and project area within its drainage area, estimate the limits of disturbance (LOD).
Site slope (%)	Estimate the representative slope steepness of the predominate drainage path within the limits of disturbance for a project area. The slope steepness is the change in elevation per horizontal distance along the slope (i.e. vertical change/horizontal change). This value is multiplied by 100 to obtain a %. For example a 4:1 (H:V) cut slope has a slope steepness of 25%. Slope may vary along the drainage path, but estimate the most representative value for the LOD. A dropdown menu will be provided to select a slope steepness that best defines the site conditions.
Site slope length (ft)	Estimate the maximum length of the slope before surface runoff becomes concentrated in a defined channel or the slope steepness decreases enough that sediment deposition begins (e.g. a flat area at

	the bottom of a steep slope). This value is estimated for the LOD of a project area. A dropdown menu will be provided to select a slope length that best defines the site conditions.
Soils	A dropdown menu will be provided to select the predominant soil texture within the LOD. Soil textures include (fine sand, very fine sand, loamy sand, loamy very fine sand, sandy loam, very fine sandy loam, silt loam, clay loam, silty clay loam, silt clay).
Drainage area of upstream watershed (square miles)	Input the watershed drainage area for each stream / river that the LOD drain(s) to. The drainage area for the stream / river should be determined at the downstream end of the LOD. If not already available, estimates of upstream drainage area can be obtained with following link: https://streamstats.usgs.gov/ss/
Approximate longitudinal slope of the stream / river (%)	Estimate the approximate longitudinal slope of the stream / river that the LOD drain(s) to. The stream slope can be approximated by estimating the elevation change along the length of the stream (i.e. vertical change in elevation /horizontal change in elevation). This value is then multiplied by 100 to obtain a percent. Slope should be estimated over a distance no less than 10 channel widths upstream and downstream. Length is measured as the total length along the stream / river's flow path and is not necessarily a straight-line distance. If there is curvature along the stream's flow path, this must be included in the length measurement. In the absence of more accurate measurements, alternative methods can be used to estimate stream /river slope, such as obtaining stream length and elevation change from the streams and contours on a 1:24,000 scale topographic map or the following link: https://streamstats.usgs.gov/ss/.

CONSTRUCTION BMP LEVEL DETERMINATION OVERVIEW

The E&S BMP level determination is intended to provide flexibility while objectively evaluating the level of protection of BMPs to reduce sedimentation risk. This determination is based on a framework that incorporates project specific activities and the AMMs that apply to those activities. Identification of applicable AMMs is an automated process in the spreadsheet tool that calculates the SES, and it requires selecting appropriate project activities from a list. Once a list of applicable AMMs is provided based on selected construction activities, the user can select

AMMs they intend to apply on a project. A decision tree is then evaluated to determine the BMP level provided by the selected AMMs.

LINKING CONSTRUCTION ACTIVITIES WITH APPLICABLE AMMS

Activities are linked to applicable AMMs by potential impact type (i.e. sediment, contaminants, physical contact, altered hydrology/connectivity, and noise). In collaboration with the USFWS, the research team identified potential impacts for 77 construction activities provided by GDOT. The research team assumes that the list of 77 activities represents the range of construction activities anticipated under the PBA. However, it is understood that unforeseen circumstances may arise, and a step was included in the BMP level determination accounting for additional activities to avoid limiting the scope of the PBA (Figure 3).

Similar to the construction activities, the research team identified each impact type that the AMM was intended to reduce. In this manner, activities can be linked with AMMs based on impact type. The research team created a database that includes all AMMs, activities, and their association. Each association within the database (> 1000), was manually reviewed to remove any associations that did not apply (i.e. AMMs that were not applicable to the specific activity). Using this database, a user can select appropriate construction activities for their project, and the database will automatically populate a list of applicable AMMS. All construction activities and their impact types are listed in Appendix B (located in Appendix II).

E&S BMP Level

AMMs are generally classified by mitigation type, which is the method by which they reduce impacts (activity restriction, source prevention, interception, and monitoring and maintenance). It can be expected that the greatest reduction of a potential impact comes from restricting an activity, followed by source prevention, interception, and monitoring and maintenance, respectively. For instance, prohibiting grading in a particular portion of the project reduces the risk of impact in that location more than installing an interception practice (silt fence). Further, if interception or source prevention practices are not in place, there are fewer practices to monitor and maintain.

A decision tree was developed for the individual methods for reducing impacts. Within each tree, required AMMs were identified based on the anticipated impact severity of the practices, the level of protection provided by the AMM, standard practice, existing special provision examples, and relative cost. The decision tree follows a series of logical questions to determine the level of protection provided. While the aim is to provide multiple alternatives or pathways to achieve a desired level of protection, some activities are limited by the number of applicable practices and achievable reduction in anticipated impacts. Consequently, there are some scenarios with few alternatives or BMP level outcomes.

A decision tree was developed for non-sediment AMMs; however, these are not incorporated into the sediment effect score. Rather, non-sediment AMM decision trees (e.g. those related to contaminants) are utilized to determine required across varying levels of species sensitivities. For instance, more protective AMMs to reduce the risk of contaminant exposure might be required for species with a greater sensitivity to chemical pollutants. Decision trees for non-sediment impacts include contaminants, physical impacts, noise, and altered hydrology/connectivity.

Once a BMP level is determined for each mitigation type based on the individual decision trees, the combinations of BMP levels for each mitigation type is evaluated to assign a comprehensive BMP level for sediment related construction impacts (Figure 3). With very high, high, and standard categories existing for each mitigation type, there are 81 possible combinations (e.g. very high, standard, high, high). Each possible combination is assigned a comprehensive level based on logical evaluation and professional judgment. A table of the 81 combinations is included in Appendix A (located in Volume II).

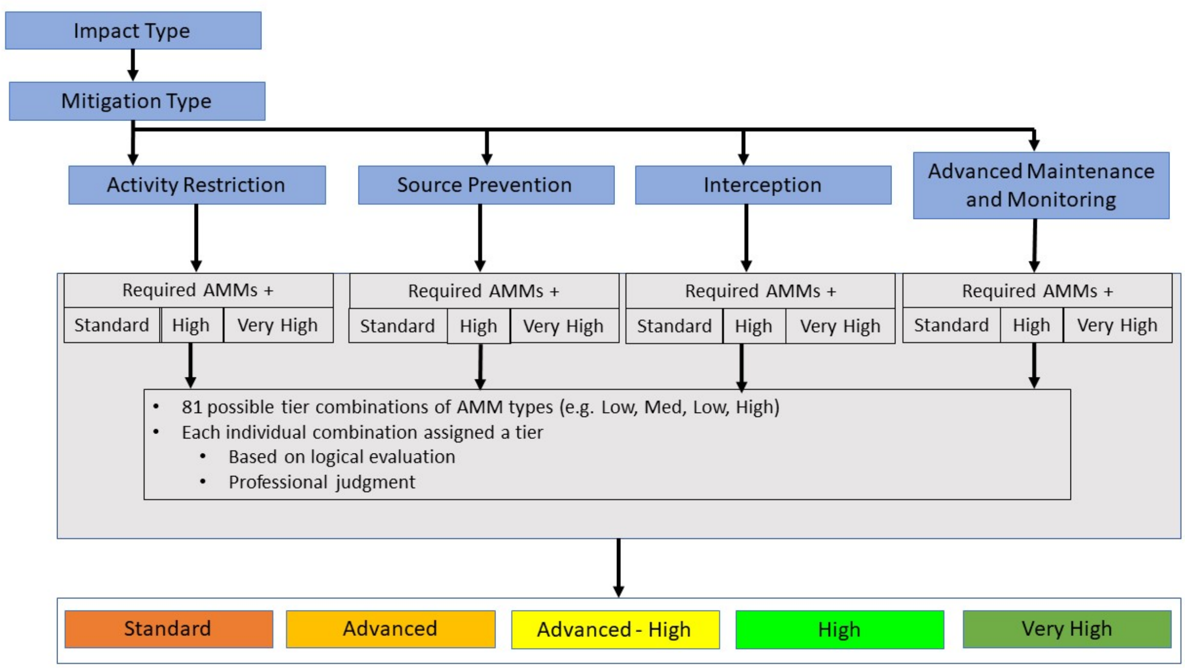

Figure 3. Diagram. Framework for achieving the final BMP level score.

CHAPTER 4. POST-CONSTRUCTION EFFECT SCORE

STORMWATER RUNOFF EFFECTS ON AQUATIC ECOSYSTEMS

Stormwater runoff from impervious surfaces is considered the most important cause of the "urban stream syndrome," the characteristic pattern of urban stream degradation in the developed world (Walsh et al. 2005). The symptoms associated with the syndrome are a flashier hydrograph, increased sedimentation and erosion, excess nutrients, increased toxicants, reduced biotic richness, and altered ecosystem processes (Paul and Meyer 2001, Meyer et al. 2005, Walsh et al. 2005, Wenger et al. 2009). Many of these symptoms occur at low levels of impervious cover, and even modest increases in imperviousness, such as that associated with an individual road project, can have a measurable effect on stream ecosystems.

In this section the research team reviews the effects of stormwater runoff on stream and river ecosystems. The research team starts with a discussion of alternative ways of measuring impervious cover. The research team then summarizes the scientific literature on the effects of runoff on different aspects of flowing water ecosystems.

DEFINITIONS AND ESTIMATION METHODS OF EFFECTIVE IMPERVIOUS AREA (EIA) AND DIRECTLY CONNECTED IMPERVIOUS AREA (DCIA)

As an important indicator of urbanization, imperviousness is widely used to estimate urban stormwater runoff in order to quantify the impacts of urbanization on hydrology. Percent total impervious area (TIA) is the most straightforward metric of imperviousness, and is defined as “the fraction of the watershed covered by constructed, non-infiltrating surfaces such as concrete, asphalt, and buildings (Booth and Jackson 1997).” However, using TIA to estimate runoff to streams may not be accurate because not all stormwater runoff from TIA drains into stormwater systems (Sohn et al. 2017). Potential problems of using TIA include: overestimated runoff volumes, peak flows, and diminished infiltration rates, and underestimated simulated runoff changes (Alley and Veenhuis 1983). Effective impervious area (EIA) can be defined as TIA that is hydraulically connected to the channel drainage system, such as streets with curbs and gutter, and paved parking lots that drain onto streets, and non-effective impervious area that infiltrates, such as a roof that drains onto a lawn (Alley and Veenhuis 1983). Previous research has shown that EIA has a much greater effect on stream hydrology than TIA (Brabec et al. 2002, Shuster et al. 2005).

Previous studies have demonstrated that EIA is a better predictor of urban runoff and stream biological and chemical response than TIA (Alley and Veenhuis 1983, Booth and Jackson 1997, Wang, Lyons et al. 2001, Hatt et al. 2004, Walsh, Leonard et al. 2004). Analyses based on TIA run the risk of overestimating peak runoff, runoff volume, and time lag between rainfall and peak runoff (Sahoo and Sreeja 2014). A closely related metric is directly connected impervious area (DCIA) (Lee and Heaney 2003, Roy and Shuster 2009, Hwang, Rhee et al. 2017, Sohn, Kim et al. 2017), which is likewise a better predictor of stream ecosystem health than TIA (Roy and Shuster 2009). Researchers have found that using DCIA improved the accuracy of runoff predictions (Seo et al. 2013), and also yielded more estimates of water quality effects (Lee and Heaney 2003). However, researchers have presented multiple definitions of EIA and DICA. Some studies considered EIA and DCIA to be equivalent (Roy and Shuster 2009, Sohn et al. 2017), while others drew a distinction (Ebrahimian et al. 2016, Ebrahimian et al. 2016, Ebrahimian et al. 2018). Definitions of EIA and DCIA from different studies are shown in Table 8.

Table 8. EIA and DCIA definitions and sources.

Effective impervious areas are those that "hydraulically connected to the channe ldrainage system."	(Alley and Veenhuis 1983)
EIA is defined as "the impervious surfaces with direct hydraulic connection to the downstream drainage (or stream) system."	(Booth and Jackson 1997)
"Effective impervious areas are defined as those impervious areas directly connected to streams or receiving waters by pipes or lined channels."	(Hatt, Fletcher et al. 2004)
EIAs are those that "drain to other impervious areas or directly to the storm-drainage system."	(Prych and Ebbert 1986)
DCIA, also referred to as "effective impervious area", is defined as "a fraction of the impervious area that is hydraulically connected to downstream drainage by a buried piped route."	(Sohn, Kim et al. 2017)

“The concept of directly connected impervious areas (DCIA) refers to a subset of impervious cover that is directly connected to a drainage system or a water body via continuous impervious surfaces.”	(Hwang, Rhee et al. 2017)
“The subset of impervious surfaces that route stormwater runoff directly to streams via stormwater pipes is called directly connected impervious area (DCIA) or effective impervious area.”	(Roy and Shuster 2009)
“The portion of TIA that is directly connected to the downstream drainage network is called EIA.”	(Sahoo and Sreeja 2016)
“Directly connected impervious area (DCIA), on the contrary, describes the part of TIA that is hydraulically connected to a drainage system.”	(Yao, Wei et al. 2016)
“Effective impervious area (EIA) is the portion of total impervious area (TIA) that is hydraulically connected to the storm sewer system.” “Directly connected impervious area (DCIA) is the portion of TIA that is directly connected to the drainage system.”	(Ebrahimian, Gulliver et al. 2016, Ebrahimian, Wilson et al. 2016, Ebrahimian, Gulliver et al. 2018)

Numerous methods have been developed to estimate EIA and DCIA (Alley and Veenhuis 1983, Laenen 1983, Prych and Ebbert 1986, Dinicola 1990, Boyd et al. 1993, Sutherland 1995, Lee and Heaney 2003, Hatt et al. 2004, Wenger et al. 2008, Han and Burian 2009, Ebrahimian et al. 2016, Sahoo and Sreeja 2016, Aulenbach et al. 2017, Hwang et al. 2017, Ebrahimian et al. 2018). These methods can be classified into two types: direct methods and indirect methods. One type of indirect methods is an empirical equation (Alley and Veenhuis 1983, Laenen 1983, Dinicola 1990, Booth and Jackson 1997, Wenger et al. 2008, USEPA 2010, Wenger et al. 2010, USEPA 2014, Sohn et al. 2017). For example, Alley and Veenhuis (1983) quantified the relationship between EIA and TIA for 19 urban basins in Metropolitan Denver by using $EIA = 0.15\ TIA^{1.41}$, an equation that was developed for 14 Denver basins in an unpublished report. However, the relationship only represented a small sample of basins in a single metropolitan area. Laenen (1983) defined the relationship between mapped and effective impervious area in Salem and parts of the Portland metropolitan area as $EIA = 3.6 + 0.43\ MIA$ (mapped impervious area). Dinicola (1990) used empirical values (40 percent, 50 percent, 66 percent, 80 percent, and 95 percent of TIA) to estimate EIAs for five land-use types. Sohn et al. (2017) estimated parcel-level DCIAs by using Sutherland's equations (Sutherland 1995) (Table 2), which are empirical formulas developed to assess DCIAs at a basin or sub-basin scale. The estimation should be based on an analysis of the hydrologic connectivity status of the impervious surfaces. USEPA (2010, 2014) also adopted Sutherland's equations to estimate DCIA based on land use types. However, DCIA may not be accurately predicted from empirical relationships with TIA because of a wide variation in connectivity across parcels. Models lack consistent performance across datasets (Roy and Shuster 2009).

Table 9. Sutherland's equations to determine DCIA (%).

Connectivity status	Selection criteria	Equation (where TIA(%) $\geq$ 1)[a]
Totally disconnected	100% drained to landscape	$DCIA = 0$
Mostly disconnected	Small percentage of urban area is storm sewered, or 70% or more infiltrate/disconnected	$DCIA = 0.01(TIA)^2$
Somewhat connected	50% not storm sewered, but open section roads, grassy swales, residential rooftops not connected, some infiltration	$DCIA = 0.04(TIA)^{1.7}$[b]
Average	Mostly storm sewered with curb and gutter, no dry wells or infiltration, residential rooftops not connected to the storm sewer or piped directly to the street curb	$DCIA = 0.1(TIA)^{1.5}$
Highly connected	Same as above, but residential rooftops are connected to streets or storm sewer system	$DCIA = 0.4(TIA)^{1.2}$[c]
Totally connected	100% storm sewered with all TIAs connected	$DCIA = TIA$

[a] The values of the DCIA and the TIA in the equations are in percentages (%). USEPA assumed the DCIA % to be zero where the TIA % is less than one (Sutherland 1995).
[b] The DCIA % to be 100% where the TIA % is higher than 99.4%.
[c] The DCIA % to be 100% where the TIA % is higher than 97.7%.

Another type of indirect method is the use of a statistical model to estimate EIA (Boyd et al. 1993, Boyd et al. 1994, Ebrahimian et al. 2016, Ebrahimian et al. 2016, Ebrahimian et al. 2018, Epps and Hathaway 2019). For example, Boyd et al. (1993, 1994) determined EIA by plotting rainfall-runoff data. Ebrahimian (2016) improved the rainfall-runoff data analysis method for estimating EIA fraction in urban catchments by eliminating the subjective part of the existing method and by reducing the uncertainty of EIA estimates. In another study, they developed a new method to estimate EIA fraction for ungauged urban watershed using only readily available spatial data by linking EIA with curve number (Ebrahimian et al. 2018). However, Ebrahimian (2016) defined EIA as a parameter that cannot be "observed" like DCIA, which renders validation impossible.

Direct methods generally involve GIS application and field investigation (Prych and Ebbert 1986, Lee and Heaney 2003, Hatt et al. 2004, Han and Burian 2009, Sahoo and Sreeja 2016, Yao et al. 2016, Aulenbach et al. 2017, Hwang et al. 2017). For example, Hwang et al. (2017) estimated DCIA based on a series of criteria using high-resolution land cover data in GIS. Lee and Heaney (2003) applied five levels of approaches that included GIS application and field investigation to estimate the DCIA of a three-block study area. Their approaches were also employed by Yao et al. (2016) to estimate DCIA. Sahoo and Sreeja (2016) determined DCIA by a semi-automated method that integrated remote sensing data, drainage networks, and a DEM of the study area. They found that compared to these directly estimated values, DCIA and EIA determined by various empirical equations were overestimated. Han and Burian (2009) estimated DCIA using a two-step process. In the first step, they classified fine-scale multispectral imagery into urban land cover to define TIA. In the second step, they used GIS to trace water flow paths from impervious areas to classify TIA as directly connected or disconnected. Direct methods tend to be more accurate than empirical equations but are much more time intensive, and so may be most suitable for estimating EIA of small areas.

HYDROLOGY, GEOMORPHOLOGY AND TEMPERATURE

The most fundamental physical effect of increased DCIA/EIA is to increase stream flashiness. With less land area available for soil infiltration, runoff increases and groundwater recharge declines (Walsh et al. 2004). Urban streams may experience peak discharges that are 2 to 5 times greater than that of predeveloped conditions (Booth and Jackson 1997, Walsh et al. 2004, 2005). The number of smaller peaks also increases, as even small rainfall events can produce sufficient runoff to cause a rapid increase in stream flow. Higher flows can cause physical washout of fauna downstream, scouring of streambeds and algal assemblages, and decreased retention of organic matter and nutrients (Wenger et al. 2009).

Increased flashiness from stormwater runoff is a major cause of altered channel geomorphology in urban streams (Vietz et al. 2016). Bed and bank erosion in urban stream channels can increase both stream width and depth compared to nonurban streams. The cycle of sedimentation and erosion of stream channels connected with urbanization was first described by Wolman (1967): first, construction-phase land use change produces inputs of sediment to streams; this is typically followed by an aggradation period when this channel filling counters the erosion triggered by runoff; after construction wanes, high flows remove sediment deposits, leading to channel incision, yet rarely to stabilization (Wolman 1967, Trimble 1997, Paul and Meyer 2001, Wenger et al. 2009). Smaller, more frequent high-flow events could have greater influence on channel geomorphology and ecology than larger, infrequent events (Walsh et al. 2004, 2005). Channel erosion in urbanizing streams can be an ongoing source of sedimentation to downstream reaches.

Stormwater runoff from heated impervious surfaces can create pulses in stream temperature (Nelson and Palmer 2007, Wenger et al. 2009). A loss of vegetative shading along the stream corridor can exacerbate temperature increases (Krause et al. 2004, Wenger et al. 2009). Reduced base flows can further increase stream temperatures (Nelson and Palmer 2007). Altered thermal regimes have the potential to exceed tolerances of cold-water species, and cause stress, mortality, and shift species assemblages toward those metabolically adapted to higher temperatures (Krause et al. 2004, Wenger et al. 2009). Trout, other salmonids, and stoneflies are especially susceptible to minor temperature changes due to a narrow window of adaptability (Jones and Hunt 2009, GSWMM 2016).

WATER QUALITY

The introduction of chemical pollutants in stream ecosystems has been shown to vary widely across geographical, geological, climatic, land use, and historical gradients, but is a universal factor in urban streams, occurring at even low levels of urbanization (Walsh et al. 2005). As Hatt el al. (2004) note, “Runoff from urban areas is one of the leading causes of water quality degradation in surface waters.” The interrelated effects of various contaminants on altered water quality in stream ecosystems is complex, and the synergistic or antagonistic effects of these compounds make it difficult to predict their consequences (Walsh et al. 2004, Wenger et al. 2009).

Metals are the most prevalent “toxicants” (chemical contaminants common in urban streams that cause lethal and sublethal effects to aquatic organisms) found in North American urban runoff, and are introduced via roadways, vehicles, atmospheric deposition, and urban surfaces (Wenger et al. 2009, GSWMM 2016). Vehicular transportation and atmospheric deposition on roadways are primary sources of pollution in urban areas, which has been studied extensively, however pollutant concentrations in roadway runoff vary depending on traffic intensity (ADT), maintenance practices, climatic variability, and proximity to anthropogenic pollution sources (Mullen et al. 2020). Typical heavy metal concentrations are up to 100 times greater in urban stormwater runoff compared to non-urban runoff (Walsh et al. 2004). Common heavy metals include cadmium, chromium, copper, iron, lead, magnesium, nickel and zinc, which are often bound to sediment and organic matter (Paul and Meyer 2001, Wenger et al. 2009, GSWMM 2016). Finer sediment particle sizes with higher POM content have greater affinity and binding capacity for metals (Paul and Meyer 2001). Common sources of heavy metals from roadways include the wear of bearings and brake linings, tire abrasion, moving engine parts, lubricants, oil and grease (Burns 2012; Burns 2015). In a study that quantified levels of contamination leaving Georgia DOT Right-of-Ways and entering nearby stormwater BMPs, Burns (2012) recorded storm events where heavy metal concentrations in the runoff from the ROWs exceeded the Georgia EPD guidelines for both acute and chronic impacts.

Wenger et al. (2009) found that sensitivity of aquatic biota to toxicants varies among taxa and can shift with sediment suspension during high storm flows. The overall impacts of heavy metals on stream ecosystems can be seen in harmful bioaccumulation along the food chain, degradation of drinking water supply, increasing cost of treatment, and ecological impacts as will be explored in the final section (Vick et al. 2012).

Sediment particles, released into urban streams from soil and stream channel erosion or construction activities, transport other pollutants bound to their surfaces, such as heavy metals, nutrients and hydrocarbons (GSWMM 2016). Suspended solids, or undissolved particulate matter that remains in the water column, can increase turbidity, which reduces light for plant growth, and can result in smothering of habitat in low-flow areas and scouring of habitat in high-flow areas (Walsh et al. 2004). As noted above, sediment plays a central role in the synergism of water quality constituents, and ultimately has major impacts on recreational and aesthetic value of surface waters, ecosystem processes, aquatic habitat, and flooding by filling drainage infrastructure (Paul and Meyer 2001, Walsh et al. 2004, GSWMM 2016).

Nutrients such as nitrogen and phosphorous are transported to aquatic ecosystems through stormwater drainage systems, agriculture, pet waste, construction activities, atmospheric deposition, automobile combustion, food, and wastewater (Bernhardt et al. 2008, Wenger et al. 2009, Carey et al. 2013). Even though N and P are essential to the ecological health of aquatic ecosystems, their excessive loading from impervious surfaces can lead to degradation of receiving waters (Carey et al. 2013). Characteristic problems associated with eutrophic aquatic systems are toxic and non-toxic algal blooms which can deplete oxygen levels causing “dead zones” and fish kills, block light to submerged aquatic vegetation, and contaminate drinking water supplies (Carey et al. 2013, GSWMM 2016). Elevated nutrient levels, and shifts in the proportions of different nutrients, can also greatly impact ecosystem processes, including nutrient uptake and retention, leaf decomposition rates, and primary production (Wenger et al. 2009). Paul and Meyer (2001) found that the primary source of phosphorous stored in soils is residential and industrial fertilization, which becomes mobilized via soil erosion through urbanization. Internal combustion engines incidentally produce nitrous oxides, providing localized sources of atmospheric nitrogen along transportation networks; this nitrogen can then be delivered to streams via precipitation and runoff.

Research has shown that the main vector of transport for pesticides (including herbicides, fungicides, and insecticides) discharged into urban streams is nonpoint source stormwater runoff (Paul and Meyer 2001). Interestingly, concentrations of pesticides in urban areas frequently exceed those found in intensive agricultural areas in the US, and can come from residential, industrial, golf course, lawn management, and aerial applications (USGS 1999, Paul and Meyer 2001). In water quality studies, pesticide detection levels frequently exceed aquatic life benchmarks set to protect aquatic biota (Wenger et al. 2009).

The remainder of the organic toxicants are part of three major groups: PAHs (polycyclic aromatic hydrocarbons) which includes organic solvents, PCBs (polychlorinated biphenyls) whose manufacturing was outlawed due to carcinogenic effects, and petroleum-based hydrocarbons, including oils, gasoline, and grease from motorized vehicles (Paul and Meyer 2001). Urban rivers in Georgia have been found to contain fish with concentrations exceeding consumption-level guidelines for these categories of toxicants (Frick et al. 1998). As for petroleum-based hydrocarbons, evidence suggests they are frequently found at concentrations determined to be stressful to sensitive stream biota (Paul and Meyer 2001).

Organisms vary greatly in their sensitivity to contaminants associated with runoff from impervious surfaces. The research team reviews the effects of contaminants on aquatic biota in Chapter 6, Volume I. The research team assesses sensitivity of imperiled species to contaminants in Appendix C, Volume II.

STORMWATER MANAGEMENT ON GDOT PROJECTS

The 2017 GDOT MS4 permit requires that new development and redevelopment projects within MS4 areas must implement structural and/or non-structural BMPs in compliance with applicable parts of the Georgia Stormwater Management Manual and Coastal Stormwater Supplement, or an equivalent stormwater management manual. Post-construction stormwater management measures appropriate for GDOT projects are described in the GDOT Drainage Manual, along with guidance on BMP design. The GDOT Drainage Manual is a comprehensive and up-to-date publication and serves as a good reference for selecting BMPs suitable to manage stormwater discharges from GDOT projects that potentially impact imperiled aquatic species.

Within MS4 areas, GDOT projects must manage for stormwater runoff quality/reduction by implementing BMPs that will retain the first inch of rainfall on the site, to the maximum extent practicable. If this is achieved, then additional water quality treatment is not required. If the one-inch rainfall cannot be retained onsite, then the runoff from the 1.2 in rainfall must be treated to remove at least 80% of the total suspended solids (TSS) load. While these standards are consistent with most current MS4 standards around the country, the 80% TSS target may not be adequate for ensuring the protection of imperiled aquatic species. Based on the scientific literature, the main mechanism by which runoff from impervious surfaces affects aquatic biota is via toxicity from contaminants such as heavy metals and hydrocarbons. Therefore, the research team has developed a method of quantifying the Post-Construction Effect Score that accounts for the impact of impervious surfaces on imperiled aquatic species, as well as the efficacy of BMPs in reducing that impact.

STORMWATER REDUCTION FACTOR

Estimated changes in effective / equivalent imperviousness based on removal of toxics are translated into biological effect scores using species contaminant tolerances reported in the scientific literature and empirical relationships between effective imperviousness and aquatic species occurrence. Toxicity reduction factors are based upon a weight-of-evidence approach that combines estimates of runoff reduction, TSS, and total metals reduction and from the Georgia Stormwater Management Manual, GDOT Drainage Manual and scientific literature.

The research team's approach is adapted from USEPA methods for estimating DCIA (*https://www3.epa.gov/region1/npdes/stormwater/ma/MADCIA.pdf*). In this approach, DCIA is defined as the portion of Impervious Area (IA) with a direct hydraulic connection to a waterbody via continuous paved surfaces, gutters, drainpipes, or other conventional conveyance and detention structures that do not reduce runoff volume.
DCIA does not include:

- IA draining to stormwater practices designed to fully meet recharge and other volume reduction criteria.
- Isolated IA with an indirect hydraulic connection to a waterbody, or that otherwise drain to a pervious area.

- Man-made impoundments, unless drained to a waterbody.
- The surface area of natural waterbodies (e.g., wetlands, ponds, lakes, streams, rivers).

Steps in the approach are as follows:

6. Determine the existing and proposed IA for the project (All GDOT projects have a measurable IA associated with the project).
7. Determine the number and type of existing and/or new BMPs used, and calculate the amount of IA draining to (managed by) each BMP.
8. Estimate a Toxicity Reduction Value (TR%) for stormwater BMPs as a function of design storm depth using the latest available version of the Total Effect Score spreadsheet. The Total Effect Score spreadsheet allows TR% to be estimated as a continuous function of any design depth from 0.5-1.8".

   The TR% for each BMP and BMP combination is based on the Total Metal Reduction Value, TSS Reduction Value (adsorbed fraction), and Runoff Reduction value (%RR; dissolved fraction) from the GSWMM and the GDOT Drainage Manual, as well as relevant scientific literature, using a weight of evidence approach. If a range of representative reduction values was not available from the weight of evidence approach, then two alternative approaches were employed to develop the regressions for estimating %TR as a function of design depth. First, for sites where it was available, the Total Metal Reduction value from the GSWMM was multiplied by credit removal values of 0.4 and 1 to create a range for different design depths. Alternatively, for sites where the Total Metal Reduction was not available, an average of TSS removal and RR% was multiplied by the same credit removal values (0.4 and 1) to create the range used in the regressions. This hybrid value reflects removal of both toxins adsorbed to solids and those dissolved in solution. The %TR values should be reviewed at least every five years to incorporate new information and evidence on BMP performance.

9. Calculate the new and existing discounted DCIA. The equations below use TR% to calculate a Net Discounted DCIA for each project to quantify a stormwater effect based on level to which new IA is treated with BMPs. The total effect can be further reduced by electing to retrofit existing, unmanaged IA on the project site with stormwater BMPs. This provides additional flexibiity by incentivizing the optional treatment of existing, unmanaged IA.

- New Discounted $DCIA_1$ = [New IA_1*(1-TR_1% for BMP_1)*(1-TR_2 for BMP_2)*…(1-TR_n for BMP_n)]

If GDOT unmanaged, existing IA is treated, the following credit is calculated:

- Existing Discounted $DCIA_1$= [Unmanaged, Existing IA_1*((TR_1% for BMP_1)+(1-TR_1% for BMP_1)(TR_2% for BMP_2))*…((TR_n% for BMP_n)+(1-TR_n%)*(TR_{n+1}))]

Note that treatment of existing IA is optional and can include retrofitting BMPs where controls do not currently exist, or installing additional BMPs to augment current management of existing IA.

10. Calculate the Net Discounted DCIA for the entire project by subtracting the Existing Discounted DCIA (if any) from the New Discounted DCIA.

- Net Discounted $DCIA_{SITE} = \sum^{n}_{I=1}$ New Discounted $DCIA_i$ - $\sum^{n}_{I=1}$ Existing Discounted $DCIA_i$

11. Determine Project's Post-Construction Effect Score.

EXAMPLE OF METHOD

A new GDOT project proposes to expand a portion of GA-197 along the Soquee River from two-lane road into four-lane highway due to expected growth and development in the area. This project falls within the newly proposed PA. The proposed expansion calls for 9.8 acres of new IA to be added to an existing 25.2 acres of IA. GDOT decides to use this project as an opportunity to retrofit existing, unmanaged IA with stormwater BMPs. They determine that 0.7 acres of the existing IA is unmanaged.

Due to the rural setting and limited ROW, GDOT decides to only use two different treatment trains designed for 1” storms for the IA:

- Treatment Train 1: OGFC (TR%=38%) and Vegetated Filter Strips (TR%=69%)
- Treatment Train 2: Vegetated Filter Strips (TR%=69%) and Grassed Channels (TR%=50%)

The new IA is in three different sized sections of 2.0, 5.6, and 2.2 acres.

To determine the Net Discounted DCIA, the equations outlined above are used:

- *New Discounted DCIA*= [2.0 acres*(1-.38)*(1-.69)] + [5.6 acres*(1-.69)*(1-.5)] + [2.2acres*(1-.69)*(1-.5)] = 1.59 acres
- *Existing Discounted DCIA*= [0.7 acres*((.38)+(1-.38)*(.69))] = .565 acres
- *Net Discounted DCIA*= 1.59 acres-0.565 acres = 1.025 acres

The project’s Stormwater Effect Score will be based upon 1.025 acres of DCIA. If GDOT had opted not to treat the existing, unmanaged IA, then the amount of DCIA would be 1.59 acres for the Stormwater Effect Score.

GREEN INFRASTRUCTURE STRATEGIES FOR GDOT PRACTICES

GDOT projects fall into three general categories of activities: major roadway projects, site development projects (rest areas, etc.), and maintenance and safety projects. Site development projects are generally less constrained by issues that are specific to linear roadway projects. Some challenges that must be considered when selecting stormwater management strategies for roadway projects are:

- Roadways are distributed across the state in a variety of site conditions
- The widespread and distributed nature of roadway assets make maintenance challenging
- GDOT right-of-way often limits the amount of space available for BMP installation
- Utility conflicts in GDOT right-of-way frequently limit space available for BMP installation
- Safety requirements must be addressed

Prior to identifying structural and non-structural BMPs for stormwater management, GDOT projects should utilize Better Site Design practices and techniques to minimize the amount of impervious area that is connected to the drainage network. This in turn can help to minimize project cost. The Georgia Stormwater Management Manual and the GDOT Drainage Manual identify the following strategies:

- Reduction of Roadway Footprint

- Porous Pavements such as Open Graded Friction Course (OGFC) and Porous European Mix (PEM) on Interstate and State Route Resurfacing and New Construction
- Using Rural Shoulder In lieu of Urban Curb and Gutter
- Conservation of Natural Feataures and Resources
  - Preserve Undisturbed Natural Areas
  - Preserve Riparian Buffer
  - Avoid Developing in Floodplains
  - Avoid Developing on Steep Slopes
  - Minimize Siting on Porous or Erodible Soils
- Lower Impact Site Design Techniques
  - Fit Design to the Terrain
  - Locate Development in Less Sensitive Areas
  - Reduce Limits of Clearing and Grading
- Utilization of Natural Features for Stormwater Management
  - Us Buffers and Undisturbed Areas
  - Use Natural Drainageways Instead of Storm Sewers
  - Use Soil Restoration Practices to Improve Native Soils (see BMP list below)
  - Restore Tree Canopy Outside of Clear-zone Limits (see BMP list below)

Despite the inherent challenges, many BMPs are applicable and effective for managing runoff from linear roadway projects. BMPs that are feasible for linear roadway projects include:

- Filter Strips*
- Grassed Channels*
- Bioslopes*
- Detention Ponds*
- Enhanced Swales (really bioretention basins with chambers)*
- Bioretention Basins*
- Stormwater Ponds & Wetlands*
- Enhanced Wet Swales^
- Infiltration Trenches^
- Sand Filters^
- Open Graded Friction Course^
- Regenerative Stormwater Conveyance[+]
- Site reforestation/Revegetation[+]
- Soil Restoration[+]
- Vegetated Riparian Buffer[#] (Appendix B: Riparian Forest Buffer BMP)

* BMPs identified by GDOT as preferred strategies for roadway projects

^ Additional BMPs listed in the GDOT Drainage Manual

\+ Additional BMPs detailed in the Georgia Stormwater Management Manual that are also feasible for linear roadway projects

\# Additional BMP proposed as a part of this programmatic agreement

COMBINING GREEN INFRASTRUCTURE PRACTICES

Stormwater runoff should be cooled, slowed and treated by a variety of practices prior to discharging into surface waters. The GDOT Drainage Manual identifies BMP treatment train combinations for GDOT projects. These are utilized primarily to achieve the 80% TSS removal required by the MS4 permit, but they can be effective in toxic constituent removal as well if they are properly sited.

Suggested BMP order for treatment trains includes: Filter Strip & Grass Channel; Grass Channel & Filter Strip; Open Graded Friction Course (OGFC) & Filter Strip or Dry Detention Basin & Grass Channel (Drainage Design for Highways, p 10-22). Other opportunities include expanding the treatment train to include additional practices such as adding bioswales or bioretention to Filter Strip & Grass Channel combinations. For example, a flow splitter could divert design storm water quality volumes into practices in sequence. Water quality volume runoff could be diverted via a grass channel into stormwater wetlands, then into level spreaders prior to entering revegetated floodplains or receiving waters, which could be enhanced or restored. Such an example would read as “Filter Strip & Grass Channel & Bioretention & Stormwater wetland & Level Spreader.” Runoff in excess of the design storm diverted by the flow splitter (not diverted for water quality treatment) could be diverted into detention ponds via grassed channels for volume control. Examples include Grass Channels or Bioretention or Filter Strips draining into Detention Ponds.

MAINTENANCE CONSIDERATIONS FOR GREEN INFRASTRUCTURE BMPS

No BMP is maintenance free if it is to maintain its design function over time. However, maintenance considerations for BMPs range from more intensive to less intensive.

The GDOT Stormwater System Inspection and Maintenance Manual specifies I&M practices for most of the BMPs listed above. Recommended practice is that each BMP is checked for compliance annually, with more frequent routine inspections after storms, or as needed based on past maintenance issues or other site-specific reasons. All GDOT stormwater BMPs are to be inspected and maintained according to this manual.

The maintenance and inspection guidance suggest several categories of effort, including vegetative management, interim maintenance, and long-term maintenance. Vegetative management includes mowing, trash removal, and mulching up to several times a year. The interim and long-term maintenance calls for sedimentation removal and structural fixes, including media replacement. If inspection and maintenance is performed, most BMPs will perform at high levels BMPs for many decades. Sediment management is critical, as sedimentation is the greatest cause of BMP performance degradation over time.

As part of this study, investigators toured several BMP sites with GDOT representatives in 2019 and noticed several maintenance or design issues that contributed to sediment accumulation. Several in field design modifications were proposed on the tour to enhance temporary ponding where runoff was bypassing the BMP and not allowed to pond and infiltrate into the media. This visit served as a beneficial inspection and underscored the need for annual inspection and seasonal vegetative maintenance.

RIPARIAN FOREST BMP

Riparian forest buffers are widely recognized as an essential and effective practice for managing polluted runoff and water quality (Lowrance et al., 1997; Wenger, 1999); however, there has been limited research on the effectiveness of riparian buffers for removing toxicants (e.g. heavy metals) from urban impervious areas. Previous research indicates that sediment deposition (heavy metals adsorb to sediment particles) and uptake by woody vegetation may help riparian areas mitigate heavy metals (Hupp et al. 1993). Two fundamental factors affecting the efficiency of riparian forest buffers are slope and length (Wenger, 1999). Sheet flow must be maintained throughout the buffer as concentrated flow greatly reduces the efficiency of the buffer. Riparian areas with low to moderate slopes can slow the force of stormwaters to reduce the amount of sediment, crop debris, and other particulates reaching streams (Klapproth and Johnson, 2009).

Based on the best available scientific information, the following equation was developed to estimate the toxicity removal of a riparian forest buffer BMP with sheet flow, width of 30-100 ft slope of 2- 8%, and protection in perpetuity as part of a GDOT stormwater management strategy:

TR = 70% - [(100’ – Width of Buffer in Feet)*(2/7)] – [(Slope of Buffer in % - 2%)*(10/3)]

The equation was designed to assign a maximum toxicity reduction value of 70% under ideal conditions (100’ width and no more than 2% slope). As width decreases and slope increases, there is a linear decrease in the TR value to a minimum of 30%. No credit will be given for the BMP if the input values are outside the set ranges for width and slope of 30-100 ft and 2- 8%, respectively. The range of buffer widths is based on a review of long term studies that examined how a buffer of 100’ is sufficient to trap sediments under most circumstances, but an absolute minimum width should be no less than 30’ to remain effective (Wenger, 1999). Another review of 80 scientific articles on vegetated buffers concluded that a 9% slope or less allowed for consistent laminar flow across the buffer zone. Above 9%, sheet flow and sediment trapping efficiencies may be significantly reduced (Liu et al., 2008).

ADDITIONAL RECOMMENDATIONS

As research and monitoring informs the design and maintenance of BMPs, the research team recommends updating the list of preferred strategies included in this document and the Total Effect Score spreadsheet. New BMPs that demonstrate effective toxicity removal performance and are feasible for linear roadway projects should be added. Because many of the BMP efficiency rates were not tested in Georgia or have results with wide variance, the research team recommends that experts assist with researching BMP efficiency. Consider including additional BMPs, such as regenerative stormwater conveyances, to the drainage manual because they are already detailed in the state guidance, but not currently listed in the GDOT manuals. The research team also recommends adding Vegetated Riparian Buffers as a BMP in the GDOT Drainage Manual.

Strive to increase ecological systems thinking across GDOT units and partners to realize co-benefits that will likely enhance landscape performance through maintenance operations that seek to simultaneously enhance aesthetics and improve environmental conditions. An example might include revegetating areas with native forb and grasses to promote pollinators (foraging and nesting).

Continue to utilize the GDOT Construction Review to minimize the need for design/construction changes during construction and to allow adequate coordination with USFWS. By implementing policy that relays any requested in-field construction alterations or maintenance operations to USFWS, or other agency, GDOT can ensure programmatic agreement compliance.

CHAPTER 5. TEMPLATE FOR A PROGRAMMATIC BIOLOGICAL ASSESSMENT FOR GDOT TRANSPORTATION PROJECTS IN THE RANGE OF IMPERILED AQUATIC SPECIES IN GEORGIA

NOTE: THIS DOCUMENT IS AN ANNOTATED DRAFT TEMPLATE AND IS NOT FOR DISSEMINATION. ANNOTATIONS AND PLACEHOLDERS ARE *[ITALICIZED AND IN BRACKETS]***.**

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INTRODUCTION

Background

Section 7 of the Federal Endangered Species Act (ESA) requires federal agencies to ensure that any action they approve, fund, or carry out is not likely to jeopardize the continued existence of any federally endangered or threatened species or result in the destruction or adverse modification of such species' designated Critical Habitat (16 USCA § 1536(a)(2)). Because they usually utilize federal funding or approval and often occur in the range of such species and their designated Critical Habitat, Georgia Department of Transportation (GDOT) transportation projects are frequently subject to Section 7 consultation requirements. Section 7 consultation can be a time-consuming process, and many federal and state agencies across the United States have developed programmatic consultation programs to both streamline consultation and improve species conservation. Programmatic consultation increases the efficiency of ESA Section 7 consultations by addressing multiple actions on a program, regional, or other basis, and may also expedite permitting processes for such actions.

Over the last few years, GDOT has been working on the main component of a programmatic Section 7 consultation for its transportation projects – a Programmatic Biological Assessment (BA). The Programmatic BA describes the potential impacts to state and federally listed and imperiled aquatic species (hereinafter "imperiled aquatic species") from GDOT transportation projects and avoidance and minimization measures to be implemented for these projects to reduce or eliminate impacts to these species. A programmatic Section 7 consultation based on this Programmatic BA will expedite permitting processes for GDOT transportation projects that fall within programmatic caps and adhere to avoidance and minimization measures. As such, the Programmatic BA supports the dual goals of streamlining Section 7 consultation and improving species conservation.

Scope

This Programmatic BA applies to future GDOT transportation projects that:

- Utilize federal funding or require federal approval;
- Are within the geographic range of, and habitats suitable for, imperiled aquatic species;
- Fall within the programmatic sideboards and project-level caps, as described below; and
- Adhere to avoidance and minimization measures and compensation measures, as applicable.

This programmatic consultation addresses federally listed species, those being considered for listing, and state-listed species. ESA Section 7 consultation is only required for federally listed threatened and endangered species and proposed species and their designated or proposed Critical Habitat, but state and federal agencies may choose to include other species in Section 7 consultation procedures. In addition, coordination with state wildlife agencies on projects that alter aquatic resources is required pursuant to the Fish and Wildlife Coordination Act. By including state-listed species in this programmatic consultation, GDOT is supporting aquatic conservation in Georgia and ensuring efficient transportation project program delivery should state-listed species become federally listed.

This programmatic consultation does not address designated or proposed Critical Habitat. GDOT transportation projects that may impact designated or proposed Critical Habitat must proceed via individual Section 7 consultation procedures.

As of *[month, year]*, the aquatic imperiled species covered are listed in Table 10. This list includes aquatic fishes, crayfishes, mussels, amphibians, reptiles, and insects, with the exception of cave-dwelling and diadromous species. Transportation projects that impact cave-dwelling species are also likely to impact listed bat species and microhabitats supporting them and these projects will therefore proceed under individual Section 7 consultations.

This Programmatic BA applies to GDOT transportation projects that result in No Effect (NE), May Affect, but Not Likely to Adversely Affect (NLAA), or May Affect, Likely to Adversely Affect (LAA) determinations. A Total Effect Score (TES), provided in Appendix A, is used to evaluate potential impacts to imperiled aquatic species from individual GDOT transportation projects and categorize them as NE, NLAA, or LAA. The TES considers the effects of both project activities and implementation of selected avoidance and minimization measures (AMMs) to evaluate the effect of the project as a whole.

Table 10. Imperiled Aquatic Species

Group	Genus	Species	Common Name	Georgia Status	Federal Status.
amphibian	*Cryptobranchus*	*alleganiensis alleganiensis*	Eastern hellbender	T	
amphibian	*Lithobates*	*capito*	gopher frog	R	A (Pe)
amphibian	*Amphiuma*	*pholeter*	one-toed amphiuma	R	
amphibian	*Notophthalmus*	*perstriatus*	striped newt	T	
crayfish	*Cambarus*	*speciosus*	beautiful crayfish	E	
crayfish	*Cambarus*	*unestami*	blackbarred crayfish	T	
crayfish	*Distocambarus*	*devexus*	Broad River burrowing crayfish	T	
crayfish	*Cambarus*	*howardi*	Chattahoochee crayfish	T	
crayfish	*Cambarus*	*scotti*	Chattooga River crayfish	T	
crayfish	*Cambarus*	*extraneus*	Chickamauga crayfish	T	
crayfish	*Cambarus*	*cymatilis*	Conasauga blue burrower	E	
crayfish	*Cambarus*	*coosawattae*	Coosawattee crayfish	T	
crayfish	*Cambarus*	*doughertyensis*	Dougherty burrowing crayfish	E	
crayfish	*Cambarus*	*fasciatus*	Etowah crayfish	T	
crayfish	*Procambarus*	*verrucosus*	grainy crayfish	R	
crayfish	*Cambarus*	*parrishi*	Hiwassee headwaters crayfish	E	
crayfish	*Cambarus*	*strigosus*	lean crayfish	T	
crayfish	*Cambarus*	*georgiae*	Little Tennessee crayfish	E	
crayfish	*Procambarus*	*gibbus*	Muckalee crayfish	T	

crayfish	*Cambarus*	*truncatus*	Oconee burrowing crayfish	T	
crayfish	*Cambarus*	*harti*	Piedmont blue burrower	E	
crayfish	*Procambarus*	*versutus*	sly crayfish	R	
crayfish	*Cambarus*	*englishi*	Tallapoosa crayfish	R	
dragonfly	*Gomphus*	*consanguis*	Cherokee clubtail	T	A (Pe)
dragonfly	*Ophiogomphus*	*edmundo*	Edmund's snaketail	E	A (Pe)
dragonfly	*Cordulegaster*	*sayi*	Say's spiketail	T	A (Pe)
fish	*Alosa*	*alabamae*	Alabama shad	T	
fish	*Cyprinella*	*xaenura*	Altamaha shiner	T	
fish	*Percina*	*antesella*	amber darter	E	E
fish	*Enneacanthus*	*chaetodon*	blackbanded sunfish	E	
fish	*Etheostoma*	*duryi*	blackside snubnose / black darter	R	
fish	*Erimystax*	*insignis*	blotched chub	E	
fish	*Cyprinella*	*caerulea*	blue shiner	E	T
fish	*Elassoma*	*okatie*	bluebarred pygmy sunfish	E	
fish	*Lucania*	*goodei*	bluefin killifish	R	
fish	*Pteronotropis*	*welaka*	bluenose shiner	T	
fish	*Cyprinella*	*callitaenia*	bluestripe shiner	R	
fish	*Percina*	*kusha*	bridled darter	E	
fish	*Pteronotropis*	*euryzonus*	broadstripe shiner	R	
fish	*Notropis*	*asperifrons*	burrhead shiner	T	
fish	*Etheostoma*	*scotti*	Cherokee darter	T	T
fish	*Etheostoma*	*ditrema*	coldwater darter	E	
fish	*Percina*	*jenkinsi*	Conasauga logperch	E	E
fish	*Macrhybopsis*	*etnieri*	Coosa chub	E	

fish	*Noturus*	*sp. cf. N. munitus*	Coosa madtom	E	A (Pe)
fish	*Percina*	*sciera*	dusky darter	R	
fish	*Etheostoma*	*etowahae*	Etowah darter	E	E
fish	*Phenacobius*	*crassilabrum*	fatlips minnow	E	
fish	*Hemitremia*	*flammea*	flame chub	E	
fish	*Percina*	*lenticula*	freckled darter	E	
fish	*Percina*	*aurolineata*	goldline darter	E	T
fish	*Etheostoma*	*parvipinne*	goldstripe darter	R	
fish	*Etheostoma*	*chlorobranchium*	greenfin darter	T	
fish	*Percina*	*crypta*	Halloween darter	T	A (Pe)
fish	*Notropis*	*hypsilepis*	highscale shiner	R	
fish	*Etheostoma*	*brevirostrum*	holiday darter	E	
fish	*Acipenser*	*fulvescens*	lake sturgeon		A (Pe)
fish	*Hybopsis*	*lineapunctata*	lined chub	R	
fish	*Etheostoma*	*chuckwachatte*	lipstick darter	E	
fish	*Noturus*	*eleutherus*	mountain madtom	E	
fish	*Percina*	*smithvanizi*	muscadine darter	R	
fish	*Fundulus*	*catenatus*	northern studfish	R	
fish	*Ichthyomyzon*	*bdellium*	Ohio lamprey	R	
fish	*Percina*	*squamata*	olive darter	E	
fish	*Notropis*	*ariommus*	popeye shiner	E	A (Pe)
fish	*Moxostoma*	*robustum*	robust redhorse	E	A (Pe)
fish	*Etheostoma*	*rupestre*	rock darter	R	
fish	*Notropis*	*scepticus*	sandbar shiner	R	
fish	*Moxostoma*	sp. 2	sicklefin redhorse	E	C
fish	*Notropis*	*photogenis*	silver shiner	E	
fish	*Percina*	*tanasi*	snail darter	E	T
fish	*Ameiurus*	*serracanthus*	spotted bullhead	R	
fish	*Phenacobius*	*uranops*	stargazing minnow	T	
fish	*Fundulus*	*bifax*	stippled studfish	E	
fish	*Micropterus*	*notius*	Suwannee bass	R	

fish	*Etheostoma*	*tallapoosae*	Tallapoosa darter	R	
fish	*Percina*	*aurantiaca*	tangerine darter	E	
fish	*Chrosomus*	*tennesseensis*	Tennessee dace	E	
fish	*Etheostoma*	*trisella*	trispot darter	E	T
fish	*Etheostoma*	*vulneratum*	wounded darter	E	
mussel	*Pseudodontoideus*	*connasaugaensis*	Alabama creekmussel	E	
mussel	*Medionidus*	*acutissimus*	Alabama moccasinshell	T	T
mussel	*Elliptio*	*arca*	Alabama spike	E	A (Pe)
mussel	*Alasmidonta*	*arcula*	Altamaha arcmussel	T	
mussel	*Elliptio*	*spinosa*	Altamaha spinymussel	E	E
mussel	*Utterbackiana*	*heardi*	Apalachicola floater	R	
mussel	*Fusconaia*	*masoni*	Atlantic pigtoe	E	A (Pr (T))
mussel	*Medionidus*	*parvulus*	Coosa moccasinshell	E	E
mussel	*Elliptio*	*arctata*	delicate spike	E	A (Pe)
mussel	*Amblema*	*neislerii*	fat threeridge	E	E
mussel	*Hamiota*	*altilis*	finelined pocketbook	T	T
mussel	*Pleurobema*	*hanleyianum*	Georgia pigtoe	E	E
mussel	*Medionidus*	*penicillatus*	Gulf moccasinshell	E	E
mussel	*Elliptio*	*purpurella*	inflated spike	T	
mussel	*Pleurobema*	*pyriforme*	oval pigtoe	E	E
mussel	*Elliptoideus*	*sloatianus*	purple bankclimber	T	T
mussel	*Strophitus*	*radiatus*	rayed Creekshell	T	
mussel	*Ptychobranchus*	*foremanianus*	rayed kidneyshell	E	E
mussel	*Toxolasma*	*pullus*	Savannah lilliput	T	
mussel	*Hamiota*	*subangulata*	shinyrayed pocketbook	E	E
mussel	*Pleurobema*	*decisum*	Southern clubshell	E	E

mussel	*Alasmidonta*	*triangulata*	Southern elktoe	E	A (Pe)
mussel	*Pleurobema*	*georgianum*	Southern pigtoe	E	E
mussel	*Medionidus*	*walkeri*	Suwannee moccasinshell	E	T
reptile	*Graptemys*	*pulchra*	Alabama map turtle	R	A (Pe)
reptile	*Macrochelys*	*temminckii*	alligator snapping turtle	T	A (Pe)
reptile	*Graptemys*	*barbouri*	Barbour's map turtle	T	
reptile	*Glyptemys*	*muhlenbergii*	bog turtle	E	T
reptile	*Graptemys*	*geographica*	Northern map turtle	R	
reptile	*Macrochelys*	*suwanniensis*	Suwanee alligator snapping turtle	T	A (Pe)
snail	*Leptoxis*	*foremani*	interrupted rocksnail	E	E

R= rare; T=threatened; E=endangered; C=candidate; A= at risk (proposed (Pr), petitioned (Pe), or species of concern (SOC))

DESCRIPTION OF PROPOSED ACTION

Introduction

As defined in the ESA Section 7 regulations (50 CFR 402.2), "action" means "all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies in the United States or upon the high seas." The term "action area"[1] is defined as "all areas to be

[1] Further clarification is provided by the national consultation FAQs at https://www.fws.gov/endangered/what-we-do/faq.html#18

affected directly or indirectly by the federal action and not merely the immediate area involved in the action."

The proposed action is to establish a programmatic consultation approach for GDOT transportation projects that utilize federal funding or require federal approval. The programmatic consultation provides a method for evaluating transportation projects – the TES is used to evaluate potential impacts to imperiled aquatic species, resulting in a No Effect (NE), May Affect, but Not Likely to Adversely Affect (NLAA), or May Affect, and Likely to Adversely Affect (LAA) determination. It provides a total of the anticipated extent of take for the first five years of the programmatic consultation; a project-specific review and project-specific incidental take statements for individual projects will be provided by USFWS.

The action area for this programmatic consultation is all lands in Georgia within the range of the imperiled aquatic species affected directly or indirectly by transportation projects. For the sake of this consultation, the action area for individual projects analyzed by the programmatic consultation will be referred to as the "project action area." Project action areas are not limited to the "footprint" of transportation projects nor are they limited by the authority of the lead federal agency. Rather, they are a biological determination of the reach of the project's disturbance of land, water, and air [insert figure showing your project area].

This section provides a general description of the projects that are covered in this programmatic consultation, with an estimation of the extent of annual project activity. It then generally describes the impact avoidance and minimization measures that may apply to projects, as appropriate.

Projects Included in This Proposed Action

Across the range of the imperiled aquatic species in the State of Georgia, there are *[number]* existing GDOT-owned transportation projects. On an annual basis, GDOT engages in section 7 consultation for *[average number]* transportation projects.

The proposed action includes multiple transportation project actions. Only those projects with TES effect scores under a maximum threshold for all potentially impacted species will be able to proceed under this programmatic consultation (see below and Appendix A). Project types included in the proposed action are:

Bridge/culvert maintenance

Bridge repair, retrofit, and maintenance activities are implemented to prolong the use and function of bridges, ensure motorist safety, and protect the environment. Bridge maintenance activities may include washing, painting, debris (or drift) removal from bridge piers, scour repair, guardrail repairs, joint replacement, lighting and signage repairs, pile encasement, and structural rehabilitation. Culverts require maintenance when at least 25 percent of their capacity is restricted by debris, sediment, or vegetation. Maintenance may also occur for damage to the structure, such as spalls. Culvert maintenance activities may include temporary stream diversion, debris removal, epoxy injections, patch repair, repair of the headwall, outfall, or wing walls, shotcrete lining, scour repair and rip-rap installation, washing, sandblasting, and repainting.

Road maintenance

Road maintenance includes pavement preservation, shoulder work, curb cuts, and striping.

Other maintenance

This category includes routine right-of-way maintenance and guard rail maintenance.

Drainage system maintenance

Drainage System Repair and Maintenance activities include all work necessary to maintain roadside ditches and channels, cross culverts and pipes, catch basins and inlets, and detention/retention basins.

Bridge construction/replacement

Bridge construction may be a component of a larger roadway construction project or a stand-alone project. Bridge replacements tend to be long-term projects requiring one or more years to complete. Major bridge replacement construction activities often include:

- Clearing and grading for road widening
- Clearing and grubbing of existing streamside vegetation

- Adding fill around approaches to raise the elevation of the bridge
- Construction of stormwater facilities
- Excavation for new bridge abutments
- Construction of bridge columns/piers/abutments
- Concrete pouring
- Pile installation and removal
- Bridge demolition
- Rip-rap placement (described in **Bridge/Culvert Maintenance**)
- Paving with asphalt or concrete
- Relocation of above or below ground utilities (described in **New Road Construction**)

Culvert construction/modification

Culverts include small concrete and box girders that do not qualify as bridges due to their size. Typically bridges less than 20 ft. wide are referred to as either culverts or structures. Any culvert over 20 ft. wide is referred to as a “bridge culvert.” Culvert replacements and extensions are typically short-duration activities requiring less than one month to complete. Typical culvert replacements and extensions involve removing vegetation at the outlet and inlet area, removing existing pavement and roadbed to extract the existing culvert, placing the new culvert, backfilling and replacing the pavement, installing armoring and headwalls, re-vegetating if necessary, and if flow is present, dewatering the work area and establishing a flow bypass prior to initiating work.

Road construction

Road construction activities are primarily projects to increase traffic flow and capacity, although objectives may include safety improvements. These projects include new road alignment, high-occupancy toll lanes, high-occupancy vehicle (HOV) lanes, intersection improvements, passing lanes, managed lanes, road realignment (including sharp curve treatments), frontage roads, and road widening. Intersection improvement projects include interchange alignments, new interchanges, roundabouts, median crossovers, and turn lanes. Widening or replacing aging bridges could occur for these projects (see *Bridge Construction/Replacement*). Constructing new or extending / replacing culverts could also occur for these projects (see *Culvert Construction / Modification*).

Safety Improvements

Safety and mobility projects may occur within both rural and urban environments. Projects in this category are designed to improve safety, traffic flow, and operations on existing road corridors.

Public Use

Public use activities include multi-use trails, park and ride facilities, parking areas, rest areas, sidewalks, waystations, scales, and welcome center facilities.

The TES is used to determine which individual projects may proceed using this programmatic consultation and which require individual consultation. It assigns different scores, based on level of anticipated impact, to project components. A Maximum Effect Score (MES) has been developed for all listed aquatic species. Projects for which the TES is under the MES for *all* species potentially impacted by the project may proceed under the programmatic consultation; those that exceed the MES for *any* species potentially impacted must engage in individual Section 7 consultation. The TES was designed so only those projects with predictable, minimized impacts qualify for the programmatic agreement. Projects with more significant impacts require individual consultation. More information on how the TES and associated tools work is in Appendix A.

If the lead federal agency or USFWS deems any project outside of the scope of considerations of this biological assessment, they reserve the right to exclude the project from programmatic Section 7 consultation.

[Here, need to estimate the total impact of this programmatic – i.e., the take of species over a period of years agreed upon by the federal agencies (potentially 5 years, based on GDOT's 5-year plan). This can be calculated by determining the incidental take for the highest possible effect score for each project type and multiplying that by the number of that type of project in GDOT's 5-year plan. Should be estimated by watershed and a statement about species recruitment should likely be included. Likely useful to put all of this into a table with a description of the process for estimating take. Should likely note that this is almost certainly an overestimate of incidental take for the 5-year programmatic timeframe because the maximum

effect score for each project was used and many projects will not reach the highest possible effect score.]

Impact Avoidance and Minimization Measures

GDOT implements standard environmental measures as part of other environmental compliance processes (e.g., USACE stream and wetland permitting, EPD erosion and sedimentation requirements) that may reduce potential effects on imperiled aquatic species. These include:

- Stream and wetland avoidance/minimization/compensation
- Compliance with Georgia erosion and sedimentation control and water quality standards via the Georgia Erosion and Sedimentation Act and Georgia Water Quality Control Act

In addition, specific avoidance and minimization measures (AMMs) related to the imperiled aquatic species will be implemented where applicable. The TES and associated tools will be used to determine which AMMs are implemented at individual transportation projects to reduce potential impacts of stressors. In some cases, impacts will be reduced to levels that are insignificant or discountable and therefore NLAA imperiled aquatic species. In other cases, take will be unavoidable even with application of AMMs, but impacts and associated incidental take will be reduced. AMMs included in the proposed action include:

- *[Insert list of AMMs here. No descriptions likely necessary if they'd take up too much space.]*

A description of each AMM, including its effectiveness at reducing the potential impacts of stressors, is included in Appendix *[#]* of the TES.

On a case-by-case basis, USFWS may approve the use of innovative AMMs that the Lead Federal Agency and/or GDOT proposes to include in individual transportation projects to reduce impacts to imperiled aquatic species. Approval of an innovative AMM for one project does not constitute approval for general use for all covered projects under this programmatic agreement. The programmatic consultation and Biological Opinion can, upon the agreement of the lead federal agency be amended to include innovative AMMs that the agencies determine are appropriate for use for all covered projects.

PROGRAMMATIC CONSULTATION PROCESS

[The statements below regarding NE, NLAA, and LAA projects would be appropriate depending on the outcome of the Biological Opinion developed for this programmatic consultation and if USFWS and the Lead Federal Agency decide to include all three types of projects in the programmatic consultation.]

This programmatic consultation provides procedures for GDOT to document NE determinations. It provides advance USFWS concurrence with NLAA determinations for qualifying projects, subject to project-level verification. For LAA determinations, this programmatic consultation provides the opinion of USFWS that projects which are consistent with the programmatic requirements, including implementation of AMMs and compensation measures, are not likely to jeopardize the continued existence of the federally listed aquatic species, and provides an incidental take statement (ITS). No jeopardy determination or ITS is provided for state-listed species. We describe the project-level processes for NE, NLAA, and LAA projects using this programmatic consultation to comply with ESA section 7 below. We also describe the process for projects only impacting state-listed species.

Projects with No Effect to Imperiled Aquatic Species

There are two primary ways that projects can result in NE to imperiled aquatic species: (1) geographic location; or (2) absence of suitable habitat. If the project is completely out of the range of any imperiled aquatic species or if there is no suitable habitat within the project action area, the project will result in NE. An absence of species or suitable habitat will be documented in GDOT's Assessment of Effects Report (AOER) and no further action is required under this programmatic.

Projects Not Likely to Adversely Affect Imperiled Aquatic Species

[The Lead Federal Agency would have to approve GDOT sending the Project Submittal Form directly to the Lead Federal Agency and USFWS at the same time. The Project Submittal Form could be part of a package of documents uploaded to IPAC instead of a separate document sent to USFWS. All specific requirements below, including time frames for review, are draft suggestions based on the Indiana Bat and Northern Long-Eared Bat Programmatic Consultations and may be amended.]

The TES and associated tools are used to determine which transportation projects are NLAA imperiled aquatic species. A project is categorized as NLAA if, due to typical project characteristics or because of a reduction in potential impacts from the inclusion of AMMs through the TES process, adverse impacts to imperiled aquatic species from the project are insignificant[2] or discountable.[3] NLAA projects may rely on this consultation with no additional site-specific consultation. Instead, there is a project review period with the Georgia Ecological Services Field Office and the lead federal agency, which consists of GDOT sending a Project Submittal Form to the Georgia Ecological Services Field Office and the lead federal agency for review prior to concurrence and delivery of any ITS and project NEPA evaluation can be completed. The Lead Federal Agency and/or GDOT will ensure that all submitted projects are within the scope of and adhere to the criteria of the programmatic consultation. Upon receipt of a Project Submittal Form, the Georgia Ecological Services Field Office will check for program consistency and request any necessary additional information. The Georgia Ecological Services Field Office and the lead federal agency both have 14 calendar days to notify GDOT if they determine the project does not meet the criteria for a NLAA determination. If GDOT is not so notified, they may proceed under the programmatic consultation. This verification period is not intended as another level of review. Rather, it is an opportunity for the lead federal agency and Georgia Ecological Services Field Office to apply local knowledge to these projects, and they may identify a small subset of projects as potentially having unanticipated impacts.

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[2] "Insignificant effects" relate to the "size of the impact and should never reach the scale where take occurs… Based on best judgement, a person would not … be able to meaningfully measure, detect, or evaluate insignificant effects." USFWS and NMFS, Endangered Species Consultation Handbook xv (March 1998).

[3] "Discountable effects" are "those extremely unlikely to occur. Based on best judgement, a person would not … expect discountable effects to occur." USFWS and NMFS, Endangered Species Consultation Handbook xv (March 1998).

Projects Likely to Adversely Affect Imperiled Aquatic Species

[The Lead Federal Agency would have to approve GDOT sending the Project Submittal Form directly to the Lead Federal Agency and USFWS at the same time. The Project Submittal Form could be part of a package of documents uploaded to IPAC instead of a separate document sent to USFWS. All specific requirements below, including time frames for review, are draft suggestions based on the Indiana Bat and Northern Long-Eared Bat Programmatic Consultations and may be amended.]

The TES and associated tools are used to determine which transportation projects are LAA imperiled aquatic species. A project is categorized as LAA if adverse impacts to listed species are reasonably certain to occur. AMMs employed to meet the Effect Limit may reduce adverse impacts and associated incidental take for individual projects. Projects involving in-water work in the range of non-migratory species with suitable habitat present in the project action area will be LAA regardless of the inclusion of AMMs and incidental take will be measured.

LAA projects may rely on this consultation in a manner similar to projects that are NLAA imperiled aquatic species, described above; however, a response from the USFWS Georgia Ecological Services Field Office is required. The Lead Federal Agency and/or GDOT sends a Project Submittal Form to the Georgia Ecological Services Field Office prior to receiving Section 7 concurrence. This form:

- Describes the proposed action (e.g., type of action, location, and involved federal agencies);

- Verifies that the project is within the scope of the programmatic consultation;
- Provides a quantification of impacts (e.g., *[xxx]*); and
- Identifies all proposed AMMs for the project.

The Georgia Ecological Services Field Office will respond within 30 calendar days[4] (instead of 135 calendar days) to consultation requests that are accompanied by a complete Project Submittal Form and are then verified as covered under this programmatic consultation.

The Georgia Ecological Services Field Office response to a complete and correct Project Submittal Form for projects with a LAA determination will be to:

- Verify that all applicable AMMs are included in the project proposal;
- Verify that the project is consistent with the programmatic sideboards for covered projects;
- Verify that the project's anticipated take will not result in an exceedance of the total allowable take under this programmatic consultation;
- Provide an incidental take statement; and
- Identify any project-specific monitoring and reporting requirements, consistent with the monitoring and reporting requirements for the program as a whole (see Section *[xx]*).

[4] 30-day clock starts upon USFWS Georgia Ecological Services Field Office receipt of a complete Project Submittal Form.

Projects with Additional Information Needs

USFWS, the Lead Federal Agency, and/or GDOT may determine that a proposed project requires additional site-specific information to determine whether it conforms to this consultation. Such projects will require the Lead Federal Agency and/or GDOT to coordinate with the Georgia Ecological Services Field Office in order to make a final determination pursuant to section 7(a)(2). If a project "may affect" any other federally listed or proposed species, additional consultation (or conference for proposed species and Critical Habitats, if applicable) is required.

Projects with Impacts to State-Listed Species Only

[The DNR Wildlife Resources Division may want to develop additional procedural requirements and include them here.]

Projects in the range of state-listed species will also utilize the TES to assess impacts and identify applicable AMMs. However, these projects should submit the Project Submittal Form to the Georgia Department of Natural Resources Wildlife Resources Division, not USFWS. No incidental take permit will be issued and any take of state-listed species will not count against the total allowable take under this programmatic consultation.

STATUS OF THE SPECIES

[Appendix ##] provides the distribution of the imperiled aquatic species and an overview of the biology and conservation needs of the species that are pertinent to the "Effects of the Action" section.

Effects of the Action

This section first describes how effects to listed species are determined for individual projects and provides a summary of project-level effects by generally describing the different categories of projects (NE, NLAA, LAA). Next, it describes effects of the action, i.e., the implementation of the programmatic consultation.

Project-Level Effects

Effects from individual transportation projects are assessed using the TES. The TES is a metric of the aggregate effect of a GDOT project on an imperiled aquatic organism over a 50-year time horizon. There are five components to the TES, which include assessment of both direct and indirect effects of individual projects. These components are:

1. Direct Impact Effect Score (DIES). This is an estimate of the potential effect of physical in-stream construction activities on the local population. It can be greatly reduced by timing restrictions on in-water work.

2. Sediment Effect Score (SES). This is an estimate of the probability of a large sedimentation event occurring during construction and the effect of such an event. It is reduced by use of more advanced sedimentation AMMs.
3. Other Construction Effect Score (OCES). This incorporates construction-phase risks to species not captured in the above effect scores, such as hydraulic fluid leaks, or effects of herbicides. Since AMMs for managing these effects are required on all projects, this score is small and cannot be adjusted.
4. Post-Construction Effect Score (PCES). This is an estimate of the effect of stormwater runoff from impervious surfaces associated with the project over a 50-year time horizon. It is reduced by use of stormwater management BMPs.
5. Reconnection Effect Score (RES). This is only included in projects that reconnect species habitat through culvert removal or replacement. It requires a project-specific estimate of the area of reconnected habitat.

In order to proceed under the programmatic consultation, individual transportation projects' TES may not exceed any species-specific MES. A project potentially impacting three listed aquatic species will, for example, need to keep its TES under the specific MESs for each of the three species. The TES can be lowered depending on AMMs implemented in the project; if the TES is greater than the MES for any of the species, AMMs utilized can be adjusted in the TES worksheet and the TES can be recalculated. If the TES can not be brought below the MES for one or more species for a project, the programmatic consultation cannot be used for that project.

More information on the TES is provided in Appendix A.

[A Project Submittal Form for this programmatic consultation may want to include a section on cumulative effects, as described here.]

Cumulative effects, discussed generally for the entire programmatic consultation in Section *[xx]* below, can vary widely from project to project and as such cannot be confidently assessed using a standardized tool like the TES. To account for these effects, the Project Submittal Form includes a section for the description of cumulative impacts for individual projects. These cumulative impacts assessments will be evaluated by USFWS when reviewing Project Submittal Forms.

[The Lead Federal Agency will have to determine whether the sentence beginning "Because this programmatic consultation provides a standardized process…" accurately reflects its opinion regarding this programmatic consultation and the accompanying Biological Assessment. It is provided here as potential language to include should the Lead Federal Agency agree with its content.]

The process for assessing individual transportation projects under this programmatic consultation addresses all of the stressors most likely to adversely affect the imperiled aquatic species, including the chronic, long-term impacts of post-construction stormwater runoff. Because this programmatic consultation provides a standardized process for both quantifying these impacts and incentivizing the implementation of post-construction stormwater AMMs – a process heretofore absent from transportation project Section 7 consultations – the Lead Federal Agency has concluded that implementation of the programmatic consultation is reasonably certain to result in less adverse impacts to imperiled aquatic species than assessing transportation projects via individual Section 7 consultations. In addition, projects that cannot limit impacts below the MES for all potentially impacted species will require individual consultation.

General effects to imperiled aquatic species from each type of project (NE, NLAA, LAA) are as follows:

NE: There are two primary ways that projects can be categorized as NE: (1) geographic location, or (2) suitable habitat absence. If the project is completely out of the range of any imperiled aquatic species or there is no suitable habitat within the project area, the project will be categorized as NE.

NLAA: There are two general ways transportation projects are categorized as NLAA. First, some projects may occur near or within suitable habitat within the range of an imperiled aquatic species, but the project will result in no effect or a discounted likelihood of effect even without the implementation of any AMMs. For example, *[provide example here]*. Second, other projects may occur near or within suitable habitat within the range of imperiled aquatic species, and implementation of AMMs will avoid or minimize impacts to the point they are insignificant or discountable. For example, *[provide example here]*.

LAA: Some projects will result in adverse effects and if all effects cannot be avoided, they will be categorized as LAA. Projects involving in-water work in the range of non-migratory species with suitable habitat present in the project action area will be LAA regardless of the inclusion of AMMs, though including AMMs may reduce the amount of incidental take for the project.

EFFECTS OF THE ACTION

Effects of the action are assessed by examining the potential impacts of each stressor associated with transportation project activities (see *[Section __ and Appendix __]*) on imperiled aquatic species or their habitat.

[Provide outline of species effect analysis with short description of each component.]

The effect analysis for each species is provided in *[Appendix __]*.

CUMULATIVE EFFECTS

Cumulative effects include the effects of future State, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this BA. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

[Language regarding USFWS below is based on other cumulative effects analyses and should be reviewed by USFWS.]

Reasonably foreseeable non-federal activities that are anticipated to occur in the action area include land conversion associated with activities such as development for residential/commercial/agricultural growth and timber harvesting. Transportation projects are connected to larger road networks that can facilitate such land conversions (e.g., new roads that open up areas to new development or other land conversion, widened or otherwise enhanced roads that bring more people and potentially more development or land conversion to an area). While these road networks are linear in nature, they traverse the entirety of the State of Georgia, and associated land conversion activities are much larger with potentially significant impacts to imperiled aquatic species and their habitat. The USFWS does not have any information on the amount or types of residential, industrial, or agricultural development that have or will occur within the action area. It is difficult to predict how much land conversion will occur since these land conversion activities are typically driven by the economy/location of the activity.

Other reasonably foreseeable non-federal activities that are anticipated to occur in the action area include municipal and agricultural surface water and groundwater withdrawals. These withdrawals are influenced in part by the regional forecasts developed pursuant to the Georgia State Water Plan. These forecasts are, however, not regulatory standards and it is impossible for USFWS to estimate the extent of, and impacts from, permitted withdrawals in the action area.

Project submittal forms for transportation projects utilizing this programmatic agreement that result in LAA will have a section for information on cumulative impacts for individual projects. These projects should address any associated development related to the road project (e.g. any new alignment project would be expected to have gas stations and other services at major intersections or capacity increasing projects may show associated development plans by local municipalities or county development agencies). If USFWS finds that cumulative impacts are reasonably certain to result in unanticipated impacts to imperiled aquatic species, it might result in the agency requiring individual consultation for that particular project.

JEOPARDY DETERMINATION /BIOLOGICAL OPINION

[The following is all placeholder language for USFWS if/when it makes it jeopardy determination/biological opinion for this programmatic consultation.]

[Need separate piece for each federally listed species. These may take up a lot of room, so it may be practical to use same language for all, if that is the opinion of the USFWS, or to move these determinations to an appendix.]

<u>*[Species #1]*</u>

After reviewing the current status of the imperiled aquatic species, the environmental baseline for the action area, the effects of the action, and the cumulative effects, it is the USFWS's BO that this state-wide program of GDOT transportation activities, as proposed, is *[USFWS enters conclusion here]*.

"To jeopardize the continued existence" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of the imperiled aquatic species by reducing the reproduction, numbers, or distribution of the imperiled aquatic species (50 CFR 402.2). In Section *[xx]* of this BO, we identified the stressors associated with the various types of transportation activities included in the proposed action, and analyzed how imperiled aquatic species individuals would respond if exposed to these stressors. From this analysis, we determined that the AMMs that will apply to projects via the TES *[USFWS provides opinion on effect of TES/AMMs here]*.

[Species #2]

[Repeat…]

INCIDENTAL TAKE STATEMENT

[This entire section is placeholder language for USFWS, which will need to write the incidental take statement. Much of this was based on the Indiana Bat and Northern Long-Eared Bat Programmatic Consultations.]

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by the USFWS to include significant habitat modification or degradation that results in death or injury to wildlife by significantly impairing essential behavioral patterns including breeding, feeding, or sheltering (50 CFR § 17.3).

Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking complies with the terms and conditions of this incidental take statement.

The measures described below are nondiscretionary, and must be undertaken by the Lead Federal Agency so that they become binding conditions of any grant or permit issued, as appropriate, for the exemption in ESA section 7(o)(2) to apply. The Lead Federal Agency has a continuing duty to regulate the activity covered by this incidental take statement. If the Lead Federal Agency: (1) fails to assume and implement the terms and conditions; or (2) fails to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse.

To monitor the impact of incidental take, the Lead Federal Agency and/or GDOT must report the progress of the action and its impact on the imperiled aquatic species to the USFWS as specified in the incidental take statement (50 CFR 402.14(i)(3)).

Table ## provides estimates of the amount of estimated take for individual imperiled aquatic species that may occur during the life of this programmatic consultation. We anticipate that take is reasonably certain to result from GDOT transportation projects in the range of federally listed aquatic species, based on the stressor-exposure response analyses of section *[xx]* of the BO (see *[Appendix __]*).

Table ##. Estimated take of imperiled aquatic species			
Species name	Estimated total take	Estimated take per year	Notes

Reasonable and Prudent Measures

The proposed action includes several measures (section *[xx]* of the BO) that avoid and minimize the incidental take of imperiled aquatic species resulting from GDOT transportation projects that federal agencies fund or approve. Because the Lead Federal Agency will not typically carry out the projects it funds or approves under the proposed action, we find that the following reasonable and prudent measure (RPM) is necessary and appropriate to minimize the incidental taking resulting from such projects:

The Lead Federal Agency will ensure that GDOT, which chooses to include eligible projects under this programmatic action, incorporates all applicable conservation measures (avoidance and minimization) in the project proposals submitted to USFWs for ESA section 7 compliance using this BO.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the Lead Federal Agency must comply with the following terms and conditions, which implement the RPM described above. These terms and conditions are nondiscretionary.

[These are verbatim from Indiana Bat, Northern Long-Eared Bat Programmatic Consultations; USFWS may want to amend.]

1. The Lead Federal Agency or its representatives will offer annual instruction to appropriate personnel who are involved in developing and implementing projects for inclusion in this programmatic action. The instrument shall inform personnel about:
    a. The criteria for determining that a project is eligible for such inclusion;
    b. Developing the information required in the Project Submittal Form and describing the process for using the Determination Key; and
    c. The administrative process for using this BO as the mechanism for project-level ESA section 7 compliance.

2. The Lead Federal Agency and GDOT will make all reasonable efforts to educate personnel to report any sick, injured, and/or dead imperiled aquatic species located in the project action area during construction, operations, maintenance, or monitoring activities immediately to the USFWS Georgia Ecological Services Field Office. Due to the number of staff/contractors, it is not expected or required to educate all personnel working in the

project action area, but only those who are most likely to observe imperiled aquatic species during the course of normal working conditions.

Monitoring and Reporting

"In order to monitor the impacts of incidental take, the federal agency or any applicant must report the progress of the action and its impact on the species to the USFWS as specified in the incidental take statement" (50 CFR §402.14(i)(3)).

Annual Report

The Lead Federal Agency will provide an annual report to the points of contact, (POCs) (as described in *[section __],* below) not later than January 31 for the preceding calendar year, of all project-level activity under their programmatic action. The report will provide the information listed below, or alternative information that the Lead Federal Agencies and USFWS agree is appropriate.

- All project species survey reports.
- Total amount of take by species estimated from ITS for the calendar year.
- Total amount of take per species and a list of projects that started construction in the last calendar year
- Total amount of take per species and a list of projects that completed construction in the last calendar year, including an estimate of take and final take for each project.
    - Any adjustments in the total amount of take compared to estimated take for completed projects and narrative explaining differences.

- A summary of any violations of ITS, Special Provisions, or standard provisions and a narrative explaining measures to remedy those violations.
- *[Other information USFWS wants…]*

CONSERVATION RECOMMENDATIONS

ESA Section 7(a)(1) directs all federal agencies to "utilize their authorities in furtherance of the purposes of [the] Act by carrying out programs for the conservation of endangered species and threatened species."

[Discretionary actions that USFWS can recommend; ESA section 7(a)(1) directs agencies to carry out programs for benefit of listed species. USFWS can include actions here that are not required but that would further conservation of listed species. USFWS can request notification of implementation of these measures.]

MONITORING AND REPORTING

Monitoring

Monitoring implementation of this state-wide consultation begins with the Project Submittal Forms, from which the USFWS Georgia Ecological Services Field Office will log key information in TAILS. Further, the points of contact (POC) will acquire additional information (e.g., *[xx]*) as necessary from GDOT and the Georgia Ecological Services Field Office. This information will be discussed at the annual review, as described above, and inform adaptive management.

Reporting

The Lead Federal Agency and/or GDOT must provide the USFWS Georgia Ecological Services Field Office a Project Submittal Form for each project they wish to include in this program for purposes of ESA section 7 compliance. The USFWS Georgia Ecological Services Field Office will enter specific data from these forms into the TAILS system.

Agency POCs will coordinate to compile the site-specific information collected for each project using the programmatic consultation into an annual report (see above). The annual report will allow the POCs to track the number of projects, type of action, *[feet/acres/area]* of habitat affected, amount of incidental take per species, etc. This report will also be used for adaptive management as described above.

ADAPTIVE MANAGEMENT

[This is all pretty much verbatim from Indiana Bat/NLEB programmatic.]

The Lead Federal Agencies and USFWS will review feedback from users and new information regarding the species' ecology, conservation, and project efforts to adaptively manage how the program is working to serve its two main purposes: streamline the consultation process and promote better conservation outcomes for imperiled aquatic species. Adaptive management guidelines are provided and the agencies will develop a more detailed adaptive management plan.

The Lead Federal Agencies and USFWS will designate POCs that will have responsibility for the ongoing implementation of the programmatic consultation. The POCs will coordinate adaptive management and reporting at the programmatic scale (analyses of data, implementation of program changes, reporting out to management, etc.). The POCs may also stipulate areas of the programmatic consultations for more active data gathering as circumstances warrant.

The Lead Federal Agencies and USFWS will conduct an annual review of the program. There will also be a five-year review, at which time the Lead Federal Agencies and USFWS shall have an opportunity to renew or amend the programmatic consultation. Standard consultation reinitiation conditions (e.g., new information on species or effects) apply. Additional information may warrant changes to the programmatic BA, either for the entire range of the species or specific geographic areas within the range. For example, new data about the impact of sedimentation on a species could warrant revising the sediment sensitivity rating for that species. Although the agencies will consider changes to the consultation on an annual cycle, such changes may occur at any time the agencies mutually agree is appropriate. At any time, the Lead Federal Agencies or USFWS may withdraw from participating in the program if either determines the program is failing to serve the intended purposes.

The annual meeting of POCs from USFWS and FHWA will:

- Discuss the annual report of covered projects;
- Assess information received from the field and new relevant research over the preceding year and determine, by consensus, whether such information warrants changes to the program;
- Evaluate the effectiveness of the tracking and reporting processes;
- Evaluate the effectiveness of the program to streamline the consultation process and conserve the imperiled aquatic species; and
- Discuss and resolve any issues related to the program.

REINITIATION NOTICE

[Statement that consultation is concluded.] Reinitiation is required when "discretionary federal agency involvement or control over the action has been retained (or is authorized by law)" and if:

- The amount of incidental take of any imperiled aquatic species is exceeded;
- New information reveals effects of the action that may affect imperiled aquatic species in a manner or to an extent not considered in this BO;
- The agency action is subsequently modified in a manner that causes an effect to imperiled aquatic species not considered in this BO; or
- A new species is listed that may be affected by the action.

If the lead federal agency does not retain discretionary involvement, GDOT will be required to coordinate under Section 7 with any new lead federal agency. The determinations made through this programmatic agreement are not transferrable to another lead federal agency that is not a signatory to this programmatic agreement.

Per the first condition above, the anticipated incidental take is exceeded when, in *[one]* calendar year, transportation projects *[exceed incidental take for an aquatic listed species.] [Probably need table with all species and allowable take below.]*

LITERATURE CITED

[Insert]

<u>Appendix A: Total Effect Score and associated materials</u>

[Insert]

CHAPTER 6. EFFECTS OF ROAD CONSTRUCTION ACTIVITIES ON AQUATIC SYSTEMS AND BIOTA

This chapter describes the effects of road construction activity on biota and the specific mechanisms of action. The effects of routine construction activities can be broadly grouped according to their impacts on aquatic systems: sediment, pollutants, noise, altered hydrology/connectivity, and physical contact. The effects of each impact, or stressor, are assessed for taxonomic groups, and then are applied to individual species in the Biological Assessment appendix (Appendix C). Unfortunately, basic biological and ecological information for many species are missing, as are data on the effects of stressors on most individual species. The research team supplements the limited available life history information with studies conducted on closely related taxa and inferences based on known sensitivities of species with similar traits and characteristics. Species with common reproductive characteristics are assigned to "reproductive guilds" based on spawning habitat, reproductive behavior, and physical characteristics of eggs/embryos. Some of these guilds are more sensitive to sedimentation or to contaminants than others, so reproductive guild membership is one line of evidence in determining overall sensitivity of a species. Similarly, "feeding guilds," sometimes referred to as trophic guilds, are based on food type, feeding position within water column, and mode of foraging, which can confer more or less exposure to contaminants.

The analysis of construction effects, and the Total Effect Score system, is focused on stream and river habitats. Organisms that do not reproduce in the water (e.g., several species of turtles) and which spend most of their time in riparian habitats (e.g., burrowing crayfish) are less affected by the mechanisms the team examined. However, these organisms may be affected directly by construction activities in riparian or wetland habitats. Those effects are not examined here, and projects that have a major impact on riparian or wetland habitats that are critical for protected species might require consultation outside of the proposed PA.

The sections below are subdivided by taxonomic group. The research team does not include snails as a group, since only one species (the interrupted rocksnail, *Leptoxis foreman*) is included; please refer to the relevant section of Appendix C for information on this species. Note that this chapter is meant to be used jointly with Appendix C. Appendix C references the syntheses provided in this chapter and applies them to each individual species to determine sensitivities to sediment and pollutants, the most critical stressors. Sensitivity to noise is assumed to be "medium" for most species, because very little data are available to assign species to sensitivity categories. The exception is for fish species, which are assigned to either a medium or high category based on traits, as described below. Physical contact sensitivity is considered "medium" for fish (which are mobile) and mussels (which are usually relocated) and "high" for all other taxa. All species are considered equally sensitive to altered hydrology/connectivity because the research team assumes this stressor is adequately management by required AMMs.

EFFECTS OF SEDIMENT ON BIOTA

Sedimentation is considered the primary potential impact of construction activities at stream crossings (Cocchiglia et al. 2012). Fortunately, of the several broad classes of construction impacts, erosion and sedimentation has the largest body of scientific literature on effects and mechanisms of action. While sedimentation is a natural process required for proper functioning of aquatic systems, rates and volumes of erosion and sedimentation now far exceed natural background levels and constitute a leading cause of impairment of water quality across the US (Izagirre et al. 2009; Richardson and Jowett 2002; Wood et al. 2005; EPA 2017). Research on the biological effects of sedimentation focuses on sand, silt, and clay particles, collectively referred to as “fine sediment.” Fine sediments are most often defined as particles <2mm, but definitions in the literature range from <1mm to <6.4mm. Suspended sediments can be directly measured as total suspended solids (TSS) or indirectly by measuring turbidity (the inhibition of light transmission).

Transportation construction activities represent a potential acute source of sedimentation to receiving waters. Amounts and characteristics of sediment expected over the course of a construction project depend on erosion and sedimentation control measures, geomorphological stability within the action area, rainfall patterns, local topography, and riparian characteristics. While a variety of measures are commonly employed to eliminate or reduce sedimentation, catastrophic failure of such measures may occasionally occur, resulting in a pulse of suspended sediment followed by increased bedded sediments after settling. Both suspended sediment and bedded sediment, collectively referred to as suspended and bedded sediments (SABS), have the potential to adversely affect aquatic organisms.

The ways in which SABS can affect aquatic organisms can broadly be divided into those that directly affect individuals and those that alter the physical environment, indirectly affecting the resident biota. Suspended sediment may directly damage or impair respiratory organs, trigger a stress response, or interfere with foraging and reproductive activities (Bilotta and Brazier 2008). Settling of sediments may smother prey and incubating embryos. In addition to the direct effects of SABS on organisms, SABS indirectly affect biota at the larger system-wide scale. Suspended sediments in areas of high flow may abrade macrophytes and associated periphyton or scour streambeds of habitat for macrophytes. Suspended sediments elevate turbidity, which reduces both the degree and depth of light penetration and inhibits photosynthesis. The reduction in energy available to primary producers results in cascading effects on higher trophic levels through the system; first on phytoplankton and periphyton, then zooplankton, then macroinvertebrates, and then fish (Newcombe and Macdonald 1991). Bedded sediments resulting from large or chronic inputs may infill coarse substrate, increasing embeddedness and decreasing benthic oxygen levels (Wharton et al. 2017). This loss or degradation of benthic habitat may adversely affect spawning success and the availability of food such as periphyton or macroinvertebrates (Henley et al. 2010). The direct effects of SABS on biota and indirect effects on habitat and water quality alter rates of growth and survival at all trophic levels, ultimately leading to a shift in the structure and composition of the community if the scale of the impact is large enough (Wood and Armitage 1997).

While direct effects on dragonflies and freshwater mussels, are assessed in subsequent sections, here the research team briefly considers general effects of sediment on aquatic macroinvertebrates (particularly aquatic insects) because of their importance as prey to fish and crayfish. Suspended sediments can abrade and clog sensitive respiratory and filter-feeding organs of aquatic macroinvertebrates (Kefford et al. 2010). Suspended sediments have also been shown to degrade the quality and quantity of food available to macroinvertebrates through scouring (of macrophytes, periphyton, and zooplankton), food infiltration (e.g. periphyton), and inhibition of primary productivity via higher turbidity (Hart 1992; Graham 1990; Izagirre et al. 2009; Yamada and Nakamura 2002). After settling, sediments may infiltrate coarse substrate, impairing water quality and degrading or eliminating habitat used for reproduction or foraging (Sear et al. 2014; Greig et al. 2007). Bedded sediments can also directly smother both adult macroinvertebrates (Conroy et al. 2018; Dobson et al. 2000) and their eggs/larvae (Rutherford and Mackay 1986; Kefford et al. 2010). To various degrees, these effects may be reduced by behavioral responses such as cleaning of breathing and feeding organs, seeking of refugia, and emigration (i.e. drift; Jones et al. 2012). Elevated levels of SABS have consistently been shown to increase drift of macroinvertebrates (Suren and Jowett 2001; Doeg and Milledge 1991).

The above mechanisms acting on a large scale (either in terms of sediment volume or duration of exposure) may result in reduced growth rates and elevated mortality (Donohue and Irvine 2003; Kefford et al. 2010). At the population level, SABS will alter abundance and diversity (Culp, Wrona, and Davies 1986; Larsen, Pace, and Ormerod 2011), leading to a shift of the community composition toward sediment-tolerant species (Wood and Armitage 1999; Rabení et al. 2005; Herrera 2016; Matthaei et al. 2006; Bo et al. 2007). Those species and taxonomic groups dependent upon macroinvertebrates as prey (e.g. fish and crayfish) will in turn shift toward communities more tolerant of sedimentation (Shaw and Richardson 2001). As with other taxonomic groups, the degree and nature of the cumulative effects of SABS on macroinvertebrates will vary by type of exposure (concentration, duration, frequency), physical characteristics of particles, sensitivity of species/group, local hydrologic conditions, available refugia, biological relationships, interactive effects with other stressors, and other factors.

Effects of Sedimentation on Fish

At the individual level, SABS may reduce survival, growth, and reproductive success, contributing to population-level changes in abundance and persistence (Richardson and Jowett 2002; Sullivan and Watzin 2010). Collectively, these effects on multiple species may alter richness and diversity, ultimately leading to shifts in community composition and structure favoring species within guilds more tolerant of sedimentation and turbidity (Chapman et al. 2014; Richardson and Jowett 2002; A.B. Sutherland, Meyer, and Gardiner 2002; Sullivan and Watzin 2010). Of the various taxonomic groups assessed in this report, most research on the effects of SABS has focused on fish. Effects on the adult life stages of salmonids have been particularly well studied due to their economic importance (Newcombe and Jensen 1996). Much less attention has been devoted to warmwater species and early life stages (Muck 2010; Bash, Berman, and Bolton 2001).

Some effects of elevated SABS act directly on individual fish via gill abrasion, impaired swimming, and elicitation of a stress response (Kemp et al. 2011). However, most effects acting on fish can be grouped according to their effects on the efficiency and success of foraging and reproduction (Berry et al. 2003). The nature, response pathway, severity, and duration (short- vs long-term) of these effects are influenced by a number of biotic and abiotic factors. Some biotic factors include: species, condition, life history, life stage of fish, and local community composition (Wilber and Clarke 2001). Abiotic factors include: sediment characteristics, nature of sediment exposure (concentration, duration, frequency, and timing), turbidity, substrate characteristics, local hydrology, and presence of refugia (Birtwell 1999). Biological responses are also influenced by interaction with additional factors: sediment-bound contaminants and organic material, temperature, pH, and dissolved oxygen (Kjelland et al. 2015). Because fish, especially predatory species, occupy the upper trophic levels in freshwater systems, they are indirectly affected by changes at lower trophic levels resulting from elevated levels of SABs (Karr 1991; Kemp et al. 2011).

While acute or chronic exposure to suspended sediment can cause mortality (Servizi and Martens 1991; Lake and Hinch 1999), it is more likely that fish, as highly mobile animals, will experience a range of sublethal effects. Examples of sublethal effects include impaired swimming performance, reduced foraging and reproductive success, and altered predator/prey dynamics (Bruce A Barton 1977; Snow, Shoup, and Porta 2018; Wilkens et al. 2015). Lethal concentrations of suspended sediment generally range from hundreds mg/L to hundreds of thousands mg/L (Birtwell 1999). Concentrations ranging from tens mg/L to hundreds mg/L generally result in sublethal effects (Bruce A Barton 1977; Birtwell 1999; Kjelland et al. 2015). Prolonged exposure to lower concentrations of SABS may also prove lethal following a stress response and inhibition of the immune system, allowing for opportunistic infections (further discussed below).

In limited suspended sediment exposures, effects are often mitigated by behavioral (avoidance/emigration) and physiological responses: increased mucus production, higher rate of coughing, and gill flaring (Berg and Northcote 1985; Goldes et al. 1988). In prolonged exposures at higher concentrations, more severe damage to the gills is caused: branchial erosion, fusion of lamellae, and incorporation of particles into gill tissue (Goldes et al. 1988; Lake and Hinch 1999; Wong, Pak, and Jiang Liu 2013). Any significant damage to the gill surface interferes with gas exchange and osmoregulation contributing to reduced growth rates and population effects (Goldes et al. 1988; A. Sutherland and Meyer 2007; Swinkels et al. 2014).

Greater exposures can also trigger an increase in circulating cortisol and a systemic stress response: hyperplasia, hypertrophy, and increased hematocrit (Awata et al. 2011; Lake and Hinch 1999; Michel et al. 2013; Redding, Schreck, and Everest 1987). The stress response, depending on duration, is accompanied by a reduction in growth and feeding, an inhibition of the immune system, and ultimately a reduced probability of survival. The stress response is an energetically expensive process, requiring reallocation of energy reserves away from growth and reproduction toward increased metabolic rate and production of heat-shock proteins (Bruce A. Barton, Schreck, and Barton 1987; Contreras-Sanchez et al. 1998; McCormick et al. 1998). The inhibition of the immune system increases vulnerability to opportunistic bacterial, viral, and parasitic infections (Goldes et al. 1988; Servizi and Martens 1991). Stress can also impair reproductive efforts by delaying or preventing spawning and by reducing egg quantity and quality (Bash, Berman, and Bolton 2001; Schreck, Contreras-Sanchez, and Fitzpatrick 2001).

Elevated SABS has been shown to directly impair feeding in fish as well as indirectly alter forage availability and influence predator/prey dynamics, ultimately leading to shifts in community structure (Becker et al. 2016; Kemp et al. 2011; Kjelland et al. 2015). A primary effect of suspended sediment is an increase in turbidity, which mediates many of these effects. Elevated turbidity reduces fish reaction distance and therefore the perception of predators, prey, and conspecifics. The impaired visual acuity of sight-feeding species reduces foraging efficiency, contributes to lower individual growth rates, and may lead to negative population-level responses (Berkman and Rabeni 1987; Chapman et al. 2014; Mcleay et al. 1987). Visual predators, whether piscivores or invertivores, are especially susceptible to the deleterious effects of increased turbidity on foraging efficiency (Vogel and Beauchamp 1999; Carter et al. 2010; Sweka and Hartman 2003; Hazelton and Grossman 2009; Huenemann, Dibble, and Fleming 2012; Shoup and Wahl 2009; Utne-Palm 2002; Zamor and Grossman 2007). Species that rely on non-visual perception of prey (olfactory, tactile, electro-reception) are unsurprisingly less affected by turbidity (Ali, Ryder, and Anctil 1977; Rowe and Dean 1998). However, ingestion of inorganic materials may still impose a significant energetic burden (Power 1984).

Fish behavioral responses to turbidity mitigate its deleterious effects to varying degrees. In the case of temporary sediment pulses, fish may seek temporary refugia, avoid sediment plumes, or emigrate from turbid areas (Berg and Northcote 1985; Bisson and Bilby 1982; Mori et al. 2018). The success of these responses depends on the presence and quality of alternative habitat within movement range of the species. Fish often increase time and daily movement devoted to foraging in turbid conditions. This behavior may be in compensation for reduced feeding efficiency or due to a reduced perception of risk to predators (Robert S. Gregory 1993; Kuliskova et al. 2009).

Some species benefit from short- or long-term limited increases in turbidity above the natural background level. Pulses of suspended sediment temporarily increase insect drift and therefore food availability for some fish (Jones et al. 2012). However, long-term elevation of SABS decreases abundance and diversity of macroinvertebrates (Bo et al. 2007; Angradi 1999; Wood et al. 2005). Increased turbidity has also been shown to decrease the perceived risk of predation and increase time spent foraging and away from cover (Gradall and Swenson 1982; R. S. Gregory and Northcote 1993; Robert S Gregory and Levings 1996; Miner and Stein 1996). In addition to species using non-visual perception to feed, planktivores have shown more efficient foraging in turbid conditions due to the concentration of zooplankton toward the surface.

The effects, whether deleterious or beneficial, of SABS on foraging behavior and efficiency in fishes can be roughly grouped into guilds. Visual predators, especially piscivores, are adversely affected by the impairment of visual acuity due to elevated turbidity (Bonner and Wilde 2002; Nieman and Gray 2018; Schwartz, Simon, and Klimetz 2011; Townsend and Risebrow 1982). Benthic invertivores and water column feeders are more negatively affected by SABS than co-occurring omnivores (Berkman and Rabeni 1987; Sullivan and Watzin 2010). Other foraging guilds, such as planktivores or the olfactory-foraging catfishes, may receive a competitive advantage, relative to other guilds. Some species are adapted to foraging in low-light conditions through use of a well-developed lateral line system (Rowe and Dean 1998) or the presence of adaptations for improved vision, such as cyprinids (Diehl 1988; Sandström and Karås 2002) and some percids (the tapetum lucidum in *Sander* sp. and *Gymnocephalus* sp.) (Sandstrom 1999).

As mentioned above, the visual acuity and reaction distance of predators, especially piscivores, is impaired by the increased scattering of light by dissolved and suspended particles in areas of greater turbidity. Visual acuity is less impaired by the much smaller number of interfering particles for fish visually foraging at very short distances, such as planktivores or larval/juvenile-stages (Utne-Palm 2002). Short-distance visual foragers may actually benefit from higher contrast between their prey and the turbid background. This improved visual acuity, coupled with a reduction in risk of predation, may increase both time spent and efficiency of foraging (R. S. Gregory and Northcote 1993; Utne-Palm 2002).

In spite of the above potentially beneficial aspect of limited turbidity to early fish life stages, research has shown the general relationship with SABS to be more complex. Turbidity has been shown to reduce growth in salmonid fry and reduce foraging efficiency in larval striped bass and juvenile Chinook salmon (Breitburg 1988; R. S. Gregory and Northcote 1993; Sigler, Bjornn, and Everest 1984). SABS also reduced growth and survival of juvenile steelhead via a shift in the macroinvertebrate community (Suttle et al. 2004). This relationship is further demonstrated by the active avoidance of suspended sediments by salmonid smolts and juveniles (Birtwell 1999; Bisson and Bilby 1982).

Individual growth and population persistence also require the presence of habitat (substrate, vegetation, etc.) suitable for the production of forage and prey communities such as insects and other invertebrates (Hegge, Hesthagen, and Skurdal 1993). Increases in bedded sediments can reduce habitat complexity by filling in pools and embedding substrate (Berkman and Rabeni 1987; Yarnell, Mount, and Larsen 2006). In addition to directly and indirectly providing forage and prey, aquatic vegetation acts as cover from predation for adults and juveniles, improves water quality, and serves as critical spawning habitat for some species (Crowder and Cooper 1982; Langler and Smith 2001). As mentioned in Section XX, SABS may lower the abundance and/or change the composition of the macrophyte assemblage via altered embeddedness, scouring, or inhibition of photosynthesis. Such changes in substrate complexity and the macrophyte community may negatively affect fish abundance and diversity, leading to altered community structure (Berkman and Rabeni 1987; Brown, Gregory, and May 2009; Langler and Smith 2001; Osmundson et al. 2002; Sandström and Karås 2002; Sullivan and Watzin 2010).

The severity of effects of SABS on reproductive success of a given species is a function of biotic and abiotic factors acting at multiple early life stages, including: courtship behavior, spawning behavior, local hydrology, preferred/available substrate, prey community, vegetation, and the nature of the sediment exposure itself. Acting through a variety of mechanisms discussed below, SABS increase mortality rates at the egg, embryo, larval, and juvenile life stages, contributing to overall decreases in richness and abundance (Chapman et al. 2014; Osmundson et al. 2002; Sandström and Karås 2002; Sear and DeVries 2008; Soulsby et al. 2001). As with other aspects of the effects of SABS on fish, research has focused on salmonids.

Prior to spawning, SABS can directly affect adults in a number of ways. SABS may reduce or delay spawning activity and lower total egg production (Burkhead and Jelks 2001; Saunders and Smith 1965). Stress resulting from the presence of suspended sediments can alter the timing or occurrence of reproduction and reduce egg quantity and quality (Bash, Berman, and Bolton 2001; Schreck, Contreras-Sanchez, and Fitzpatrick 2001). By interfering with visual perception of conspecifics, elevated turbidity has also been shown to impair social interactions such as territoriality and courtship, leading to reduced overall reproduction (Burkhead and Jelks 2001; Mcleay et al. 1987).

The spawning and embryonic development periods are regarded as the life history stages with the highest levels of individual mortality and are therefore the most critical to overall reproductive success (Gray, Chapman, and Mandrak 2012). One characteristic of these early stages contributing to such high levels of mortality is the immobility of eggs and incubating embryos and their reliance on the suitability of the immediate habitat. Habitat chosen for spawning/incubation must be initially suitable and must also provide hospitable conditions for the duration of the developmental period.

Sedimentation events during the spawning season can negatively affect eggs or larvae in a number of ways (Clark Barkalow and Bonar 2015; George et al. 2015; Jennings et al. 2010; Moon et al. 2014; Suedel, Wilkens, and Kennedy 2017). Suspended sediment in areas of high flow can reduce survival of embryos on or within substrate via scouring (Lapointe et al. 2004). Fine sediments settling directly the membrane surface block micropores and impair movement of oxygen into the egg/embryo (S.M. Greig, Sear, and Carling 2005; Yamada and Nakamura 2009). Increased embeddedness by bedded sediment reduces the flow of water through the substrate, dropping levels of DO and inhibiting the removal of metabolic wastes (Duerregger et al. 2018; S. Greig, Sear, and Carling 2007). This reduction of intra-substrate circulating levels of DO can be exacerbated by the overall decrease in DO resulting from decomposition of organic material associated with eroded sediment; though this effect is more pronounced in agricultural sediments. Lastly, settled bedded sediment can block the upward movement of nascent larvae toward the water column, alter timing of emergence, and increase mortality (Bašic et al. 2019; Jensen et al. 2009; Crisp 1993).

Increased embeddedness of substrate may degrade or eliminate suitable spawning habitat, with the severity of effect often differing by spawning guild. Gravel-, broadcast-, cavity-, and crevice-spawners, are especially susceptible to elevated embeddedness (Berkman and Rabeni 1987; Mol and Ouboter 2004; Osier and Welsh 2007). Other species' spawning modes are much less susceptible to SABS, fin-fanners (ex. centrarchids) and mound-builders (ex. some cyprinids), and may actually have a competitive advantage in sedimented environments (Maitland and Hatton-Ellis 2003; Morris 1954; A.B. Sutherland, Meyer, and Gardiner 2002). While members of some guilds can initially reduce the deleterious effects of bedded sediment by fanning chosen nest sites (salmonid redds), this behavior does not mitigate the effects of subsequent sedimentation events. Species that spawn on macrophytes are less susceptible to potential smothering by sediment pulses. However, reproduction by these same species may be inhibited if SABS scour or smother requisite macrophytes.

Juveniles are less able than adults to minimize the negative effects of pulses of sediment due to their more limited mobility (Wood and Armitage 1999). Because of the greater risk of predation, juveniles are generally more reliant on the presence of vegetation and other forms of cover (Bardonnet and Heland 1994; Fausch 1993; Mitchell et al. 1998; Shirvell 1990). Increased embeddedness of substrate by SABS may impair the function of substrate to serve as both cover for juveniles and as habitat for prey organisms. (Hegge, Hesthagen, and Skurdal 1993; Langler and Smith 2001; Osmundson et al. 2002; Suttle et al. 2004).

Most research investigating the many effects of SABS approach the subject as a single stressor. In real world situations, fish and other biota contend with a variety of combined biological, chemical, and physical stressors. A stress response induced by exposure to SABS may prove lethal or be exacerbated by multiple interacting stressors such as contaminants, elevated temperatures, or lowered dissolved oxygen (DO). Pesticides, herbicides, metals, PAHs, and other contaminants washed from urbanized areas or roadways may interact with SABS to adversely affect fish (Herbrandson, Bradbury, and Swackhamer 2003; Holmstrup et al. 2010; Kimberly and Salice 2014; Schiedek et al. 2007; Schulte 2007). Interactive effects and response pathways for sediment-bound contaminants are discussed in the Pollutants sub-section. Elevated water temperatures, whether natural or otherwise (loss of riparian zone, poor stormwater management, etc.), exacerbate hypoxic stress directly by increasing metabolic oxygen demand and indirectly by lowering the oxygen content of water (Bonga 1997). Elevated temperatures and lowered DO have both been shown to interact with SABS via multiple pathways (Liljendahl-Nurminen, Horppila, and Lampert 2008; Mari et al. 2016; Nyanti et al. 2018). Dissolved oxygen is further decreased both by the decomposition of organic material associated with eroded sediments and the system's overall reduction in levels of photosynthesis due to elevation of turbidity by suspended sediment. Bedded sediments reduce benthic water flow near and within the coarse substrate, inhibiting exchange with water nearer the surface and higher in oxygen. Fish populations geographically near the upper limits of thermal tolerance are especially vulnerable to the deleterious effects of temperature elevation and its interaction with other stressors.

As mentioned above, the members of some ecological guilds (e.g. reproductive and foraging guilds) are more sensitive to the presence or introduction of SABS than others. Because of the inability of early life stages to avoid SABS, the reproductive fish guilds are likely the most important in determining the sensitivity of a given species. Crevice-spawning species (e.g. Cyprinella) and gravel/cobble spawners are likely more sensitive because of their need for clean substrate both initially and for the duration of embryonic development (Berkman and Rabeni 1987, Schwartz 2011, Sutherland et al 2002; Helms et al 2005). Species that attach their eggs to structure such as boulders or macrophytes are likely less sensitive to SABS because eggs/embryos are elevated above where SABS settle out. Nest-building species (e.g. mound-building cyprinids) may be relatively insensitive to the initial presence of sediment because they can construct a sediment-free nest (Sutherland et al 2002), but still sensitive to sedimentation events following fertilization. Nest-fanning species that first build and then maintain their nests free of fine sediments during embryonic development (e.g. centrarchids) are less sensitive (Sutherland et al 2002; Helms et al 2009). Membership within a foraging/trophic guild is also relevant to estimation of a species' sensitivity to SABS. Invertivorous fish species are likely to be more sensitive both to the initial presence of bedded sediments and sedimentation events because of the documented adverse effects of SABS on their prey base (Berkman Rabeni 1987, Sullivan and Watzin 2010). Omnivorous species are likely not sensitive because of their ability to adjust foraging habits in response to sedimentation presence or events (Schweizer and Matlack 2005, Sullivan and Watzin 2010). Some fish species are considered habitat specialists due to the narrow range of habitat characteristics where they occur (e.g. spring-dwelling fishes). A sedimentation event may have a disproportionate adverse effect on these species because of the difficulties and risks associated with relocation to alternative preferred habitat (Alford 2013).

Effects of Sedimentation on Mussels

Sedimentation from anthropogenic activities is considered a primary factor for classifying more than 40% of U.S. river kilometers as impaired (USEPA 1990). Other than habitat destruction, sedimentation is the most widely invoked explanation for mussel declines (Brim Box and Mossa 1999). A characteristic feature of many declining mussel populations is a lack of juveniles, suggesting that one or more factors is limiting reproduction or recruitment (Haag 2012). Despite widespread implications of the role of sediment in mussel declines (see Haag 2012); the few substantive studies on the topic allow minimal generalization to the broad scale at which declines have occurred (Gascho Landis and Stoeckel 2016). Sediment can be broadly categorized as bedded (e.g. deposited or embeddedness) or as suspended, each with their own adverse effects on aquatic fauna such as mussels.

Fine bedded sediments adversely affect aquatic organisms by filling interstitial spaces between larger particles (Wood and Armitage 1997; Brim Box and Mossa 1999, Henley et al. 2000) and reducing available space for benthic organisms, particularly juvenile freshwater mussels (Petts 1984, Richards and Bacon 1994, Geist and Auerswald 2007). Clogging of interstitial spaces in streams most adversely affects recruitment by smothering channel bed habitat and suffocating juveniles (Bauer 1988, Brim Box and Mossa 1999, Osterling et al. 2010, Denic et al. 2014, Denic and Geist 2015, Hansen et al. 2016, Pulley et al. 2019). The embeddedness results in insufficient hyporheic exchange between free-flowing water and the interstitial zone (Geist and Auerswald

2007) and the insufficient exchange encourages anoxic conditions harmful to juvenile mussels (Belanger 1991, Geist and Auerswald 2007). Juvenile mussels are likely more susceptible to the effects of embeddedness because they are smaller and less mobile than adults, who have the ability to move several meters per day to seek out better habitat.

Suspended sediment usually occurs as pulses at very high levels only for short periods following storm events (Waters 1995), but potentially has a wide range of direct or indirect effects on mussel physiology and reproduction. Suspended sediments are reported to clog mussel gills (Ellis 1936) and reduce feeding and respiration rates (Aldridge et al. 1987). The capability of mussels to evade stressors such as sediment is limited due to their generally sessile life history, so mussels avoid stressors by reducing their filtration rate through closing their valves partially or entirely for extended periods (Kramer et al. 1989, Ostroumov 2001, McIvor 2004). Reduced filtration rate due to elevated suspended sediment has been reported to lead to reduced mussel growth (Foe and Knight 1985…in Tuttle et al. 2019). More recently, Tuttle-Raycraft et al. (2017) reported significant reductions in clearance rate (suspension feeding rate) for juveniles and adults of four species of freshwater mussels (*Lampsilis fasciola, L. siliquoidea. Ligumia nasuta,* and *Villosa iris*) exposed to elevated suspended solids at concentrations ≥ 8 mg/L. The decrease in feeding was fivefold greater in juveniles than adults, indicating that vulnerability to this stressor differs across life stages. Similarly, high levels of suspended solids were shown to decrease respiration rates in zebra mussels (Madon et al. 1998…in G-L 2016) and marine clams (Grant and Thorpe 1991). Reduced filtration rate and the resulting reduced respiratory efficiency may explain the reduction in glochidia development reported in brooding *Regenia ebenus* by Gascho Landis and Stoeckel (2016). These results suggest that effects of suspended sediments may occur throughout the Unionid mussel lifecycle and may differ by reproductive strategy and other guilds.

Effects of elevated levels of suspended sediment on mussel reproduction may include several stages of their complex lifecycle. High suspended sediment levels may disrupt fertilization by interfering with sperm transfer to females. For example, environmentally relevant (>20 mg/L) elevated suspended solid (TSS) concentrations resulted in a sharp reduction in the number of female *Ligumia subrostrata* that became gravid, though the specific mechanism was not identified (Gascho Landis et al. 2013). As mentioned previously, reduced filtration rate and the resulting reduced respiratory efficiency in brooding females may result in a reduction in glochidia development. In addition, glochidia attachment and metamorphosis can be reduced by elevated suspended sediment. Beussink (2007) reported that very high concentrations of suspended clay (1250 – 5000 mg/L) resulted in reduced attachment and metamorphosis success of *Lampsilis siliquoidea* glochidia on fishes (Beussink 2007). As stated previously, reductions in clearance rate (suspension feeding rate) due to sediment exposure were five-fold greater for juveniles than adults, indicating stage-specific vulnerability to sediment. Gascho Landis and Stoeckel (2016) reported different effects of elevated organic TSS concentrations in a short-term brooding species (*Reginaia ebenus*) and a long-term brooder (*Ligumia subrostrata*). Suspended sediment reduced recruitment for both species, but the mechanism differed and was more severe for the short-term brooder. In the short-term brooder (*R. ebenus),* eggs were fertilized but few glochidia developed at TSS >20 mg/L; in contrast, the long-term brooder (*L. subrostrata)* had low fertilization rates at TSS >20 mg/L but those females with fertilized eggs successfully produced viable glochidia. Furthermore, elevated suspended sediment may decrease visibility of

Unionid lures to fishes, thereby reducing the probability that the parasitic glochidia will successfully attach to a suitable host (McNichols et al. 2011). In a similar manner, sediment may also reduce the visibility of conglutinate packets of glochidia and superconglutinates, which require visual attraction of a suitable host fish. Sediment may also cause host fish to seek better habitat, thereby limiting the reproductive potential of a mussel population. Fish species, such as darters, less tolerant of elevated sediment are hosts for several of the imperiled mussels in GA and may be quickly extirpated from an area receiving sediment pollution (see Effects of Sediment on Fishes).

Differences in sediment effects on short-term and long-term brooders may be explained by differences in gill structure in these reproductive guilds; short-term brooders typically utilize all 4 gills for brooding glochidia, whereas long-term brooders generally utilize only 2 of their 4 gills for glochidia. Despite the shorter brooding time, respiratory demands remain and the limited gill surface area available is likely critical for uptake of oxygen, ions and other nutrients and discharge of ammonia and other waste products. In addition, short-term brooders are widely known to abort their glochidia when exposed to stressors. Effects of sediment on abortion of broods has not been described but is possible and worthy of additional study. For long-term brooders, the additional respiratory surface area afforded by having 2 gills remain dedicated to respiration likely makes them less sensitive to respiratory distress that may be induced by elevated suspended sediment.

Because of the differential effects of sediment on short-term versus long-term brooders, these reproductive life history traits, or “reproductive guilds,” can serve as a clear means by which mussel species may be grouped into sediment tolerance categories. In addition to the reduced glochidial development and increased likelihood of brood abortion, short-term brooders may also be more sensitive to sediment by virtue of the synchronous timing of their reproductive periods such as spawning and brooding. If a sedimentation event occurs during the spawning/brooding period, a greater proportion of the potential year-class would be effectively eliminated by a given sedimentation event. Species with protracted or asynchronous spawning/brooding periods have a greater chance of recruiting juveniles to the population outside of the sedimentation event.

Direct and indirect effects of sediment on fertilization success, glochidia viability and development, glochidia attachment to suitable hosts, survival of juveniles, respiration of mature adults (brooding and non-brooding), and presence of suitable host fish clearly demonstrate the potential for adverse effects on mussel populations in streams and rivers with sediment pollution, both embedded and suspended. The effects of sediment are difficult (or impossible) to isolate in a field setting due to the complex nature of ecosystems; for example, other pollutants such as pesticides or persistent organic compounds and metals are often associated with sediment. Sediment is also often associated with urbanization, which usually introduces a slew of additional stressors such as thermal pollution, altered hydrology, habitat fragmentation (genetic isolation), and altered land cover, just to name a few.

Effects of Sediment on Crayfish

SABS have been implicated in the declines of numerous crayfishes including the Shasta Crayfish (Light et al. 1995), Slenderclaw Crayfish (USFWS 2018), Nashville Crayfish (USFWS 1986), Cave Crayfish (USFWS 1993), Big Sandy Crayfish, and Guyandotte River Crayfish (USFWS

2016). Because of a relative scarcity of research devoted to the effects of sediment on crayfishes, a portion of the literature reviewed here draws from data on non-native crayfishes, which may differ in their biology and ecology from native crayfishes.

At the individual level, SABS may reduce survival, growth, and reproductive success of freshwater organisms, which could contribute to population-level changes in abundance and persistence (Newcombe and MacDonald 1991; Bilotta and Brazier 2008). Collectively, these effects on multiple species may alter richness and diversity, ultimately leading to shifts in community composition and structure favoring species within guilds more tolerant of sedimentation and turbidity (Newcombe and MacDonald 1991; Bilotta and Brazier 2008; Larsen et al. 2011). Although these relationships have been examined extensively for fishes and some macroinvertebrates, relatively few studies have examined the individual and population-level effects of SABS on crayfish.

SABS may affect individual crayfish directly (e.g. gill fouling), indirectly (e.g. food and habitat quality) or both. The nature, response pathway, severity, and duration (short- vs long-term) of these effects are influenced by a number of biotic and abiotic factors. Biotic factors include species, condition, life history, life stage, and local community composition (Wilber and Clarke 2001; Jones et al. 2012). Abiotic factors include sediment characteristics, nature of sediment exposure (concentration, duration, frequency, and timing), turbidity, substrate characteristics, local hydrology, and presence of refugia (Jones et al. 2012). Biological responses are also influenced by interactions between sediment and other factors such as sediment-bound contaminants and organic material, temperature, pH, and dissolved oxygen (Berry et al. 2003).

Crayfish mortality due to acute or chronic exposure to elevated SABS has not been observed in laboratory experiments (Rosewarne et al. 2014; Suren, Martin, and Smith 2005). Field and laboratory studies have shown, however, that crayfish experience a range of sublethal effects related to SABS, such as impaired respiratory function (Rosewarne et al. 2014) and reduced burrowing ability (of a stream-dwelling tertiary burrower; Dyer, et al. 2015). Additional sublethal effects may be mediated through crayfish interactions with its environment: Suren et al. (2005) suggest that correlations between crayfish declines and increased SABS are likely linked to adverse effects of sediment on crayfish habitat and food resources.

Elevated concentrations of suspended sediment have been shown to cause damage to gill structures and impair respiratory function in crayfish (Rosewarne et al. 2014). However, there is no clear evidence that gill fouling by sediment causes mortality or reduces fitness, even at very high sediment concentrations (up to 20,000 NTUs) (Rosewarne et al. 2014; Suren et al. 2005). This is supported by field observations that concluded at least some crayfishes are tolerant of moderate to high concentrations of suspended solids based on the presence of those species in locations with high turbidity (Haddaway et al. 2015).

Gill fouling by sediment may act as a pathway for colonization by microbiota, some of which are parasitic (Bauer 1998). Therefore, excess sediment may increase crayfish exposure to gill parasites. Gill parasites adversely affect gill structure and function, which can induce functional hypoxia in affected individuals (Rosewarne et al. 2014). The interactive effects of exposure to excess sediment and gill parasites has been shown to compound gill damage, but did not result in significant effects on fitness or mortality (Rosewarne et al. 2014).

The documented sub-lethal, rather than lethal, effects of sediment may be due to biological and behavioral features of crayfish (Reiber 1995; Demers et al. 2006; Gäde 1984). Relative to the exposed gills of other freshwater invertebrates, crayfish gills are protected by a carapace that may shield sensitive gill structures from physical abrasion. Gill structures (setea and setobranches) positioned between each gill passively filter particles that may otherwise accumulate in gill tissue and may be especially important in environments with high suspended solid concentrations (Holdich and Reeve 1988). Molting has also been suspected as a key mechanism for reducing gill fouling (Bauer 1998; Martin et al. 2000). Active gill cleaning behavior has been documented in some species (Bauer 1998, 2008; Batang and Suzuki 2000). In addition, gill fouling by microbes, organic material, and sediment may be reduced in some crayfishes through a symbiotic gill-cleaning relationship with branchiobdellidan worms (Brown, Creed, and Dobson 2002; Lee, Kim, and Choe 2009; Skelton et al. 2016).

Crayfish burrows may serve as refuges from exposure to the direct effects of SABS. Species classified as "primary burrowers" rarely leaving their burrows or their surroundings and may never inhabit open water (Hobbs 1981); therefore, these species may avoid exposure to SABS entirely. Secondary and tertiary burrowers use burrows only during certain life stages or seasons (Hobbs 1981) and therefore may be directly exposed to SABS only while inhabiting open water. Non-burrowing crayfishes may avoid direct exposure to disturbances by excavating simple burrows (Berrill and Chenoweth 1982) or by using the burrows of other species (Johnston and Robson 2009).

Sediment may adversely affect crayfish food resources by reducing the abundance and/or quality of primary producer, macroinvertebrate, and detrital resource biomass. SABS may reduce primary producer biomass by smothering macrophytes or reducing light availability by increasing turbidity (Berry et al. 2003). As described above, sediment also has wide ranging adverse effects on macroinvertebrates that can lead to decreased diversity and abundance of prey (J. Jones et al. 2012; Bilotta and Brazier 2008). Bedded sediment may bury detritus and carrion, reducing food availability (R. Mitchell 2009). In contrast, organic matter in sediment is suspected to be a food resource for at least some crayfish species (Helms and Creed 2005).

Crayfish have long been considered opportunistic omnivores because they consume detrital, animal, and plant material (Chidester 1912). However, recent research has described variation in diet selectivity and flexibility depending on species, ontogeny, and environmental conditions (R. Mitchell 2009; Alexandra Marçal Correia 2003; Johnston, Robson, and Fairweather 2011; Grey and Jackson 2012). Because diet and trophic position of crayfishes vary (Johnston, Robson, and Fairweather 2011), some species may be more vulnerable to changes in prey availability than others, and crayfish that exhibit diet flexibility may be more tolerant to the adverse effects of SABS. The diet of some crayfish varies seasonally (Guan and Wiles 1998), and the diet of at least one species is correlated with prey abundance (Alexandra Marçal Correia 2003; Grey and Jackson 2012). This suggests that some crayfishes may be able to tolerate SABS-induced changes in the prevalence of prey through diet flexibility.

Diet composition of some crayfishes shifts from a higher proportion of animal material in early life stages to a higher proportion of detritus or plant material in adult life stages (Rabeni, Gossett, and McClendon 1995; Guan and Wiles 1998). Therefore, the research team may expect differential responses to excess SABS by life stage in some species.

Deposited sediment may reduce the abundance and quality of refugia by filling interstitial spaces (Dyer, Worthington, and Brewer 2015). Crayfish require refuges to avoid predators (Stein and Magnuson 1976; Jurcak-Detter et al. 2016), during vulnerable periods such as during and immediately after molting (Jurcak-Detter et al. 2016), and during environmental extremes, such as floods (Bubb, Thom, and Lucas 2006). (Dyer, Worthington, and Brewer 2015).

SABS may also be associated with increased organic matter deposition, which could lead to increased microbial respiration and thus reduced dissolved oxygen (S.B. Mitchell, West, and Guymer 1999). Some crayfishes breathe air and may be able to avoid low dissolved oxygen conditions (Morris and Callaghan 1998; Taylor and Wheatly 1981; Wheatly, De Souza, and Hart 1996; McMahon and Wilkes 1983). In general, stream-dwelling crayfish are considered to be more sensitive to low dissolved oxygen than burrowing crayfish species (Crandall and Buhay 2008).

Some crayfishes may be able to mitigate the adverse effects of bedded sediment through excavation or bioturbation (E. Jones, Jackson, and Grey 2016; Albertson and D. Daniels 2016; Johnson, Rice, and Reid 2011). However, the research team expects such improvement of sedimented habitat to be significant only at high crayfish densities. McLay and van den Brink (2016) hypothesized that bioturbation of bedded sediment by crayfish may induce density-dependent population control.

Excess SABS may adversely affect embryo development and survival. SABS increase mortality of fish eggs though physical abrasion and smothering (Chapman et al. 2014), but no studies have examined the direct effects of SABS on crayfish eggs. Depending on the species, female crayfish may care for incubating embryos for weeks to more than a year (McLay and van den Brink 2016; Thiel 2003). Females attach and carry embryos on the abdomen from the time of fertilization to the emergence of juveniles. During this time, females aerate and clean the embryos (Andrews 1906). After hatching, juveniles remain physically attached to the female for weeks to months and rely on her for shelter, food, and protection (McLay and van den Brink 2016; Breithaupt et al. 2015; Hazlett 1983). Therefore, the potential for direct adverse effects of SABS on embryos and juveniles may be greatly reduced by the extensive maternal care. After copulation, females may retreat to refugia during embryo development where they remain without eating and likely without leaving until the embryos hatch (Reynolds 2002). Therefore, if bedded sediment reduces the quality and availability of refugia (as discussed above), juvenile crayfish recruitment may be adversely affected.

Poor environmental conditions may induce emigration and emersion by crayfish. Some crayfishes seek air to avoid low dissolved oxygen concentrations or other environmental stressors (Morris and Callaghan 1998; Taylor and Wheatly 1981; Wheatly, De Souza, and Hart 1996; McMahon and Wilkes 1983; Grey and Jackson 2012). Downstream dispersal (i.e. drift) has been documented in crayfish, especially in relation to large flows (Englund 1999; Bernardo et al. 2011)(but see (Bubb, Thom, and Lucas 2006)); however, it is unclear if crayfish drift to avoid poor habitat conditions or if they are dislodged. Crayfish movement studies have found wide ranging variation in active movement of crayfish even among individuals of the same population (Bubb, Thom, and Lucas 2006). Crayfish movement has been found to be closely related to water temperature (Bubb, Thom, and Lucas 2006; Aquiloni, Ilhéu, and Gherardi 2005; E. Jones, Jackson, and Grey 2016); therefore, cold water conditions may constrain the ability of crayfish to actively disperse. This suggests that crayfish may be more vulnerable to disturbances in cold water environments than disturbances in warm water environments where crayfish can actively disperse to avoid poor conditions.

Elevated turbidity levels caused by greater inputs of suspended sediments may adversely affect crayfish predator-prey dynamics, intra-specific communication, and reproduction. Higher turbidity may lead to prey switching or reduced effectiveness of visual predators of crayfish. Increased turbidity was associated with a shift in diet from crayfish to piscine prey in field studies of large-mouth bass (Shoup and Lane 2015). Therefore, crayfish may experience less predation pressure in turbid environments. However, increased turbidity also could reduce crayfish foraging efficiency (Correia, Bandeira, and Anastácio 2007), although crayfish likely compensate by relying on chemical and tactile cues (Moore and Bergman 2005). In some cases, excess SABS may even increase crayfish foraging success by reducing prey refugia (Mathers, Rice, and Wood 2019). Elevated turbidity may affect communication between crayfish conspecifics. Antagonistic encounters between conspecifics last longer and are more intense in dim light than bright light (Bruski and Dunham 1987; Van der Velden et al. 2008). This suggests that elevated turbidity may increase aggressive and territorial behavior, which may lead to greater energetic demands and reduced fitness of highly territorial species (Crook, Patullo, and Macmillan 2004; Bergman and Moore 2003). Higher turbidity may also reduce the effectiveness of visual cues used in sexual selection by crayfish. Both chemical and visual cues are uniquely important for mate selection in at least one crayfish species (Aquiloni and Gherardi 2008, 2010). Therefore, increased turbidity associated with suspended sediment may alter patterns of mate selection in crayfish species. In fishes, increased turbidity has been shown to increase time spent investigating mates (Sundin, Berglund, and Rosenqvist 2010). This suggests an increased energetic cost and reduced strength of sexual selection (Järvenpää and Lindström 2004; Candolin, Salesto, and Evers 2007), which may affect long term population viability (Cally, Stuart-Fox, and Holman 2019).

In addition to direct investigations into the exposure or sensitivity of particular crayfishes to sediment, habitat guild membership is used to inform sediment sensitivity estimates. Among the species considered in this report, the research team has identified two major groups (i.e. guilds) of crayfishes that are relevant to the key stressors associated with land development (i.e. sediment and pollutants). The research team defines these guilds based on the likelihood of a crayfish species to be found in open water habitat and burrow habitat. “Open water” crayfishes include species classified as stream, lake, and tertiary burrowers by Hobbs (1981) because they are most likely to be found in above ground, open water habitat. These species are generally restricted to lotic and lentic environments, and rarely retreat to burrows, but are capable of burrowing. “Burrowing” crayfishes include primary and secondary burrowers, which are most likely to be found in subterranean habitat (Hobbs 1981). These species burrow in stream banks, riparian areas, areas where the water table is sufficiently high, wetlands, and even upland environments. Open water crayfishes are more likely to be exposed to the effects of sediment than burrowing crayfishes. Elevated levels of suspended and bedded sediments cause a wide array of adverse effects on open water crayfishes, including direct damage to tissue and reduced quality and quantity of habitat and food resources. On the other hand, the habitat of primary burrowers (and some secondary burrowers, but those species are not assessed in this report) is relatively isolated from surface water influences, so members of this guild have no practical exposure route to the effects of sedimentation that may result from construction activities.

Elevated levels of SABS degrade the physical habitat and food base that crayfish rely upon. At the individual level, SABS directly and adversely affect crayfish foraging, reproduction, growth, and survival. At the population level, these effects result in reduced abundance and persistence.

Effects of Sediment on Amphibians

At the individual level, suspended and bedded sediments have been shown to reduce survival, growth, and reproductive success of amphibians. These adverse effects may result in population-level changes in abundance and occupancy, ultimately reducing the likelihood of a population to persist over time. Collectively, these effects on multiple species may alter richness and diversity, leading to shifts in the amphibian portion of community composition and structure, favoring species more tolerant of sedimentation and turbidity.

Elevated turbidity in streams has been negatively correlated with both salamander occupancy (Pugh et al. 2015, Smith 2011) and with overall amphibian species richness (Hecnar and

M'Closkey 1996). Salamander density has also been negatively correlated with amount of suspended particulate matter present, even when accounting for forage availability (Orser and Shure 1972). Several species of stream salamanders have been shown to rely on chemical cues instead of visual cues to detect potential predators (Hickman et al. 2004, Zabierek 2014). This, coupled with relatively poor eyesight in some species, suggests that predator avoidance may not be significantly affected by increased turbidity, at least in these species. However, at least one laboratory trial demonstrated antipredator behavior was reduced in more turbid environments (Zabierek 2014). Foraging may be impaired by elevated levels of suspended sediment in predatory fish that rely on vision to detect prey, but one study found no effect of turbidity on salamander foraging efficiency (Secondi et al. 2007), suggesting perhaps some salamanders rely on chemosensory or movement sensing abilities as opposed to vision for prey location.

In addition to the above effects of suspended sediment, elevated bedded sediment may adversely affect amphibian habitat, forage, and reproduction. Several studies found greater embeddedness and smaller substrate particle sizes to be associated with reduced amphibian occurrence (Tumlison et al. 1990, Stoddard and Hayes 2005). Streams affected by increased sediment loading have been shown to support lower species richness of amphibians (Ashton et al. 2006, Corn and Bury 1989). A number of studies, using a variety of metrics, suggest that increased substrate sedimentation and embeddedness adversely affect stream amphibian abundance. In an experimental manipulation of pebble and cobble density, higher density of larger rocks led to higher salamander density for all life stages (Davic and Orr 1987). A modeling experiment showed Coastal Giant Salamander (*Dicamptodon tenebrosus*) density increased 1.06 times on average for each 10% increase in proportion of substrate that was cobble or larger (Leuthold

2010). Sedimentation from logging activities is associated with lower densities of several amphibian species (Corn and Bury 1989, Welsh and Olliver 1998). Multiple studies have shown negative associations between stream amphibian abundance and substrate embeddedness (Ashton et al. 2006, Lowe and Bolger 2002, Lowe et al. 2004, Mosely et al. 2008, Welsh and Hodgson 2008). In some cases, twice as many salamanders were found in reference streams compared to streams with more silt and less boulder (Wood and Williams 2013). Sedimentation and fine substrates have also been correlated with reduced stream amphibian abundance and biomass (Murphy et al. 1981, Corn and Bury 1989). However, the effects of bedded sediment on stream amphibian density are not universally detrimental. One stream amphibian sampling effort captured the highest density of salamanders in a stream with the highest sediment deposition rate (Pehek and Stanley 2015). Sepulveda and Lowe (2009) found embeddedness and proportion of fine substrate to be the best predictors for density of Idaho Giant Salamander (*Dicamptodon aterimmus*), a close relative of the Coastal Giant Salamander mentioned previously. A review of amphibian capture rates found that most were associated with unique habitat conditions, especially substrate, leading to difficulty in making generalizations about effects of a given stressor on multiple species (Martin and McComb 2003).

Many aquatic salamander larvae and adults feed on aquatic insect larvae and other benthic macroinvertebrates. Road construction and experimental sediment additions have been shown to reduce aquatic macroinvertebrate diversity, richness, density, and presence of sensitive taxa, and to increase drift (Hedrick et al. 2007, Matthaei et al. 2006, Larsen and Ormerod 2010). While increased drift could temporarily lead to an increase of food availability for ambush hunting aquatic amphibians, in the long term, macroinvertebrate community changes could lead to reduced food resource availability for amphibians.

Bedded sediments can also be important in determining the ability of stream amphibians to find refuge from predators and environmental stressors. Some aquatic salamanders use interstitial spaces to avoid drought by burrowing into the stream bed (Dowling 1956, Tumlison et al. 1990). In situ recapture probability has been shown to increase with sediment levels (Honeycutt et al. 2016), suggesting predators may have less difficulty locating and capturing individuals, and amphibian visibility in lab mesocosm experiments also increased with sediment levels (Leuthold 2010). These patterns are likely species- or functional group-specific as another study found no interaction found between predation and embeddedness (Lowe et al. 2004). More broadly, survival of Pacific giant salamanders has been shown to decrease with increased sedimentation (Honeycutt et al. 2016).

Much of the scientific literature, and this discussion specifically, focuses on the effects of sediment inputs on amphibians in streams, but some limited research has focused on pond and wetland habitats. Adult and larval amphibian abundance, species richness, and diversity have been shown to be reduced in ponds with higher turbidity (Brodman et al. 2003, Schmutzer et al. 2008).

Evidence is mixed on the relative sensitivity of the larval stage of amphibians to the adverse effects of sediment input, compared to adults. However, being restricted to the aquatic habitat, larvae are likely to experience much greater exposure to sediments than adults. The adult stages of many amphibian life cycles have the option of escaping into the terrestrial environment in the case of a significant aquatic disturbance. Amphibian eggs/embryos rely on the processes of water circulation for removal of wastes and respiration in the same way as fish eggs (Greig Sear Carling 2005) and are likely similarly susceptible to the adverse effects of smothering by fine sediment. While this has not yet been documented, it was suggested as the mechanism responsible for reduced salamander abundance in areas with increased silt cover and reduced proportion of boulder substrate (Wood and Williams 2013).

Elevated levels of sediment have been shown to adversely affect growth and development in amphibians. When investigating tadpoles of an Australian species of frog, Gillespie (2002) found reduction in both growth and development rates in enclosures with more sediment deposition. In another study on salamanders in Ozark ephemeral streams, metamorphosis occurred earlier in streams with greater embeddedness, while more salamanders stayed paedomorphic in streams with lower embeddedness (Bonnett and Chippendale 2006). The observed earlier metamorphosis may reflect a risk assessment decision between the sub-optimal habitat of embedded streams as a paedomorph and the inherent dangers and costs associated with the terrestrial environment as an adult (e. g. predators, dessication, greater energetic demands of movement).

In multiple field studies, occurrence, density, and abundance of larvae of several salamander species have all been negatively correlated with substrate embeddedness and positively correlated with substrate particle size (Barr and Babbit 2002, Miller et al. 2007, Stoddard and Hayes 2005, Welsh and Hodgson 2008, Welsh and Olliver 1998). Experimental manipulations of substrate composition over short time periods found greater larval salamander abundance in areas with larger particle sizes (Davic and Orr 1987, Parker 1991; but see Keitzer et al. 2013, Nowakowski and Maerz 2009). Higher proportions of fine sediments were also found to increase downstream drift of salamander larvae (Barrett et al. 2010). Larval salamander survival has been shown to be highest in experimental mesocosms filled with gravel, as opposed to combinations of finer materials (Barr and Babbit 2002). Salamander larvae are known to use interstitial spaces between large-diameter substrates for refuge (Corn and Bury 1989, Miller et al. 2007). Some larvae burrow meters below the stream bed to escape drought conditions (Miller et al. 2007). As mentioned previously, infilling of these interstitial spaces by sediment may prevent larvae from making use of these refugia.

While much of the scientific literature on stream amphibians focuses on larval and paedomorphic salamanders, sediment has also been shown to have adverse effects on the larval stage of Anurans, commonly referred to as tadpoles. Dupuis and Steventon (1999) and Gillespie (2002) reported lower tadpole densities in streams and experimental enclosures with more fine sediment and debris. Tadpole development and growth rates were also adversely affected by increased sediment deposition (Gillespie 2002). Similar to adult salamanders, tadpoles have been shown to rely on chemical cues for identification of predators (Takahara et al. 2012), suggesting that the potential adverse effects of turbidity on predator avoidance may be moderated. Suspended sediments increase turbidity and reduce light penetration through the water column, inhibiting growth of periphyton, which many tadpoles rely upon as a food source, and other primary producers (Kupferberg et al. 1994, Power 1990, Wood and Armitage 1997). While tadpoles can feed on detritus or organic sediments instead of algae, consumption of detritus alone can lead to slower development and growth, which may in turn lead to delayed metamorphosis (Gillespie 2002, Kupferberg et al. 1994).

Most research investigating the many effects of suspended and bedded sediment approach the subject as a single stressor. In real world situations, amphibians contend with a variety of combined biological, chemical, and physical stressors. A stress response induced by exposure to suspended and bedded sediment may prove lethal or may be exacerbated by multiple interacting stressors such as contaminants, elevated temperatures, or lowered dissolved oxygen. Predation responses in the presence of sediment including use of substrate refugia are addressed above. Barr and Babbit (2002) found highest densities of a salamander larva in areas with primarily boulders and little sand or bare rock, but this association disappeared when brook trout (*Salvelinus fontinalis*) were not present.

Elevated sediment inputs reduce dissolved oxygen levels by shading out primary producers and increasing microbial respiration due to deposition of organic matter commonly associated with sediment (Mitchell, West, and Guymer 1999, Wood and Armitage 1997). Aquatic larval and paedomorphic life stages of amphibians use gills or cutaneous respiration, and therefore rely on minimum levels of dissolved oxygen. Dissolved oxygen levels below a species- and life stage-specific threshold, are stressful for larval and adult amphibians and may lead to decreased hatching success, slowed growth, behavioral shifts, and decreased survival (Sacerdote and King 2009, Wassersug and Seibert 1975, Woods et al. 2010).

In contrast to the above findings that describe adverse effects of sediment on amphibians, some studies have documented either no effect of sediment or, in some cases, empirical relationships suggesting beneficial effects. One study found the highest densities of salamander larvae in areas of fine silt sediment and deciduous leaf litter (Nowakowski and Maerz 2009). Peterman (2008) found a negative association between two-lined salamander (*Eurycea bislineata*) larval abundance and sedimentation, but also found that blackbelly salamander (*Desmognathus quadramaculatus*) larvae were unaffected. Other studies similarly found no correlation between larval abundance of some salamander species (Keitzer and Goforth 2012, Lowe et al. 2004) and either substrate embeddedness or proportion of fine substrate.

Effects of Sediment on Turtles

The direct and indirect effects of SABS on turtles are generally understudied, relative to the existing literature on fish and aquatic invertebrates. For instance, the research team knows of no studies documenting SABS to cause direct mortality, inhibition of swimming performance, or elicitation of a stress response in turtles. However, elevated sediment inputs have been reported to adversely affect freshwater turtles indirectly through degradation of their habitat and effects on their prey base.

Aquatic turtles, like most fish, rely on visual detection of prey (Parmenter and Avery 1989), which can be impaired by turbidity (Zamor and Grossman 2007). However, an experiment by Grosse et al. (2010) tested the effect of increasing turbidity on the probability and time to prey capture by conditioned painted turtles (*Chrysemys picta*). They found turbidity had no significant effect on ability to capture prey. Prey capture time did increase by 25 seconds between treatments, from 2 to 40 nephelometric turbidity units (NTUs), but turbidity only accounted for 2% of the variation in capture time. While some rivers in the Southeast exceed turbidity levels of 100 NTUs, visibility of turtles in 40 NTUs was very poor, so increased levels may not further inhibit turtle foraging ability (Grosse et al. 2010). These findings suggest turtles may rely more heavily on olfactory cues to locate prey in more turbid waters and may be less susceptible to the adverse effects of increased turbidity, relative to fish.

Sedimentation of habitat has been associated with population reductions of several species: flattened musk turtles, *Sternotherus depressus;* smooth softshells, *Apalone mutica;* Illinois mud turtles, *Kinosternon flavescens;* map turtles, *Graptemys spp.*; and western pond turtles, *Actinemys marmorata* (reviewed by Bodie 2001). Flattened musk turtles, endemic to Alabama, rely heavily on the presence of rock crevices within streams for habitat, which may be clogged by fine sediment (Kenneth Dodd 1990). In contrast, bog turtles, *Glyptemys muhlenbergii,* have been reported as more abundant in areas with silty substrate (Stratmann et al. 2020).

As with other taxonomic groups, foraging generalists are likely less susceptible to the effects of SABS on their prey base than turtle species with specialized diets. Turtles that feed exclusively on mollusks may be required to move more and farther to find sufficient forage following declines of their prey due to the adverse effects of SABS (e.g. clogged gills, reduced feeding efficiency, smothering of juveniles; Brim Box and Mossa 1999, Ryan et al. 2008, Sterrett et al. 2011).

Effects of Sediment on Dragonflies

Studies have found an overall decrease in richness and density of macroinvertebrates in sites that are impacted by increased sediment loads (Buendia et al. 2013, Wantzen 2006, and Hogg and Norris 1991). Buendia et al. (2013) found that increasing sediment loads led to an increase in taxa with shorter life cycles, smaller body size, deposit feeders, and tegument respiration. Many aquatic invertebrate taxa are at least somewhat vulnerable to the effects of sedimentation, with a decreased taxa richness of scrapers, predators, gatherers and filterers observed with increasing sediment (Rabení et al. 2016). Impacts from sediment on aquatic invertebrates may be direct or indirect, such as the direct impact on filamentous gill functioning or indirect impacts on availability of foraging resources such as algae or prey (Graham 1990, Herrera 2016).

Dragonfly nymphs, as the aquatic life stage, are a predatory group (Order Odonata) that rely on visual acuity to locate prey. Sedimentation may affect dragonfly nymphs both directly and indirectly through a variety of mechanisms, many of which have yet to be explicitly investigated. Like most benthic invertebrates, increasing sediment deposition can adversely affect habitat suitability for both dragonfly nymphs and their prey. Dragonfly larvae are engulfing predators that may stalk or ambush their prey items (Tennessen 2008). Their diet spans various aquatic invertebrate groups including Chironomidae and other Diptera, Ephemeroptera, Plecoptera, and amphipods (Tennessen 2008, Burcher and Smock 2002, Folsom and Collins 1984). While there may be some preferential feeding, they will likley consume what is available, including conspecifics (Folsom and Collins 1984, Buskirk 1992). Thus, a reduction in prey abundance from a sedimentaiton event will likely adversely affect dragonfly nymphs.

Because dragonfly nymphs are visual predators, increased turbidity may impair their ability to locate prey (Wantzen 1998). In one experimental feeding study, Kefford et al. (2010) did not see a significant reduction in dragonfly feeding with sediments additions that resulted in turbidity measures between 500 and 1,000 NTU. However, these studies should be viewed with caution since they were focused on feeding rate over a short time period in a laboratory setting.

Dragonfly nymphs associated with different substrate types, ranging from sand to rocky sediment, may be differentially impacted by sedimentation events based on habitat preferences (Worthen and Horacek 2015). Studies generally found a decrease in Odonata abundance when comparing sites affected by high sediment loads versus sites upstream that did not receive high sediment inputs (Wantzen 2006, Hogg and Norris 1991).

EFFECTS OF POLLUTANTS ON BIOTA

The general term pollutant (i.e. contaminant) may be broadly applied to a range of chemical, biological, and abiotic (e.g. light, thermal) inputs that cause biological harm through various mechanisms. This discussion of aquatic pollutants will focus on those compounds associated with active construction sites and with post-construction vehicular traffic, which are transported to nearby waterways via stormwater runoff. Roadway pollutants are comprised of organic (e.g. polycyclic aromatic hydrocarbons, ‘PAH’s) and inorganic (e.g. metals) compounds derived from vehicle exhaust, vehicle wear (tires, brakes, and body frames), road surface wear, deicing salts, and herbicides (Aryal et al. 2010; Loganathan, Vigneswaran, and Kandasamy 2013). Many of these compounds are known carcinogens and have been shown to cause a variety of adverse effects to aquatic biota. With the exception of large spills, roadway pollutants found in post-construction stormwater are generally regarded as chronic rather than acute exposures.

For purposes of organism exposure, metals are often divided into two categories: essential metals that are necessary for proper biological functions in trace amounts, and non-essential metals that serve no known biological function. In both cases, environmental concentrations or body burdens above toxicity thresholds result in biological impairment, damage, or mortality. Metals commonly found in roadway pollutants include arsenic, cadmium, chromium, copper, iron, mercury, manganese, lead, and zinc (Loganathan, Vigneswaran, and Kandasamy 2013). These metals may exist in several chemical forms (i.e. oxidation states, valency states, chemical species), each with their own physical and chemical properties affecting their interaction with and toxicity to biota.

Effects of Pollutants on Fish

Fish are exposed to metals in the aqueous environment directly at their gills and indirectly, through diet, at their intestines (Poulton et al. 1995). Because most metals are hydrophilic, the direct uptake of metals in fish is generally passive and constitutes the primary route of exposure. Over time, the uptake and loss rates for inorganic metals will equilibrate at steady state tissue concentrations proportional to the ambient concentration. An exception to this is the behavior of organometallics such as methylmercury. Their exposure route is via prey organisms and their primary uptake site is in the intestines (Meyer et al. 2005). Organometallics are lipophilic and are not passively lost from fish tissues like hydrophilic metals (DeForest and Meyer 2015). Therefore, they are more likely to bioaccumulate in fatty tissues (i.e. liver and muscle) and biomagnify at higher trophic levels, resulting in predatory species with tissue concentrations disproportionate to ambient concentrations (Depew et al. 2012; Wang 2013).

Metals generally accumulate in all vital fish organs, but the highest concentrations are found in the gills (as the site of direct contact), liver, and kidneys (as metabolic and detoxification tissues). The adverse effects of metals to fish have been observed in respiratory, digestive, excretory, immune, reproductive, nervous, and endocrine systems (Javed and Usmani 2019). The presence of metals in tissues directly and indirectly increases the amounts of free radicals and reactive oxygen species, which cause oxidative stress and result in damage to membranes (mainly lipids), enzymes, and nucleic acids (Jezierska and Witeska 2001). This damage to tissues often manifests as lesions, functional impairment, structural damage, and tumor growth. Oxidative stress imposes greater energy demands and, if prolonged, leads to depletions of energy reserves and, eventually, proteins. The genotoxic effects (damaged DNA) of exposure to metals, when occurring in somatic and germ cells, may reduce fitness and survival of offspring (Javed and Usmani 2019; Jezierska and Witeska 2001).

In the colder regions of the U.S., de-icing salts are now considered a major pollutant in freshwater systems near roadways. While Georgia uses a relatively small amount of de-icing salts, they may be a significant contributor to the suite of pollutant stressors in northern parts of the state. In contrast to the long-held view that de-icing salts are almost entirely flushed from aquatic systems by winter rains, research suggests a lag in clearance time resulting in low levels of salt that persist for months and sometimes years (Findlay and Kelly 2011). Because of this, the more sensitive early life stages of fish and other biota may be exposed to higher than expected levels of salts during spawning seasons, especially if those seasons occur during summertime low-flow periods (Findlay and Kelly 2011; Hintz and Relyea 2017, 2019). Research into the effects of acute exposure to salts suggests that fish are more tolerant than other taxonomic groups, but more work is necessary to extend this generalization to chronic exposures (Hintz and Relyea 2019). Additionally, it is difficult to differentiate between the often sub-lethal effects of chronic exposure to road salts and other co-occurring roadway pollutants (Findlay and Kelly 2011).

Hydrocarbons (i.e. oil) are common organic constituents of the mixtures of pollutants deposited onto roadways by vehicle exhaust (particularly diesel), lubricating oil, tire wear, and pavement wear. Some organic compounds with known biotoxic properties are dioxins, polychlorinated biphenyls, PAHs, and phthalate esters (Loganathan, Vigneswaran, and Kandasamy 2013; Trombulak and Frissell 2000). Because of the longstanding awareness of their high toxicity and sheer volume released into the environment, most of the literature on organic roadway pollutants has centered on PAHs (Logan 2007; Shen et al. 2013), so this discussion will focus on that group of compounds. There are over one hundred PAH compounds with varying structure, functional groups, and differing physical and chemical properties that influence their interactions with other organic compounds (Cousin and Cachot 2014). As lipophilic molecules, PAHs may float at the water's surface or be associated with particles suspended in the water column, but are more often bound to bedded sediments (Logan 2007).

Uptake of most PAHs occurs by direct contact with polluted sediments, ingestion of prey that have accumulated PAHs, or incidental ingestion of polluted sediments (Vignet et al. 2014). While PAHs are generally lipophilic, some have relatively low levels of lipophilicity and fish are more likely to uptake those compounds through the gills (Logan 2007). Accumulation of PAHs are a function of environmental/dietary exposure and elimination, either passively or through active biotransformation. Most fishes are capable of metabolizing, detoxifying, and eliminating PAHs, but this varies by species and compound (Dupuy et al. 2014). Like organometallics, PAH accumulation reaches the highest concentration in fat-containing tissues such as liver, muscle, and bile (Logan 2007). Biological effects of PAHs on fish will also vary by species and compound, as well as life stage (further discussed below). Observed effects include: narcosis, lesions, tumors, physical deformities, physiological impairment, compromised immune function, and genetic damage to eggs/sperm (Larcher et al. 2014; Vignet et al. 2014).

The early life stages of fish (egg/embryo, larva, juvenile) are more sensitive to the adverse effects of pollutants than the adult stage (Barjhoux et al. 2017; Le Bihanic, Morin, et al. 2014; Le Bihanic, Clérandeau, et al. 2014; Jezierska and Witeska 2001). Exposure may also be higher, as embryos and larvae are more likely to take up pollutants such as PAHs by direct contact with bedded sediments (Jezierska, Ługowska, and Witeska 2009). Both metals and PAHs affect early life stages in many of the same ways described for adults: greater energetic demands for detoxification, physical malformations, physiological impairment (e.g. osmotic function, chemosensory, immune system), narcosis, and genotoxicity (Carreau and Pyle 2005; Logan 2007; Jezierska and Witeska 2001; Sures 2008). However, many of these effects are manifested at critical developmental stages (e.g. initial gill circulation, fin movements, and exogenous feeding). One aspect of vulnerability sometimes overlooked is an increased likelihood of incurring adverse mutagenic (including teratogenesis and carcinogenesis) effects, not because of greater inherent sensitivity, but because of greater probability of genetic exposure. In early life stages, there is a greater proportion of cells undergoing active cell division, leading to more opportunities for interaction between DNA and mutagenic compounds (Nikinmaa 2014). Ultimately, these effects result in reduced individual growth and greater mortality at all life stages, which may lead to population-level effects such as lower abundance and diversity.

Understanding of the effects of pollutants on fish is made more difficult by the many sources and mechanisms of interacting factors, both biotic and abiotic. Increasing levels of pH, hardness, and salinity reduce the bioavailability of metals due to competitive uptake of ions at the gill surface (Javed and Usmani 2019; Jezierska and Witeska 2001). Independent of other factors, metals have been shown to interact with each other in complex ways, with effects ranging from synergism to antagonism (Sauliutė and Svecevičius 2015). Levels of oxygenation and temperature interact with metals to alter their chemical form and toxicity; for example, in hypoxic conditions (chemically reducing) many metals bond with sulfides to form immobile complexes with low bioavailability (Javed and Usmani 2019). In contrast, organic pollutants are less stable in waters with higher oxygenation and persist much longer in anoxic sediments (Logan 2007). Increasing levels of dissolved and suspended organic materials, often from the breakdown of plants, interact strongly with organic pollutants and decrease their availability to biota (Logan 2007).

Many of the strongest and most common environmental interactions occur between pollutants and bedded sediments (Taylor and Owens 2009). Sediments often bind to pollutants, reducing their bioavailability (Vodyanitskii 2013; Logan 2007; Jezierska and Witeska 2001); however, transformation and biodegradation rates in anoxic environments are much lower, so sediment-associated pollutants tend to persist for much longer periods of time (Hylland 2006). Greater amounts of organic materials within sediments increase the adsorption of organic pollutants, sequestering them (Perrichon et al. 2014). The particle size of sediments influences interactions, with smaller particles having greater surface area increasing adsorption by pollutants (Jezierska and Witeska 2001; Perrichon et al. 2014). One consequence of the previously described inverse relationship between oxygenation and some metals is that when metal-polluted sediments are resuspended, toxic chemical forms of the metals become available for uptake (Javed and Usmani 2019). In this way, sediments often function firstly as a pollutant sink and secondly as a long-term secondary source of pollution exposure for biota.

Stormwater originating from upslope areas adjacent to stream crossings may deliver herbicides used to control roadside vegetation (McPherson, Moreland, and Atkins 2003). Because the mechanism of action for herbicides usually targets photosynthesis, most herbicides are not acutely toxic to fish (Solomon et al. 2013). However, chronic exposures to low levels of herbicides have not received much attention in the scientific literature. While some effects at the individual level have been documented (reproduction, stress, olfaction, and behavior), doses were either not environmentally relevant or were not linked to population-level effects (Solomon et al. 2013). Fish may be indirectly affected by herbicides via a reduction in both primary productivity and the presence of vegetation acting as cover (Relyea 2005; Diana et al. 2000). However, stormwater runoff of terrestrial herbicides is unlikely to result in concentrations sufficient to adversely affect aquatic plants (Solomon et al. 2013).

In the absence of a riparian buffer at stream crossings (Correll, Jordan, and Weller 1992), stormwater may deliver nutrients (nitrogen and phosphorus) from upslope areas directly to the stream. Effects of elevated nutrients on lotic systems have been documented extensively in the U.S. and around the world (Smith 2003, Allan 2004, Dubrovsky et al. 2010). Mechanisms by which fish may be indirectly affected include: decreased DO, shifts in leaf decomposition rates, and shifts in algal assemblages (Greenwood et al. 2006; Carpenter et al. 1998).

Thermal pollution, often in combination with other stressors, may adversely affect fishes and other aquatic biota through various pathways at multiple life stages. By removing riparian vegetation, transferring heat from terrestrial areas, and reducing sub-surface inputs, both construction-phase activities and post-construction infrastructure may elevate temperatures of nearby streams (Nelson and Palmer 2007; Herb et al. 2008). By itself, an elevation in temperature increases metabolic rate, oxygen demand, and energetic demands, thereby influencing growth rate and overall fitness (Beitinger, Bennett, and McCauley 2000; McBryan et al. 2013). It is possible that extreme temperature increases and associated deoxygenation may result in mortality (McBryan et al. 2013; I.J. Morgan, McDonald, and Wood 2001; Sand-Jensen and Pedersen 2005), especially for species living in habitats near the upper end of their thermal tolerance or in seasonally high temperatures (Beitinger, Bennett, and McCauley 2000; I.J. Morgan, McDonald, and Wood 2001). However, it is more likely that any adverse effects of thermal pollution from construction activities or stormwater will be sub-lethal and will result from complex interactions with other pollutants, such as roadway-associated pollutants (Ficke, Myrick, and Hansen 2007). Elevated temperatures have been shown to generally increase the toxicity of many common pollutants, both organic and inorganic (Ficke, Myrick, and Hansen 2007; Sokolova 2013; Patra et al. 2007; Kennedy and Ross 2012). While research is limited, early life stages of fishes are more sensitive than adults to changes in temperature (Blaxter 1992; Drost et al. 2016; Moyano et al. 2017) because of their immobility and inability to moderate the effects of such changes (Burt 2011).

The sensitivity of a given species to pollutants is likely a function of innate tolerance, elevated exposure via direct contact (e.g. benthic species), and/or foraging/trophic guild (e.g. piscivores). The research team uses guild membership to estimate pollutant sensitivity for species for which other data (such as species-level experimental studies) are lacking. Because so many pollutants are bound to sediments, the reproductive guild members whose early life stages are closely associated with sediments, resulting in high levels of exposure (Jezierska, Ługowska, and Witeska 2009), are likely to be more sensitive overall (e.g. broadcast-spawners). For the same reason, those fishes whose early life stages are not associated with sediments are likely less sensitive to pollutants: egg-attachers, gravel nest-builders, and crevice-spawners. The adult members of habitat guilds associated with sediment (e.g. benthic species) are likely to have higher levels of chronic exposure (Vignet et al. 2014) and therefore be more sensitive than members of habitat guilds not associated with sediment (e.g. pelagic; Mol and Ouboter 2004). Lipophilic pollutants (e.g. PAHs and organometallics) bind to fatty tissues, bioaccumulating in larger, longer-lived species (DeForest and Meyer 2015). These same pollutants also tend to biomagnify through trophic levels, resulting in higher body burdens and greater exposure in predatory species (e.g. piscivores; Depew et al. 2012; Wang 2013). For the same reason, drift-feeding and herbivorous species likely carry lower body burdens of pollutants and are therefore less sensitive overall.

Effects of Pollutants on Freshwater Mussels

'Pollutants' is a broad term, inclusive of chemical (e.g., metals, organic chemicals, ions), physical (e.g. trash), physiochemical (e.g., temperature, pH, dissolved oxygen), and biological (e.g., invasive species) contamination of an environment. Characterization of freshwater mussel sensitivity to all pollutants is beyond the scope of this document; therefore, for this section the research team will primarily focus on risks to mussels from chemical and physiochemical pollution associated with vehicles and road crossings or roads near streams. Polycyclic aromatic hydrocarbons, heavy metals, ions (de-icing salts), and thermal pollution (i.e., warm water runoff) are contaminants associated with roads that have received the greatest attention to date (Folkeson et al. 2016).

Freshwater bivalves, particularly North American unionid mussels, are often considered among the most sensitive organisms to chemical exposures (see Van Hassel and Farris 2007_ch 1). This conclusion has been drawn from a combination of relatively recent widespread losses of freshwater mussel diversity in a variety of habitat types as well as a growing body of results from laboratory testing. Laboratory testing has indeed confirmed that compared to other organisms, freshwater mussels are relatively sensitive to some contaminants, such as unionized ammonia, copper, cadmium, nickel, chloride, and some pesticides (Augsburger et al. 2003, Cherry et al. 2002, Wang et al. 2007, Bringolf et al. 2007, Newton and Bartsch 2007, Moore and Bringolf 2020). Several of these contaminants, including cadmium, nickel, and chloride, are commonly associated with vehicles or roadways. Furthermore, Wang et al. (2016) directly compared the acute sensitivity of juvenile mussels from a broad range of mussel species (5 species, 2 families, and 4 tribes) to 10 chemicals from different classes (ammonia, metals, major ions, and organic compounds). Their study confirmed the findings of previous studies that had demonstrated mussels are among the most sensitive taxa for ammonia, metals, and ions, but are generally not among the most sensitive organisms to organic chemicals (Wang et al. 2016). In the same study, mussels representing different families or tribes had similar sensitivity to most of the compounds, regardless of the toxic mode of action (Wang et al. 2016). This study also concluded that fatmucket (*Lampsilis siliquoidea*), one of the most commonly tested mussel species, is generally representative of other mussel species and should provide good estimates of risk to mussels as a taxon for purposes of derivation of water quality criteria. As a specific indication of the relative sensitivity of mussels compared to other aquatic biota, in 2009 the U.S. Environmental Protection Agency revised its aquatic life criteria for ammonia in response to a growing body of mussel ammonia toxicity data. With mussel data included, the acute ammonia criterion dropped from 24 mg TAN/L to 17 mg TAN/L and the chronic criterion was reduced from 4.5 mg TAN/L to 1.9 mg TAH/L (https://www.epa.gov/wqc/aquatic-life-criteria-ammonia). Freshwater mussel toxicity data are likely to drive upcoming water quality criteria revisions for nickel and chloride as well. However, mussels are not the most sensitive organisms to all contaminants, in fact, mussels consistently demonstrate relative tolerance to several common insecticides (pesticides) such as malathion, chloripyrifos, pendimethalin, and fipronil (Keller and Reussler 1997, Bringolf et al. 2007a, Bringolf et al. 2007b). Such variation in sensitivity among mussel species and compounds indicates that any notion of extreme sensitivity attributed directly to the taxa should carefully consider the uncertainty associated with any species for any given combination of chemicals and conditions (Van Hassel and Farris 2007). However, some general trends have emerged with regard to mussel sensitivity, including general effects of life stage, phylogenetic group (i.e., tribe), and different chemical classes.

There is general consensus that adult mussels are less acutely sensitive to contaminants when compared with early life stages (glochidia and early juveniles); however, adult mussels have the ability to sense toxicants in the water and they close their valves to avoid exposure (Naimo 1995, Van Hassel and Farris 2007, Cope et al. 2008). This valve closure (avoidance) response makes accurate measures of sensitivity difficult to obtain in short-term tests with adults. The duration of any avoidance behavior of adult mussels is likely to vary with respect to factors such as species, age, shell thickness and gape, properties of the toxicant, and temperature (Cope et al. 2008). However, this avoidance response is likely less problematic in chronic, low-level exposures (similar to those in nature) which could yield more accurate estimates of toxicity (Strayer et al. 2004). Relatively little information exists with regard to sensitivity to chronic contaminant exposures in adults (Cope et al. 2008) but at least one study indicated that results of chronic tests with adults may yield similar results as short-term survival tests with early life stages. Keller et al. (2007) exposed adult mussels of 2 species to copper in 28-day chronic tests and reported similar adverse effects concentrations among the adults and 9 species of glochidia and 7 species of juveniles exposed in 24-hr toxicity tests. More research is needed to better define the relative sensitivity of different life stages of mussels under various conditions, but until such information is available, standardized tests with glochidia and juveniles will likely be used for derivation of water quality criteria and other guidelines for protecting mussel populations.

As described previously in this document, the life history of freshwater mussel species includes several critical periods that can be vulnerable to stressors such as pollutants (McMahon and Bogan 2001, Waters 2007). Contaminants can disrupt discrete periods such as spawning, fertilization of ova, development of larvae (glochidia), release of glochidia from females, and attachment of glochidia to suitable host fish for metamorphosis to a free-living juvenile mussel (Cope et al. 2008). Water quality, sediment quality, health of host fish, and diet (at all stages) all have the potential to affect survival of these life stages and subsequent reproduction and recruitment to the population (Cope et al. 2008). For this section the research team describes the primary exposure routes, durations, and effects of roadway-associated contaminants for adults, larvae (glochidia), and juveniles.

Adult mussels burrowed or partially burrowed in the sediment may be exposed to contaminants via water, sediment, pore water, or diet. Few, if any, studies have evaluated dietary contaminant exposure in mussels; however, this represents a potentially substantial source of exposure. Exposure duration in water, sediment, or pore water may range from very brief (seconds to minutes) to chronic (years to decades) depending on the nature of the pollutant. Persistent organic compounds, including PAHs, can have half-lives of many years; however, other organic compounds such as some pesticides, may degrade within minutes to hours. The most persistent chemicals are insoluble in water and therefore reside in sediment, whereas less persistent compounds are typically more water soluble, thus found in overlying water rather than sediment. More soluble compounds, such as ions, are taken up across gill membranes during respiratory activity and generally do not accumulate in mussel tissues, though they may be elevated in hemolymph (blood).

Another factor that could greatly influence contaminant exposure routes and uptake by adult mussels is burrowing activity patterns (Cope et al. 2008). Historically, mussels were believed to be primarily suspension-feeders positioned at or above the sediment surface, and thus, most of their contaminant exposure was from surface water (Naimo 1995). However, mussels are reported to obtain up to 80% of their food by deposit feeding (Raikow and Hamilton 2001), siphoning in food from the sediment and pore water, and pedal-feeding directly from the sediment (Yeager et al. 1994, Vaughn and Hakenkamp 2001). Therefore, contaminants in sediment and pore water are potentially important sources of exposure during the 50-74% of the time that mussels are burrowed beneath the sediment surface (Amyot and Downing 1997, Schwalb and Pusch 2007).

Numerous studies have reported adverse effects of pollutants on adult mussels, such as survival, shell morphology, metabolic rates, enzyme activity, DNA damage, and other endpoints (for a review see: Van Hassel and Farris 2007_Ch 2). For example, survival of adults of multiple species was reduced following exposure to malathion (Keller and Russler 2007), ammonia (Cherry et al. 2005), and parathion (Kopar et al. 1993). Metabolic rates have generally been elevated in adults in response to heavy metal exposure (Cheney and Criddle 1996; Naimo et al. 1992; Rajalekshmi and Mohandas 1995) and thermal stress (Ganser et al. 2015; Haney et al. 2020). Physiological endpoints such as enzymes and glycogen have been measured in response to metals (Naimo et al. 1992) and thermal stress (Fritts et al. 2015), and generally enzyme levels increase (indicating tissue damage), whereas glycogen levels are reduced (suggesting depletion of energy reserves). DNA damage has been reported in mussels exposed to PAHs (Prochazka et al. 2012) and lawncare chemicals (Conners and Black 2004), and altered RNA transcriptional profiles were evident in mussels exposed to metals (Bertucci et al. 2017).

Little, if anything, is known about the potential for adverse effects of contaminants on mussel spawning and fertilization (Cope et al. 2008). However, at least one study (Bringolf et al. 2010) has described stimulation of release of sperm (i.e., spermatozeugamata, sperm 'spheres') upon exposure to a waterborne contaminant (fluoxetine). Viability of the sperm was not evaluated in that study. Effects of contaminants on sperm motility and viability are better described for fish (for a review see: Hatef et al. 2013). As such, the possibility that environmental contaminants might adversely affect sperm motility or viability has implications for recruitment and is an area of urgent need for further research.

Although there are some advantages to working with adults, the greater sensitivity of early life stages and practicality of working with smaller, more abundant individuals has resulted in a focus on development and use of standardized toxicity test methods for glochidia and juveniles (ASTM 2016). Availability of standardized testing protocols has resulted in a rapid increase in the number of published toxicity studies with mussel early life stages since the early 2000s. As such, data on the relative sensitivity of mussels to specific contaminants and sensitivity compared to other aquatic organisms has primarily been generated using the standardized methods with early life stages; standardized test methods do not exist for adult freshwater mussels. The standardized guidelines for early life stages utilize short-term tests (6-96 hr) with mortality (or shell closure in glochidia, which is functional death) as the primary endpoint. Longer-term tests (e.g., 28 days) have been utilized with juvenile mussels as well, and follow conventional wisdom indicating that extended exposures typically result in sublethal effects at lower concentrations than reported for acute tests (Bringolf et al. 2007; Wang et al. 2010; Wang et al. 2018).

Mussel glochidia have been reported among the most sensitive organisms tested for several ionic compounds including ammonia, chloride, potassium, sulfate, copper, nickel, and zinc (Augspurger et al. 2003; Milliam et al. 2005; Keller and Reussler 2006; Wang et al. 2007; Bringolf et al. 2007; Gillis et al. 2011; Wang et al. 2017). However, mussel glochidia also appear to be relatively insensitive to some organic pollutants including certain PAHs and pesticides (Bringolf et al. 2007; Wang et al. 2017). Keller et al. (1998) tested the toxicity of diesel fuel contaminated sediments on glochidia of *Lampsilis siliquoidea* and *Lasmigona costata* and juvenile *Villosa villosa*, with ambiguous results. Weinstein (2000) tested glochidia of *Utterbackia imbecillis* to characterize the acute toxicity of photo-activated fluoranthene (a PAH) and found that mussels were less sensitive than other aquatic species but that the presence of low UV intensities increased the sensitivity of glochidia by 45X times. In 2001, Weinstein and Polk (2001) repeated this experiment with anthracene and pyrene (also PAHs) and reported similar results with photoactivation. However, little is known about the toxicity of PAHs found at relatively low levels in streams with little urbanization in their watersheds. Research is needed in this area to better understand the chronic risks of PAH exposure to mussels.

Glochidia may be exposed to pollutants via water while they are brooding in the marsupial gill of the adult female, or in water following release from the marsupial gill but prior to encystment on the host fish. Glochidia exposure in the marsupial gill may be weeks (short-term brooders) to many months (long-term brooders). Glochidia packaged in conglutinates may be protected from at least some chemicals in the water such as copper (Gillis et al. 2008); however, glochidia brooding in the marsupial gill membrane have been demonstrated to be susceptible to at least some water-soluble chemicals, such as perfluorooctanesulfonate (PFOS; Hazelton et al. 2012) and fluoxetine (Hazelton et al. 2013). Effects in these studies included reduced viability upon release from the marsupium and reduction in viability over time, as well as reduced ability to attach and metamorphose on a host fish. Other dissolved chemicals, such as metals, can cross gill membranes as well and may affect brooding glochidia. Currently only one study (Jacobsen et al. 1997) has evaluated the effects of a metal (copper) on brooding glochidia. Although Jacobson et al. (1997) did not report any changes in glochidia viability immediately following release from the marsupial gill, duration of viability (as reported affected by Hazelton et al. 2012, 2013) was not evaluated by Jacobson et al. (1997). Additional studies are needed to better characterize the risks of metals and other dissolved compounds to brooding glochidia.

Exposure of free glochidia (i.e., released from the marsupium into the water) is dependent on host attraction strategy and can likely last from hours to weeks. Luring species (such as those in Tribe Lampsilini) and those that release glochidia in conglutinate packets likely have the lowest exposure duration in water. With these strategies, glochidia are released into the water from the marsupium or conglutinate when a host fish strikes the lure or conglutinate packet. In these species, glochidia would attach to the fish's gills within seconds; though the encapsulation process on gills takes anywhere from 2 – 36 hours (Ingersoll et al. 2007). Glochidia from species that release glochidia in mucous nets, such as those in the genus *Elliptio,* may be exposed to contaminants in water for several days as well (Cope et al. 2008). Regardless of host attraction strategy, the duration for encapsulation is similar, suggesting that 24 hour toxicity tests are an appropriate duration for glochidia standardized test methods (Fritts et al. 2014).

General consensus is that encysted glochidia are afforded a measure of protection from waterborne contaminants (Jacobsen et al. 1997, Rach et al. 2006). However, another potential route of exposure for glochidia to contaminants is during the encystment period on the host fish, which typically lasts a few weeks to months (Cope et al. 2008). The effects of contaminant body burdens of host fish on glochidia metamorphosis success is largely unexplored, though as true parasites (Fritts et al. 2013), glochidia consume host (gill) tissues so potential exists for transfer of contaminants from host tissues to glochidia.

For a given chemical, sensitivity of juvenile mussels appears to be similar to that of glochidia when comparing acute toxicity tests; however, longer (sub)chronic juvenile tests (~28 days) demonstrate that extended exposures with juveniles result in sublethal effects at lower concentrations (Bringolf et al. 2007; Wang et al. 2010; Wang et al. 2017; Wang et al. 2018). Similar to adults, juvenile mussels are exposed to contaminants via sediment, pore water, diet, and surface water. Juveniles typically become infaunal benthic organisms that typically remain burrowed beneath the sediment surface for 2 to 4 years (Strayer et al. 2004). This infaunal existence may strongly influence contaminant exposure routes and subsequent uptake during the first few years of life. Residence beneath the surface necessitates deposit feeding because juveniles have poorly developed gills that are incapable of filter feeding (Yeager et al. 1994). During this time, juveniles ingest algae, bacteria, and other organic matter for nutrition (Waters 2007) via pedal feeding, a type of deposit feeding by which juveniles utilize their foot to bring food particles into their shell for intake. The relative importance of exposure to juvenile mussels to contaminants in overlying surface water, pore water, sediment, or food has not been adequately assessed (Cope et al. 2008).

As with the other taxonomic groups covered in this report, some life history traits are likely to alter the sensitivity or exposure of mussel species to roadway-associated pollutants. The glochidia of long-term brooders have longer period of potential exposure to dissolved contaminants that may cross the gill membrane. The glochidia of luring mussels (lampsilines) and conglutinate-producing mussels, have much shorter period of exposure to the ambient water and any dissolved compounds (e.g. metals). Likewise, the glochidia of mussels that produce mucus webs, are likely to have a longer period of exposure to these compounds. While these life history traits may serve to inform the discussion of exposure routes, their utility as a means to estimate overall sensitivity to pollutants is likely limited due to the overwhelming influence of the primary exposure route for early life stages: deposit-feeding by juveniles. This exposure route is common to all freshwater mussel species and does not allow for much differentiation of pollutant tolerance among reproductive guilds.

Effects of Pollutants on Crayfish

Many crayfish, like other aquatic taxonomic groups, are chronically exposed to mixtures of anthropogenic compounds including metals, hydrocarbons (e.g. PAHs and PCBs), pesticides and herbicides, excess nutrients, and road salts. Field and laboratory experiments have demonstrated lethal and sublethal effects of pollutants on crayfish (Klobučar et al. 2012, Breithaupt et al. 2015, Pavel Kozak 2016); however, most of these studies evaluated the effects of pollutants on hardy, invasive species such as the Red Swamp Crayfish (*Procambarus clarkii*) and other species of commercial interest. Fewer studies have examined pollutant sensitivity of rare species. Even so, pollutants have been identified as the second most important driver of declines of North American crayfishes (Richman et al. 2015).

At the individual level, pollutants may reduce survival, growth, and reproductive success of freshwater organisms, which could contribute to population-level changes in abundance and persistence. Collectively, these effects on multiple species may alter richness and diversity, ultimately leading to shifts in community composition and structure favoring tolerant species. Pollutants may affect individual crayfish directly (e.g. reduced chemosensory), indirectly (e.g. food quality) or both. The nature, response pathway, severity, and duration (short- vs long-term) of these effects are influenced by biotic and abiotic factors. Biotic factors include species, life history, and life stage (e.g. Holdich 2002a, Hubschman 1967, Allert et al. 2011). Abiotic factors include pollutant characteristics, nature of pollutant exposure (concentration, duration, frequency, and timing; Steele et al. 1992, Anderson et al. 1997), and substrate characteristics (e.g. Logan 2007, Kunz et al. 2005). Biological responses are also influenced by interactions between pollutants and other factors, such dissolved oxygen (Javed and Usmani 2019).

Metals readily accumulate in the skeleton and tissue of crayfish adults, juveniles, and eggs and are retained for long periods of time (Guner 2010, Kouba et al. 2010). Crayfish may be exposed to metals directly, such as through the gills during respiration (Anderson et al. 1997) or indirectly, such as through their diet (Allinson et al. 2000). Because most metals are hydrophilic, direct uptake of metals by crayfish is generally passive and likely the primary exposure pathway.

Over time, metal uptake and loss rates of inorganic metal tissue concentrations will equilibrate proportionately to ambient inorganic metal concentrations. However, crayfishes may also bioaccumulate metals and organometallics above ambient concentrations (Alikhan et al. 1990, Anderson et al. 1997, Aluma et al. 2017). Crayfish may biomagnify organometallics by consuming contaminated prey items (Parkst et al. 1988); however, biomagnification is probably limited relative to other taxonomic groups due to the lower trophic position of most crayfishes (Schmitt et al. 2011).

Exposure to metals may increase rates of crayfish mortality. In a field experiment, caged crayfish experienced significantly higher mortality and reduced growth downstream of a lead mining operation, relative to reference sites (Allert et al. 2008). Effects of laboratory exposure to copper varied by dose: high doses caused adult and juvenile mortality, and low doses caused juvenile mortality and reduced adult growth (Hubschman 1967).

Metal exposure also causes deleterious sublethal effects on crayfish, including impaired gill function (Torreblanca et al. 1987, Meyer et al. 1991, Alexopoulos et al. 2003, Ward et al. 2006), gill damage (Torreblanca et al. 1987), impaired hepatopancreatic function and structural damage (Reddy and Fingerman 1994, Anderson et al. 1997, Bollinger et al. 1997), increased likelihood of infection (Ward et al. 2006), impaired chemosensory function (Sherba et al. 2000, Lahman et al. 2015), and altered locomotion and behavior (Wigginton and Birge 2007). Exposure to metals also impairs regeneration capacity, among other deleterious effects (e.g. inhibited limb regeneration and reduced growth) in crustaceans (e.g. shrimp and crabs) (Weis et al. 1992).

In some cases, crayfish may readily accumulate metals without adverse effects (Woolson et al. 1976, Besser and Rabeni 1987, Kouba et al. 2010). Relative to species of other taxonomic groups (two fish, *Cottus bairdii* and *Oncorhynchus mykiss*; one amphipod *Hyalella azteca*) a crayfish species (*Orconectes hylas)* was found to be tolerant of acute exposure to a mixture of metals (Kunz et al. 2005). However, the same study found the crayfish to be more sensitive than the two fish species, but less sensitive than the amphipod, to a chronic exposure to those metals (Kunz et al. 2005). Crayfish employ coping mechanisms in response to metal exposure. A mucus gill sheath may reduce uptake of metals by the gills (Anderson 1978, Anderson and Brower 1978b); however, excessive gill mucus may reduce gill function and lead to asphyxiation (Alexopoulos et al. 2003). Molting of the exoskeleton reduces the total body burden of accumulated metals (Knowlton et al. 1983, Simon and Garnier-Laplace 2005). Although the toxic effects of metals could be mediated by molt frequency, crayfish are especially sensitive to the toxic effects of metals during molting (Wigginton and Birge 2007, Allert et al. 2009, Allert et al. 2011). Crayfish regulate internal concentrations of essential metals but are unable to regulate nonessential metals (Roldan and Shivers 1987, Chambers 1995, Anderson et al. 1997). This mechanism may be limited when exposed to essential and nonessential metals simultaneously (Meyer et al. 1991). However, Steele et al. (1992) found no effect of a metal blend on crayfish behavior or survival.

Crayfish life-stage, sex, and body size interact with metal sensitivity and exposure. Juvenile crayfish are more sensitive to metal exposure than adult crayfish (Wigginton and Birge 2007). Juvenile crayfish are generally thought to eat a higher proportion of animal material and feed more frequently than adults (Gutiérrez-Yurrita et al. 1998; but see Larson et al. 2016, Stites et al. 2017), which could result in high rates of metal exposure relative to adults (Farkas et al. 2003).

In metal-contaminated, low food quality environments, exposure may increase due to increased feeding frequency (Lin et al. 2004). Heit and Fingerman (1977) found that male crayfish (*Faxonella clypeata*) are more sensitive to mercury than females. Tunca et al. (2013) found that males contained significantly higher concentrations of metals relative to females collected from the same waterbody; however, Bagatto and Alikhan (1987) found no effect of sex on metal accumulation. Higher surface area to volume ratio, higher metabolic rate, and increased permeability (due to more frequent molts) may contribute to higher metal concentrations in younger stream crayfish (Knowlton et al. 1983, Allert et al. 2008). There is inconsistent evidence for the relationship between metals and crayfish size; Anderson and Brower (1978a), Dickson et al. (1979), and Mirenda (1986a, b) found no relationship between the size of crayfish and the accumulation of metals, whereas other studies (Knowlton et al. 1983, Bennet-Chambers and Knott 2002, Allert et al. 2011) found a negative relationship between crayfish size and metal concentrations. Metal exposure may stunt crayfish size (Allert et al. 2008), and because crayfish body size is positively correlated with clutch size (Larson and Magoulick 2008), metal may reduce crayfish fecundity (Allert et al. 2011). Thus, if a crayfish population is exposed to metals, and metals reduce crayfish size, a population's growth rate may also be stunted.

Although no studies have examined the effect of deicers on crayfish, some studies have examined the effects of saltwater_intrusion into freshwater crayfish habitat. Salinity has been shown to reduce crayfish adult survival (Anson and Rouse 1994), juvenile survival and hatching rate (Holdich et al. 1997), foraging (Lung et al. 2012), locomotion (Mills and Geddes 1980), and growth and molt frequency (Mills and Geddes 1980, but see Hesni et al. 2009). Some crayfishes alter osmoregulation to cope with increased salinity (Holdich et al. 1997, Green et al. 2011). In marine environments, increased salinity synergistically increases the toxicity of nutrients to crustaceans (Jiann-Chu and Chi-Yuan 1991); however, the research team is not aware of research that has demonstrated this interaction in freshwater crayfish.

Hydrocarbons are a diverse group of compounds, and crayfish sensitivity likely varies depending on species, compound, compound interactions, and concentration (Dickson et al. 1982, Achazi et al. 1998), Holdich 2002a). Body burden of hydrocarbons in crayfish often reflects ambient conditions (Jewell et al. 1997, Levengood and Schaeffer 2011, Paulik et al. 2016). Adverse effects of hydrocarbons on invertebrates include embryo-toxicity (Bellas et al. 2008), gonadal lesions (Schäfer and Köhler 2009), narcosis (Mäenpää et al. 2009), altered protein regulation (Zheng et al. 2008), and reduced chemosensory function (Blumer 1973, Atema and Stein 1974, Pearson and Olla 1980, Gauthier 2012, Jensen et al. 2012). Exposure to sediment and triphenyl phosphate caused crayfish to drift in a laboratory setting (Fairchild et al. 1987). In hydrocarbon-contaminated, low food-quality environments, exposure may increase because "in order to maximize net energy gain, benthic invertebrates adjust their feeding rate based on available food/energy resources in sediment" (Lin et al. 2004). Like other taxonomic groups, crayfish and other invertebrates may metabolize and inactivate hydrocarbons (Gewurtz et al. 2000); however, this ability is greater is some other groups, like fish (Livingstone 1998).

Effects of herbicides/pesticides on crayfish include mortality (Jarboe and Romaire 1995), delayed and reduced growth (Frontera et al. 2011, Velisek et al. 2019), gill damage (Benli et al. 2016), reduced nerve and muscle function (Chalmers et al. 1986), reduced hepatopancreatic function (Stara et al. 2019), reduced chemosensory function (Wolf and Moore 2002, Cook and

Moore 2008, Browne and Moore 2014), genetic damage (Costa et al. 2018), reduced anti-predator behavior (Wolf and Moore 2002, Sohn et al. 2018), and increased physiological stress responses (Benli et al. 2016). Pesticides can accumulate in the tissue of a crayfish (*Cambarus bartonii*) and persist for multiple years (Dimond et al. 1968). Crayfish sensitivity to herbicides/pesticides varies by compound (Eversole et al. 1996, Paul and Simonin 2006, Barbee et al. 2010). Some pesticides may be metabolized by crayfish (James 1994, Escartín and Porte 1996, Nordone et al. 1998). Naqvi et al. (1987) found that adult crayfish sensitivity to the least toxic pesticide used in their experiment was 500,000 times lower than the most toxic pesticide in their experiment. Paul and Simonin (2006) found that crayfish were 1-2 orders of magnitude more sensitive to permethrin than fish. However, McLeese (1976) found that crayfish were much more tolerant to fenitrothion than Atlantic salmon (*Salmo salar*). Crayfish assimilated a molluscicide at a lower rate than soft-bodied invertebrates (Sanders 1977), which suggests crayfish may be more resistant to molluscicides than other stream macroinvertebrates. Crayfish drift behavior is positively correlated with exposure to pesticides/insecticides (Poirier 1987).

Sensitivity to herbicides/pesticides interacts with crayfish sex, exposure history, life-stage, and body size. Previously exposed female crayfish experienced significantly more DNA oxidation than male crayfish in a study of one pesticide (penoxsulam) (Costa et al. 2018). Other studies found no evidence for an interaction between crayfish sex and herbicide/pesticide exposure or sensitivity (Sohn et al. 2018). Exposure history may influence sensitivity of subsequent herbicide/pesticide exposure. Individuals from a previously exposed populations may be more susceptible to herbicides/pesticides than a naive population (Costa et al. 2018, Paul and Simonin 2006). Sensitivity and exposure to herbicides/pesticides varies by crayfish life-stage and body size. Acute pesticide sensitivity was highest among young-of-the-year crayfish, intermediate among juveniles, and lowest among adults (Naqvi et al. 1987, Buřič et al. 2013). Smaller crayfish are significantly more sensitive to herbicides/pesticides (Paul and Simonin 2006).

Excess nutrients (ammonia, nitrite, and nitrate) cause increased mortality, reduced growth, and increased molt frequency in decapod crustaceans including crayfish (reviewed in Romano and Zeng 2013). Long term exposure to nutrients may increase crayfish susceptibility to infection (Yildiz and Benli 2004), although in some cases crayfish may be able to recover from temporary effects of acute exposure (Meade and Watts 1995, Yildiz and Benli 2004). Chronic nutrient pollution could result in eutrophication in receiving waterbodies. This may lead to community shifts and reduced water quality, including hypoxia. Although crayfish are generally considered tolerant of hypoxic conditions (Reynolds et al. 2013), hypoxic conditions exclude a crayfish from a lake systems in New Zealand (Kusabs et al. 2015). One laboratory study found no effect of the interaction between hypoxia and excess nitrite on a crayfish (Broughton et al. 2018).

The severity of the above effects of pollutants are often modified by interactions with biotic and abiotic factors such as temperature, environmental chemistry, presence/type of substrate, diet, and preferred habitat. Increased temperature increases the acute toxicity of cadmium, copper, zinc, and to juvenile crayfish (Khan et al. 2006). Similarly, pyrethroid insecticides are more toxic to terrestrial insects and fish at lower temperatures (Bradbury and Coats 1989). Like in fish, interactions between metals may result in antagonistic or synergistic effects on crayfish (Guner 2010). Crayfish sensitivity to pollutants varies by species (Gherardi et al. 2002, Wigginton and Birge 2007). For example, a widespread, invasive crayfish (*Procambarus clarkii*) is considered very tolerant to environmental stress including extreme temperature, moisture, salinity, low dissolved oxygen, and pollutants but is unlikely to be a good proxy for native, narrowly distributed species (Gherardi et al. 2002). Metals, hydrocarbons, herbicides/pesticides, and other toxins accumulate in and bind to sediment and benthic organic material (Paul and Meyer 2001, Fletcher et al. 2019). Therefore, species that occupy habitats with fine sediment/soils and organic material, such as burrows and depositional areas in streams and lakes, may experience greater exposure to accumulated pollutants than cobble- and riffle-dwelling species. For the same reason, historical pollution may disproportionately affect species in close contact with sediment. Metal concentrations in soil pore water downstream of a former mining sites were significantly higher than surface water concentrations (Brumbaugh et al. 2002, Allert et al. 2008, Allert et al. 2013), and soil pore water metal concentrations were negatively correlated with the occurrence of a burrowing species (Allert et al. 2011).

In addition to direct evidence, membership in crayfish habitat guilds helps to inform estimates of pollutant sensitivity for the species covered in this report. Limited evidence suggests open water crayfishes (stream, lake, and tertiary burrowers; Hobbs 1981) may be more sensitive to pollutants than burrowing crayfishes (Simon and Morris 2009). Therefore, the research team estimates that sensitivity is similar among crayfish in both the burrowing and open water groups. In contrast, the exposure to pollutants associated with roads is likely to vary greatly by habitat guild. Many roadway pollutants bind to and accumulate in benthic sediments and may be transported by groundwater into surrounding hydric riparian soils (Hylland 2006). This process effectively converts riparian soil from a pollutant sink into a secondary source of exposure for crayfishes in close contact with the soil. As a result, primary burrowers (in contact with water table, see (Horwitz and Richardson 1986) are exposed to pollutants through direct contact with contaminated burrow water and soil as well as through their diet of underground invertebrates, plant material, detritus, and the soil itself. Therefore, primary burrowers likely experience substantially greater exposure to pollutants, compared to open water crayfishes. For example, the sediment in riparian soil may contain higher pollutant concentrations than surface water, such as in areas receiving chronic, low concentration pollution and areas affected by legacy pollution sources (Brumbaugh et al. 2002).

Exposure of crayfish to pollutants in real world scenarios are more accurately represented by chronic exposure to low concentrations of complex pollutant mixtures, which may be reflected in studies investigating the relationship between crayfish occurrence and urbanization. Although few studies have examined the effect of landscape characteristics on the occurrence of crayfishes, the occurrence of rare crayfishes is often correlated with forest cover (Burskey and Simon 2010, Larsen et al. 2011, Frisch et al. 2016). Westhoff et al. (2006) acknowledged the challenges of modeling the occurrence of a rare crayfish at the landscape scale.

Exposure of crayfish to pollutants may be lethal, but more often causes a wide range of sublethal effects. At the individual level, pollutants impair crayfish physiological function, reduce growth and lifespan, inhibit locomotion, and reduce reproductive success. At the population level, the cumulative effects of chronic exposure to pollutants result in reduced abundance and persistence.

Effects of Pollutants on Amphibians

Amphibians are generally regarded as a taxonomic group that is particularly sensitive to environmental pollutants (Alix et al. 2014; Babini et al. 2016; Croteau et al. 2008; DeGarady and Halbrook 2006; Diaz et al. 2020; Dorchin and Shanas 2010; Dupler et al. 2019; Ferreira et al. 2004; Herkovits et al. 1998; Kincaid, Floyd, and Unger 2018; Russell, Gillan, and Haffner 1997; Wake and Vredenburg 2008). A suite of characteristics are cited as the reasons for amphibians' sensitivity and their utility as biological indicators: semi-permeable skin and eggs, cutaneous respiration and gill use, movements between aquatic and terrestrial habitats, and ontogenic shifts in food sources (Babini et al. 2016; Croteau et al. 2008; DeGarady and Halbrook 2006; Diaz et al. 2020; Dorchin and Shanas 2010; Dupler et al. 2019; Ferreira et al. 2004; Kincaid, Floyd, and Unger 2018). In contrast, in a review of the amphibian toxicology literature, Kerby et al. (2010) found that while amphibians are generally more sensitive to phenols than other taxa, they are not more sensitive to environmental pollutants in general. Like many other taxonomic groups, individual species can have vastly disparate sensitivities (Kerby et al. 2010). For a single toxicant, amphibian species in the same family can have 5-10 fold differences in LC50 (Gratwicke 2008). In a meta-analysis, Egea-Serrano et al. (2012) found that pollutants as a whole most often cause moderate adverse effects on survival and condition of amphibians, as well as large increases in frequency of developmental abnormalities.

A variety of metals have been found in tissues of frogs collected from streams with watersheds dominated by agricultural and/or urbanization (Bank et al. 2007; Loumbourdis and Wray 1998). Some metals from automobiles (e.g. cadmium, copper, lead, and zinc) have been shown to have sublethal to lethal effects on amphibians (Croteau et al. 2008). Lead concentrations in green frog

(*Rana clamitans*) and bullfrog (*Lithobates catesbeianus*) populations along highways have been correlated with daily traffic volume and sediment concentrations (Birdsall et al. 1986). Sediment metal concentrations have been shown to affect species richness and occurrence of individual amphibian species in constructed urban wetlands (Simon et al. 2009). Tolerance to metal contamination varies greatly by species (Arrieta et al. 2004; Ferreira et al. 2004). For example, wood frog (*Lithobates sylvaticus)* survival through metamorphosis has been negatively related to metal levels in stormwater pond sediments, but this relationship was not detected in sympatric American toads (Gallagher et al. 2014). Cadmium was reported to delay metamorphosis and inhibit gonad development in Iberian ribbed newts (*Pleurodeles waltl*) (Flament et al. 2003). Some metals bioaccumulate in amphibians, leading to higher concentrations in tissues than in the environment (Bank et al. 2007; Herkovits et al. 1998).

Amphibians may also be adversely affected by PCBs present in water or bound to sediments. The effects of this pollutant are likewise dependent on both species and life stage. PCBs may alter sex ratios and increase the prevalence of intersex condition in frogs (Jofre and Karasov 2008; Reeder et al. 1998). PCBs have been shown to affect gut microbial communities in frogs (Kohl et al. 2005). In contrast, Pezdirc et al. (2011) found concentrations of PCBs in the tissues of the cave salamander, *Proteus anguinus anguinus,* up to three times the levels found in cave sediments, but observed no adverse effects. Other studies have similarly failed to document effects of PCB exposure on amphibian abundance, species richness, habitat occupancy, growth, survival, physiology, and behavior (DeGarady and Halbrook 2003; Jung 1997; Gibbs, Rouhani, and Shams 2017). PCBs may have more apparent and severe effects on early life stages in amphibians, discussed in the below section on early life stages.

Salts from deicing compounds spread on roads are another source of chemical contamination to road-adjacent streams and wetlands. Amphibian tolerance thresholds to salinity are mostly unknown (Adamus 2001), but acute toxicity thresholds and effects on survival are dependent on species (Collins and Russell 2009; Karraker and Ruthig 2009). Wood frog survival through metamorphosis has been negatively related to salt levels of stormwater pond sediments while American toads (*Anaxyrus americanus*) seem to be unaffected (Gallagher et al. 2014). Salt concentrations in roadside ponds influence amphibian community structure, excluding less tolerant species such as spotted salamanders (*Ambystoma maculatum*) and wood frogs (Collins and Russell 2009). Road salts can also have adverse effects on the immune systems of amphibians, leading to higher parasite loads in individuals exposed to higher concentrations (Milotic, Milotic, and Koprivnikar 2017).

Pesticides have been shown to alter, often with adverse effects, amphibian growth, development, immune function, community structure, behavior, and survival (Bridges 1999; Brodman et al. 2010; Buck et al. 2015; Forson and Storfer 2006; Relyea 2004; Rollins-Smith et al. 2011). These effects may be exacerbated with exposure to multiple varieties of pesticide simultaneously (Relyea 2004). Impairment of immune function constitutes an especially harmful effect when coupled with exposure to pathogens such as ranaviruses and chytrid fungal infections (Green, Converse, and Schrader 2002; Pereira et al. 2013).

Thermal pollution of aquatic habitat by runoff from impervious surfaces at road crossings constitutes another significant pollutant, one that may interact with other pollutants or stressors

(Gandhi and Cecala 2016). Ectothermic organisms like amphibians may be adversely affected by alterations to their thermal environment. Two-lined salamander (*Eurycea bislineata*) larvae hatched in urban streams grew larger than individuals hatched in reference streams (Barrett et al. 2010). The authors attributed this difference to higher water temperatures and reduced intraspecific competition resulting from lower survival (Barrett et al. 2010).

The empirical effects of urbanization on amphibians may be used as a proxy for exposure to a real-world composition of various chemical compounds at relevant concentrations. Reviews on amphibian responses to urbanization found generally negative relationships between urbanization and amphibian richness, presence, and abundance, with altered community structure (Hamer and McDonnell 2008, Scheffers and Paszkowski 2012). Urbanization has been associated with reduced amphibian species richness (Barrett and Guyer 2008; Calderon et al. 2019; Canessa and Parris 2013; Hamer and Parris 2011) and reduced richness of native amphibians (Riley et al. 2005). Several studies found a negative relationship between urbanization and amphibian occupancy in streams and wetlands (Canessa and Parris 2013; Clark et al. 2008; Dietrich 2012; Johnson et al. 2011; Smallbone, Luck, and Wassens 2011). Salamander survival (Price et al. 2011; Price et al. 2012), abundance (Calderon et al. 2019; Price, Browne, and Dorcas 2012; Price et al. 2006) and population densities (Bowles et al. 2006; Orser and Shure 1972) have also been shown to be inversely related to urbanization. In one study, stream salamander abundance four years after initiation of development had declined by 60% to 90% compared to pre-urbanization levels (Price, Browne, and Dorcas 2012). These responses are often species- and context-specific. As a counter-example, Lane and Burgin (2008) found higher species richness in urban sites. While overall species richness declined with urbanization, Hamer and Parris (2011) found individual species' responses were more variable.

Many studies focus on empirical relationships and fail to elucidate the mechanisms underlying these effects, investigate vital rates, or describe long term population effects (Barrett and Price 2014; Scheffers and Paszkowski 2012); however, some work has explored these relationships more fully. Axolotl (*Ambystoma mexicanum*) consumption rates were reduced and diet compositions shifted in response to the presence of contaminants in urban runoff (Chaparro-Herrera et al. 2013; but see Barrett et al. 2012). Indirect effects of urbanization on amphibians may occur via the composition of their prey base. Riley et al. (2005) found lower diversity of macroinvertebrate communities in urban streams, with dominance by species tolerant of water quality degradation. Because many larval and adult salamanders rely on macroinvertebrates as a primary food source, such changes in macroinvertebrate communities may adversely affect their diet and ultimately their growth and reproductive rates. Some studies have shown mixed results of urbanization on amphibian stress responses; Mondelli (2016) found short-term stress hormone release was unaffected by urbanization, but chronic stress responses were dampened in individuals from urban populations. In contrast, Gabor et al. (2018) found no effect in a similar study.

Increased watershed imperviousness, another metric that may be used as a proxy for the effects of roadway-associated pollutants, has been shown to affect amphibian site occupancy, community diversity, and abundance. In general, imperviousness is negatively related to amphibian site occupancy (Simon et al. 2009, Trumbo et al. 2012, Guderyahn et al. 2016; but see

Alix et al. 2013). Amphibian species richness has been negatively correlated with percent impervious land cover adjacent to wetland habitat (Michael 2014; Simon et al. 2009). Some of these effects may be scale dependent; Eakin (2018) found positive relationships between impervious cover and amphibian population sizes at large scales, but the opposite at smaller scales. Other studies found no effect of imperviousness on amphibian abundance in general (Rizzo et al. 2016; Weaver 2012; Weaver and Barrett 2017) or survival of salamanders specifically (Price et al. 2011).

Though the life cycles of many amphibians include a terrestrial adult stage, the early life stages of most amphibians are strictly aquatic. Being restricted to an aquatic habitat means eggs and larvae are more likely to be exposed to waterborne compounds. Because of the differences in food sources, respiratory strategies, and habitat requirements of these early life stages, their susceptibility to environmental contaminants differs from that of adults. Contaminants accumulated in energy-rich tissues during the larval stages may be mobilized during metamorphosis (Bleiler et al. 2004). Concentrations at which contaminants are lethal to amphibian eggs and larvae are highly specific to individual contaminants and taxa. Argentine toad (*Rhinella arenarum*) embryos are susceptible to copper toxicity at much lower concentrations than Pb, Ni, Hg, Al, or Cd (Herkovits and Helguero 1998). Bullfrog tadpoles are relatively tolerant to copper compared to larvae of other amphibian species (Ferreira et al. 2004). Ferreira et al. (2004) found that at high doses, bullfrog tadpole tissue copper concentrations declined over time. Accelerated egestion, known to be an effective control in other aquatic organisms, is hypothesized as the mechanism for reduction of copper concentrations (Ferreira et al. 2004). Cadmium, lead, and zinc are toxic to anuran embryos and larvae at varying concentrations (Herkovits et al. 1998; Herkovits and Perez-Coll 1997).

Sublethal effects of contaminants on amphibian early life stages include reduced survival, increased frequency of morphological abnormalities, and altered growth and development rates (Chen et al. 2006; Dorchin and Shanas 2010; Eakin 2018; James and Little 2003; Nixdorf, Taylor, and Isaacson 1997). Tadpoles of multiple anuran species chronically exposed to cadmium exhibit increased growth and reduced time to metamorphosis (Gross, Chen, and Karasov 2007; Gross et al. 2009; James and Little 2003). Green tree frog (*Dryophytes cinereus*) tadpoles exposed to environmentally relevant concentrations of aluminum exhibited reduced body size and swimming performance (Jung 1997). Lead exposure lead to developmental deformities, reduced motility, and delayed metamorphosis in tadpoles (Chen et al. 2006). Dorchin and Shanas (2010) found road runoff adversely affects the growth and development of European green toad (*Bufo viridis*) embryos and larvae. Road salts reduced tadpole motility and survival, disrupted sex ratios, and increased frequency of developmental deformities (Donoel et al. 2010; Karraker and Ruthig 2009; Lambert et al. 2017). DeGarady and Halbrook (2006) found that juvenile anurans accumulated PCBs in higher concentrations than adults. While DeGarady and Halbrook did not observe adverse effects, toxicity and developmental deformities resulting from PCB exposure have been documented in other studies (Rosenshield, Jofre, and Karasov 2009; Sun-Kun and Soung-Yung 2004). Jung (1997) showed reduced length of green frogs to be the only effect of long-term laboratory exposure of eggs to PCBs; there was no observed impact on physiology, movement, morphological development, or hatching success. In field studies, lower survival and hatching success were documented at sites with higher sediment PCB levels

(Jung 1997; Karasov et al. 2009). Rosenshield, Jofre, and Karasov (2009) found PCB exposure reduced swimming speed, growth, survival, and hatching success.

Increasing levels of urbanization have been shown to reduce larval stream salamander survival and abundance (Price, Browne, and Dorcas 2012; Price et al. 2011), though effects on individual organisms are mixed. In one study, stream salamander larval abundances were nearly halved following a period of urbanization in the watershed (Price, Browne, and Dorcas 2012). Increasing imperviousness was also a useful predictor of reduced larval salamander density and abundance (Barrett, Helms, Guyer, et al. 2010; Miller, Hess, and Moorman 2007), though there are exceptions to its utility (Price et al. 2011). For example, Eakin (2018) found urbanization to be positively related to tadpole size, development rate, and survival; with imperviousness positively related to body condition and tail dimensions.

Aside from direct toxicity and developmental effects on amphibians, pollutants often interact with other stressors to further impact amphibian populations. Relyea and Diecks (2008) found that tadpoles in mesocosms were not directly affected by chronic pesticide treatments, but the pesticides did reduce zooplankton populations. This effect released phytoplankton from predation, leading to algal blooms, which subsequently reduced dissolved oxygen and pH and shaded out periphyton, tadpoles' main food source. While tadpoles can feed on detritus or organic sediments instead of periphyton, consumption of detritus alone can lead to slower development and growth (Gillespie 2002, Kupferberg et al. 1994), which may in turn lead to delayed metamorphosis (Kupferberg et al. 1994). This trophic cascade induced by low concentration, chronic contaminant input had 25 times the impact of a higher concentration single treatment (Relyea and Diecks 2008). A similar chronic pesticide exposure mesocosm experiment carried out by Relyea and Hoverman (2008) resulted in increased predation on amphibian larvae by Odonate nymphs, without affecting survival of the Odonates.

Impervious surfaces can increase runoff volume and adversely affect riparian and aquatic vegetation cover (Canessa and Parris 2013). Because stream frogs rely on vegetation for shelter in the larval stage and for egg attachment as breeding adults, adverse impacts on vegetation could directly impact stream amphibian survival and reproduction (Canessa and Parris 2013). Canessa and Parris (2013) found a positive relationship between proportion riparian vegetation and stream amphibian species richness.

Effects of Pollutants on Turtles

As with other taxonomic groups, freshwater turtles may be exposed to a variety of construction- and roadway-associated pollutants. Direct and indirect exposure to these pollutants may cause adverse biological effects at the individual level (e.g. reduced growth, lower survival), but may also cause adverse effects at the population level (e.g. higher mortality, reduced persistence). The primary pollutants investigated in the literature are metals and organic pollutants, such as PCBs and PAHs.

Turtles may be exposed to PCBs through a variety of routes: ingestion of contaminated prey, directly from water, maternal transfer, or from egg incubation in contaminated soil. PCBs have been shown to reduce bone mass and impair immune function in turtles (Holliday and Holliday 2012, Yu et al. 2012, Ming-ch'eng Adams et al. 2016). Juveniles collected in areas with high concentrations of PCBs and other organic pollutants had slower growth and higher rates of deformities, some of which caused mortality (Bishop et al. 1998, de Solla et al. 2008). The environment-dependent sex determination (e.g. temperature) of most turtle species constitutes an additional response pathway lacking in many other taxonomic groups. Estrogen-like compounds such as PCBs have been shown to cause feminization or intersex condition in hatchlings, leading to skewed population sex ratios and altered sexual selection processes. A female with smaller body size induced by PCB exposure will produce smaller hatchlings, which may reduce fecundity because smaller hatchlings have lower survival rates (Janzen 1993; Holliday and Holliday 2012).

The other primary class of organic pollutants, PAHs, have also been shown to adversely affect eggs and hatchlings. Eggs exposed to PAHs and crude oil had lower embryo survival and higher deformity rates (Van Meter et al. 2006). Specific deformities included malformed carapacial scutes, a lack of pigmentation, missing digits, blood collection around the brain, and organ herniation (Van Meter et al. 2006). Research on this subject is relatively scarce due to a limited understanding of PAH metabolism in turtles, lack of reliable biomarkers, and turtles' protected status.

Metals such as lead, cadmium, and mercury have been shown to bioaccumulate in turtles and may be passed maternally to eggs (Yu et al. 2011, Hopkins et al. 2013). Methylated mercury is a more toxic form and may be passed from gravid females to their eggs (Hopkins et al. 2013). The biological effects of exposure to metals are similar to those of other organic pollutants: deformities, increased mortality, and altered behavior (Yu et al. 2011, Hopkins et al. 2013).

Châteauvert et al. (2015) found a positive relationship between mercury body burden and trophic level, as well as animal size.

The scientific literature and the above discussion of aquatic pollutants focuses on dietary exposure to organic compounds, which bind to the fatty tissues of organisms and allow them to bioaccumulate and biomagnify through trophic levels. While freshwater turtles primarily breathe air, many (if not most) possess the ability for aquatic respiration through one of three surfaces: skin, buccal/pharyngeal cavity, and cloacal cavity (Bagatto and Henry 2000, FitzGibbon and Franklin 2010). There has been little research done to investigate the degree to which aquatic respiration serves as a direct exposure pathway for hydrophilic metals or other compounds. It is likely that these pollutants affect turtles in many of the same ways as other organisms (e.g. fishes) discussed in previous sections (Yu et al. 2011).

Effects of Pollutants on Dragonflies

Odonate taxa (dragonflies and damselflies) responses to urbanization and associated pollutants often vary, though they generally respond negatively (Villalobos-Jiménez et al. 2016). Larval Odonate (i.e. nymphal) abundance, specifically in members of the Families Gomphidae and Libellulidae, has been shown to be lower in sites subject to mining and urbanization influences (Moreno et al. 2009). Girgin et al. (2010), however, found that species in these families had a higher tolerance to heavy metals (e.g. manganese and nickel) in urban streams than those in others. This same study also found one species of dragonfly (Family Gomphidae) to be sensitive to cadmium, boron, and iron pollution. When compared to other aquatic macroinvertebrates, dragonfly nymphs often have higher tolerances to heavy metals and mid-chlorinated PCB congeners (e.g. PCB-153 and PCB-138), as they can sequester these chemicals in their exoskeletons (Yu et al. 2013; Tollett et al. 2009). However, dragonfly nymphs exhibit more sensitivity to copper exposure than either cadmium or lead in toxicity assays (Tollett et al. 2009).

Species assemblages of dragonflies were especially poor (or absent) in stream reaches subject to diffuse, long-term urban and agricultural pollution (Ferreras-Romero et al. 2009). Likewise, semivoltine dragonflies (i.e. species that take two years to complete their life cycle) were found to dominate aquatic macroinvertebrate assemblages in reaches not influenced by urbanization, while uni- and bivoltine species (those species that have one to two generations per year) were more common in developed catchments (Ferreras-Romero et al. 2009). When compared to populations in natural ponds, dragonfly nymphs inhabiting highway stormwater runoff ponds showed a higher degree of DNA damage, which was also significantly correlated with PAH and Zinc concentrations in the sediment (Meland et al. 2019).

Burrowing dragonfly larvae have direct contact with sediments contaminated with heavy metals and can act as a conduit through which mercury biomagnification occurs through aquatic food webs (Haro et al. 2013; Goutner and Furness 1997). Though shedding their nymphal exoskeleton through metamorphosis provides some reduction in the body burden of heavy metals and organic compounds, adults can both retain and continue to accumulate substantial concentrations of pollutants in their tissues as they feed (Buckland-Nicks et al. 2014; Tweedy et al. 2013).

EFFECTS OF NOISE ON BIOTA

The increasing presence of anthropogenic noise in aquatic environments is well documented. Sources of this noise span a broad range: daily vehicular traffic, shipping, sonar, seismic airguns, construction activities, and others. Elevated levels of background noise and acute sounds may affect aquatic biota at the individual and population levels in a number of ways: emigration from suitable habitat, reduced foraging/breeding success, inhibition of migration, and impaired intra-species communication (e.g., related to reproduction, territoriality). Research into the effects of noise on aquatic biota primarily has been focused on marine systems, with much of the work devoted to marine mammals. Research on freshwater fish is limited and no studies have examined population-level effects (Popper and Hawkins 2019).

While daily traffic across stream-crossing structures produces low levels of continuous sound, the discussion of noise in aquatic systems will focus on noise from common construction activities in freshwaters, especially the sounds generated by pile driving and demolition. Rather than continuous sounds, pile driving and demolition activities produce impulsive sounds that are generally short and more intense (high peak sound pressure and short duration, <1 s). These impulsive sounds propagate through water directly or indirectly (after reflecting off borders such as substrate or the water surface) as well as through the substrate material itself. Impulsive sounds may be measured as particle motion (either particle velocity m/s or acceleration m/s2), the instantaneous peak sound pressure, or as the sound exposure level over time (Pa^2/s), with repeated impulsive sounds measured as cumulative sound exposure over time. Other important features of aquatic sounds that may influence biological responses are the rise time and frequency spectrum. The characteristics of sounds produced by pile driving are highly variable, depending on substrate, hammer size/type, pile diameter/material, etc., with most energy below 500 Hz but up to 1000 Hz (Dahl, de Jong, and Popper 2015; Reyff 2012).

Effects of Noise on Fish

Fish detect sounds in multiple ways and rely on that information for essential biological functions, including: the location and selection of mates, detection of predators and prey, and in territorial communication with conspecifics (Hawkins and Popper 2018; Sand and Bleckmann 2008). All fish detect sounds by changes in particle motion, but some also use changes in sound pressure (Nedelec et al. 2016; Popper and Hawkins 2018). Evidence of the importance of hearing is the fact that more than 800 fish species across more than 100 families are known to produce sound (Bass and Ladich 2008). Most fish detect sounds across a relatively wide range of frequencies (<50 Hz to ~400 Hz and up to 4000 Hz; Ladich and Fay 2013). The presence of a swim bladder is considered a major factor in the ability to detect sounds as well as the sensitivity to intense sounds. The swim bladder acts as a transformer and amplifier of sounds, increasing stimulation of the otoliths (ear bones). Auditory acuity depends in part on the structure and location of the swim bladder in relation to the ear (Popper et al. 2003). Some fish have sophisticated features directly linking the swim bladder to the inner ear, such as the Weberian apparatus/ossicles in otophysan fishes, which include minnows, suckers, and catfishes (Popper and Fay 2011).

Daily traffic across stream-crossing structures or construction activities may produce levels of continuous noise sufficient to mask the sounds used by fish to locate prey, detect predators, or find/identify mates (Purser and Radford 2011; Luczkovich and Keusenkothen 2008; de Jong et al. 2018; Holt and Johnston 2015). Some fish (e.g. *Cyprinella venusta*) have been observed to alter their own sound-production behavior to compensate for noisy background conditions (Holt and Johnston 2014).

Exposure of fish to repeated high energy impulsive sounds (e.g., pile driving, demolition) may result in damage to tissues and/or mortality (Caltrans 2001; Casper et al. 2016). Specific organs damaged by rapid oscillation of the swim bladder walls, following exposure to high energy sounds, include the brain, liver, kidney, gonads and the swim bladder itself (Hastings and Popper 2005; Halvorsen et al. 2011). If fish are located in close proximity to the sources of high energy impulsive sounds at a construction site, it is possible that physical damage or mortality may result. However, this has only rarely been demonstrated in a lab or field setting (Caltrans 2001) and it is more likely that the effects on fish and other biota will be temporary or behavioral in nature. Temporary hearing loss (i.e. temporary threshold shift) may result from exposure to intense sounds for a short duration or less intense sounds for longer duration. While experiencing hearing loss, fish may be at risk due to poor communication and impaired predator/prey interactions. Unsurprisingly, some species are more sensitive to sound and are more prone to temporary hearing loss, depending on the duration and intensity of exposure (Smith and Monroe 2016). In the many investigations examining sound exposure, hearing returned after a period of minutes to days in all cases (Smith and Monroe 2016). Documented behavioral responses to impulsive sounds include: startle response, feeding disruption, area avoidance, elevated ventilation rates, altered swimming speed, bottom-diving, and altered schooling behavior. Stress responses have also been studied (Mickle and Higgs 2018); however, research in this area is limited and clear conclusions as to biological relevance have been difficult to develop (Popper and Hawkins 2019). Additionally, fish may show reduced or absent responses to intermittent impulsive sounds over extended periods, either through habituation or temporary hearing loss (Radford et al. 2016).

Because of sound attenuation in water, the high energy sound waves likely to result in tissue damage are generally limited to a small area near the source (Dahl, de Jong, and Popper 2015). As mobile animals, fish are likely to emigrate from areas with high levels of intense noise and are therefore generally less susceptible to many of the potential adverse effects of exposure to impulsive sounds (Dahl, de Jong, and Popper 2015; Popper and Hastings 2009). However, some species may be biologically motivated (e.g. spawning migrations) to remain in or continue through areas with high noise levels.

Very little research, all of it focused on marine species, has examined the effects of intense sound on fish eggs/embryos or larvae (Popper et al. 2014; Govoni et al. 2008; Bolle et al. 2016). The nature of exposure for early life stages, relative to adults, may differ in important ways. Because of the reduced mobility of larvae and eggs/embryos, early fish life stages are potentially exposed to impulsive sounds for longer durations than the mobile adult stage. The eggs/embryos of many species are in direct contact with stream substrate, which may act as a medium for the transverse sound waves resulting from pile driving or demolition activities. In those species possessing swim bladders, that organ may develop during the larval stage making them susceptible to the effects of impulsive sounds. In spite of the potential influence of the above factors, the overall sensitivity of early life stages of fish to impulsive sounds has been found to be comparable to that of the adult life stage (Debusschere et al. 2014; Popper et al. 2014; Andersson et al. 2017).

In spite of the generally sparse data describing the effects of impulsive sounds on freshwater fish, some broad trends are discernable and are useful for estimating overall sensitivity. There is a strong correlation between sensitivity to impulsive sounds and the presence of a swim bladder, as well as its location and degree of connectedness to the ear (Halvorsen et al. 2011; Carlson 2012). Popper et al. (2014) used this correlation to group adult fish and early life stages into sensitivity categories when developing interim guidelines for exposure to impulsive sounds resulting from pile-driving and demolition/explosive activities (Table 7.3 and Table 7.2, respectively, in Popper et al. 2014). These guidelines have been widely adopted, and the research team uses this as the basis for classification. However, the research team simplified Popper's classification into just two groups: (1) those species with special structures mechanically linking the swim bladder to the ear, which are considered to have high sensitivity to sound, and (2) all other taxa, which are considered to have medium sensitivity to sound. The high sensitivity group includes minnows, suckers, catfishes, and shad.

Effects of Noise on Mussels

While some work has been done to investigate the effects of anthropogenic sound on invertebrates, the research team knows of no studies that directly examine the potential impacts of anthropogenic sounds on freshwater mussels (Popper and Hawkins 2015). However, Amyot and Downing (1997) postulated that the turbulence, noise, or vibrations made by personal watercraft caused 10% of an Eastern Elliptio (*Elliptio complanata*) population to bury themselves into the sediment during periods of high activity. Similarly, underwater vibrations from noise have been reported to cause valve closure in marine mussels (*Mytilus edulis*; Roberts et al. 2015). Additionally, marine mussels exposed to pile driving noise have been reported to demonstrate higher clearance rates, suggesting increased metabolism due to stress (Spiga et al. 2016). Based on the responses in marine mussels, freshwater mussels are also likely to be adversely affected by noise, such as increased stress along with reduced filter feeding, reproduction, and locomotion. Potential impacts of noise on known host fishes must also be considered and are explored in depth elsewhere in this report (Appendix C). Sensitivity of host fish to noise should be heavily considered in decision-making, as avoidance of specific areas due to anthropogenic noise by host fishes could significantly impact reproduction in freshwater mussels.

Effects of Noise on Crayfish

Although crustaceans have no known air filled cavities nor pressure-sensitive organs (Patullo and Macmillan 2015), several studies found that at least some marine and freshwater decapod crustaceans can perceive and produce sound and vibrations (Favaro et al. 2011). The purposes of these sounds are likely to deter predators, alert conspecifics of predation danger, for mating behaviors, and competition (Favaro et al. 2011). Chemical and visual cues are considered to be the primary modes of communication between freshwater crayfish and their surroundings (Bouwma and Hazlett 2001, Moore 2007, Aquiloni et al. 2008). However, acoustic cues are suspected to bolster communication between crayfish in some cases, such as in dark or turbid environments or in air (Favaro et al. 2011, Buscaino et al. 2012). At least some freshwater crayfishes may use morphological features to produce sounds (Sandeman and Wilkens 1982, Favaro et al. 2011).

Anthropogenic noise can disrupt communication and alter the behavior of marine crustaceans. For example, increased anthropogenic noise has been associated with reduced feeding and reduced predator avoidance behavior in crabs (Wale et al. 2013). Less is known about the effects of anthropogenic noise on freshwater decapods. In a laboratory experiment, a crayfish (*Procambarus actutus*) exposed to anthropogenic noise displayed more aggressive behavior and reduced overall activity, compared to the control group (Hopson 2019). In a second experiment, Hopson (2019) found that anthropogenic noise reduced all activity including aggressive behavior. Based on evidence from studies of marine decapods, decreased activity could increase mortality due to higher predation and reduced feeding. The research team knows of no evidence suggesting crayfish can acclimate to increased anthropogenic noise (Hopson 2019).

Vibrations associated with impulsive sounds caused by anthropogenic activities (e.g. pile-driving, demolition) may have adverse effects on benthic organisms (Roberts and Elliott 2017). Crabs detect and are sensitive to vibrations up to 300 meters away, which results in increased alertness and altered locomotion. The research team knows of no studies that have investigated the effects of impulsive sounds on freshwater crayfish.

Effects of Noise on Amphibians

Amphibians use sound production and perception for communication, orientation, and defense (Jakob Christensen-Dalsgaard and Elliott 2008; Diego-Rasilla and Luengo 2004; Natale et al. 2011; Brodie 1978). The most commonly recognized form of amphibian sound production is anuran mating calls. However, salamanders also use calls for communication with potential mates and other conspecifics (Crovo, Zeyl, and Johnston 2016; Wyman and Thrall 1972).

While most amphibians have the ability to sense sounds underwater, in the air, and in the ground (Smotherman and Narins 2004), the majority of studies on effects of anthropogenic noise on amphibians have been focused on the terrestrial life stages of anurans. Adult frogs have been shown to respond to increasing noise levels by reducing activity (Lukanov, Simeonovska-Nikolova, and Tzankov 2014), but most research examines the effects of traffic noise interference with mating calls. The Lombard effect describes a common strategy of animals attempting to make themselves heard over background noise by increasing amplitude of vocalizations. In addition to increasing amplitude, amphibians can increase call rate, pitch, and duration in response to increased background noise (Parris, Velik-Lord, and North 2009; Penna, Pottstock, and Velasquez 2005).

While terrestrial calls are more familiar, underwater hearing and underwater mating calls are used by many frog species (Jakob Christensen-Dalsgaard and Elliott 2008; Simmons et al. 2014). *Xenopus laevis*, a commonly studied aquatic frog species, can localize calls underwater and can be conditioned to react to underwater sounds (J Christensen-Dalsgaard, Breithaupt, and Elepfandt 1990; Vedurmudi, Christensen-Dalsgaard, and Van Hemmen 2018). Bullfrogs are more sensitive to low frequency sounds underwater than in the air (Lombard, Fay, and Werner 1981). Adult salamanders also rely on underwater sounds for communication. Smallmouth salamanders produce underwater calls when held in tanks in a lab (Coleman 2016). California newts (*Taricha torosa*) make multiple sounds for communication on land and in the water (Davis and Brattstrom 1975). When kept in groups, the entirely aquatic two-toed amphiuma (*Amphiuma means*) produces clicking sounds but does not vocalize when kept individually (Crovo, Zeyl, and Johnston 2016).

Much of the research on amphibian sound communication focuses on adults, but hearing and sound production are also important to early life stages. Background noise can also affect the larval stage of anurans in their aquatic habitats. Tadpoles reduce consumption rates and increase activity levels in response to traffic noise (Castaneda et al. 2020). Bullfrog tadpoles have similar auditory sensitivity to adults, except without a high frequency peak (Weiss, Stuart, and Strother 1973).

High volume sound (e.g. impulsive sound from pile-driving) can also directly damage amphibians hearing abilities (Fehrenbach 2015; Simmons et al. 2014). Exposure to unusually high pressure level sounds can damage amphibian hair cells, leading to a reduction in ability to hear the entire range of frequencies normally perceptible (Fehrenbach 2015; Simmons et al. 2014). This mechanism is poorly studied, but a handful of experiments have demonstrated that anurans and salamanders have the ability to regenerate damaged hair cells within a time frame of between a few days and a month (Detwiler 1998; Fehrenbach 2015; Simmons et al. 2014; Taylor and Forge 2005). This ability to regenerate damaged organs could serve to reduce amphibians' sensitivity to impulsive sounds.

Effects of Noise on Turtles

Almost all of the very limited research into the effects of noise on turtles has focused on marine species (Popper et al. 2014), with only Christensen-Dalsgaard et al. (2012) examining a freshwater species (red-ear slider, *Trachemys scripta elegans*). Most studies examined the hearing ability of turtles rather than the behavioral or physical effects of sound. However, some studies did report behavioral effects (e.g. emigration) of impulsive sounds on marine turtles (Gitschlag and Herczeg 1994, Weir 2007).

As with fish, elevated levels of traffic noise may interfere (i.e. "masking") with the detection and directionality of sounds that are biologically important to turtles (Popper et al. 2014). Impulsive sounds may cause freshwater turtles to avoid or emigrate from areas near construction activities. These behavioral effects may result in reduced foraging efficiency, altered mate selection, or reduced predator avoidance. It is possible that impulsive sounds may cause stress, loss of hearing, or physical harm to turtles, but this has not been documented. Because turtles are mobile animals and can avoid areas with such activities, if they do experience any effects, they are more likely to be behavioral in nature.

Effects of Noise on Dragonflies

While the research team knows of no studies on the effects of noise on dragonflies. Based on the response of other organisms, as reviewed above, exposure of dragonflies to construction-related impulsive sounds may result in behavioral effects such as startle-response, altered activity, and altered predator avoidance. Exposure may also cause physical changes including stress, reduced growth, or tissue damage that may result in mortality. However, this is entirely speculative, and in the absence of any literature the research team considers dragonflies to be moderately sensitive to sound.

EFFECTS OF ALTERED HYDROLOGY ON BIOTA

Hydrologic conditions may be altered by some construction activities, particularly the use of jetties, temporary platforms and coffer dams. The majority of the scientific literature related to altered hydrology focuses on either permanent, large-scale alterations (e.g. dams, culverts) or increased flashiness of flow regimes due to increasing urbanization. The effects of temporary structures on flows are less studied (but see draft report by Bledsoe et al. 2019).

In-stream structures, whether permanent or temporary, may redirect or constrict the flow of water. Redirected flow may scour and destabilize banks, increasing erosion of bank sediment and elevating downstream levels of suspended and bedded sediment. Constriction of flow increases water velocity, which may in turn lead to mobilization of existing bedded sediments (i.e. scour), degradation of local habitat, and inhibition of upstream movement (e.g. spawning migration).

The required permits for in-water work issued by the USACE (Savannah District, Regional Permits 30, 31, 32, 33, 34, and 35) stipulate that no more than 33% of the wetted channel width may be blocked by temporary work structures. In the absence of a detailed analysis of a particular structure's hydraulic effects, the research team considers this limitation on channel blockage, coupled with the temporary nature of such structures and their effects, to be sufficient to minimize adverse effects of altered hydrology to the species covered by this report.

EFFECTS OF PHYSICAL CONTACT ON BIOTA

Some construction activities have the potential for equipment or construction materials to come in direct physical contact with individuals of a protected species or their habitat. In the case of an individual, physical contact may result in injury or death. In the case of habitat, contact may result in temporary or permanent degradation or loss of habitat.

Construction equipment performing in-water work may strike or crush individual organisms. The risk of this depends on the type of equipment (e.g. barges, excavators, pile-drivers, earthmovers), its method of installation/operation, and the sensitivity of the species (see below). Temporary in-water use of equipment may also indirectly affect a species by harming macroinvertebrates, smothering/removing macrophytes, or altering stream structure. The physical structure of the stream may be permanently altered, to the detriment of the protected species, by compacting stream bed materials, changing stream flow patterns, or exposing stream banks to elevated erosion that may deliver sediment to the stream for long periods of time.

Similarly, construction materials (e.g. concrete, fill, riprap, demolition debris) or structures may come into physical contact with individuals of a species, possibly resulting in injury or death. Materials that are removed from a work site pose a risk to individuals and habitat both at the time of introduction and at the time of removal, but their effects to habitat are temporary. Materials permanently placed into the stream environment (e.g. concrete/fill used in culvert installation or channel modification) constitute an ongoing loss of habitat. Permanent materials or structures may also alter local hydrology, which may in turn affect habitat.

The early life stages of many taxonomic groups are most at risk of direct physical contact with construction equipment/materials, as they have reduced ability for movement and avoidance of these effects. The most sensitive taxonomic groups are those with no capacity for perception or movement away from such activities (e.g. freshwater mussels).

Fish and their habitat may be directly or indirectly affected in many of the same ways described above. Adult fish have relatively low risk to physical contact because of their ability to perceive and avoid in-water construction activities. Early life stages of fish (eggs/embryos/larvae) have higher risk due to their lower ability to perceive or avoid threats.

Because mussels are practically incapable of perception or avoidance of the relevant physical threats, they are at the greatest risk of striking or smothering by construction equipment and materials, especially those individuals engaged in siphon-feeding or lure displays. However, the direct effects of physical contact on a population are often reduced by relocation of individuals prior to in-stream activities. Additionally, mussel reproduction may be reduced via the indirect effects of physical contact on host fish.

Crayfish and their habitat may be directly or indirectly affected in many of the same ways as fish. The primary difference is the attachment of early life stages (incubating embryos and young juveniles) to the adult females. Adult females (and males) have relatively low risk to physical contact because of their ability to perceive and avoid in-water construction activities (relative to the early life stages of fish). Because of the attachment to adult females, developing embryos and young juveniles have similarly reduced exposure to physical contact. In contrast, while independent juveniles can perceive and avoid in-water activities, they are less mobile than adults and may still be adversely affected by direct physical contact.

Amphibians, turtles, and their respective habitats may be directly or indirectly affected in many of the same ways described above. Adult amphibians and turtles have relatively low risk to physical contact because of their ability to perceive and avoid in-water construction activities (especially those amphibian species with terrestrial adult life stages). Early life stages of amphibians (eggs/embryos/larvae) have higher risk due to their inability to perceive or avoid threats. Because the turtle species covered in this report lay their eggs on terrestrial sites, there is little risk posed by physical contact with in-water construction equipment or materials.

Dragonflies (aquatic larvae) are at relatively high risk of physical contact because of their low ability to perceive and avoid in-water construction activities.

CHAPTER 7. CONCLUSION

The research team has developed a flexible system, the Total Effect Score (TES), for assessing the impact of road construction projects on imperiled freshwater species that accounts for project characteristics, site characteristics, and species sensitivity. The system gives GDOT greatly increased flexibility to design and construct projects in the way that is most cost efficient, without sacrificing protection for imperiled species. The system is implemented with a spreadsheet tool that is designed to be user friendly and provide outputs that dovetail with existing systems.

The research team has also provided the blueprint for a programmatic agreement (PA) for road construction projects that affect freshwater systems in Georgia and may impact aquatic species protected under the Endangered Species Act. The programmatic agreement is intended to cover both informal and formal consultation under a single streamlined system, which should substantially reduce consultation time and increase predictability. The actual programmatic agreement and biological opinion will need to be written by USFWS with input and concurrence from GDOT, FHWA, and the US Army Corps of Engineers. However, the research team has supplied a template for the PA itself, a full biological assessment to support the biological opinion, and the TES system and tool to operationalize the PA. One piece that is not included is a method for calculating take of species listed under the Endangered Species Act. This will also need to be developed by USFWS, but the information supplied in this document should make this a straightforward task.

The information provided in this report represents the best scientific data available at this time and is an appropriate basis for the PA. However, several components of the PA and the TES should be reviewed on a periodic basis to incorporate new research. This is particularly true for the species sensitivities to sedimentation and contaminants: for many of the covered species, there have been no dedicated studies of sensitivity up to this point, so the research team has inferred sensitivity based on traits and studies of related species. Future research will likely cause many of these sensitivities to be revised. Fortunately, updating these values in the TES tool is a straightforward task. Other elements of the TES that should be reviewed periodically (at least once every 5 years) are the efficiencies for erosion & sedimentation BMPs and stormwater management BMPs. The research team believes that the core of the TES system itself is robust and with periodic updates will provide a reliable basis for the PA for the foreseeable future.

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