Biological Opinion

FWS Log #04ET1000-2017-F-0212

Office of Surface Mining Reclamation and Enforcement approval of Surface Mining at Kopper Glo Mining, LLC, Cooper Ridge Surface Mine, OSMRE Permit Number 3270, Claborne County, Tennessee

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Executive Summary

A Biological Opinion (BO) is the document required under the Endangered Species Act (ESA) that states the opinion of the U. S. Fish and Wildlife Service (Service) as to whether a proposed federal action is likely to jeopardize the continued existence of listed species or result in the destruction or adverse modification of designated critical habitat (DCH). This BO addresses the effects to the blackside dace (*Chrosomus* [= *Phoxinus*] *cumberlandensis*) and Indiana bat (*Myotis sodalis*) resulting from surface coal mining as authorized by Kopper Glo Mining, LLC, Surface Mining at Cooper Ridge Surface Mine (the action), Claiborne County, Tennessee under OSMRE Permit 3270.

Section 9 of the ESA and regulations issued under section 4(d) of the ESA prohibit the taking of endangered and threatened species, respectively, without special exemption. Federal agencies may obtain such exemption through the Incidental Take Statement of a BO that supports a non-jeopardy finding for their proposed actions.

"To jeopardize the continued existence of a listed species" 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 a listed species in the wild by reducing the reproduction, numbers, or distribution of the species (50 CFR §402.02). This BO examines whether the proposed surface mining and reclamation operation, implemented in a manner that is consistent with the commitments included in the SMCRA permit number 3270 is likely to jeopardize the continued existence of the blackside dace or Indiana bat.

Based upon recent Service guidance regarding application of the northern long-eared bat (*Myotis septentrionalis*) 4(d) Rule to Federal actions that may affect the species, we determined that incidental take of the northern long-eared bat as a result of the proposed action is not prohibited because tree removal activities that would occur as a result of the action would not: (1) remove any tree or trees within 150 feet of a known occupied maternity roost tree from June 1 through July 31 (2) remove any trees within 0.25 mile of a northern long-eared bat hibernaculum at any time of year, or (3) have any impacts to hibernacula. Although we have relied upon the findings of the northern long-eared bat programmatic biological opinion for the final 4(d) rule (U. S. Fish and Wildlife Service 2015a, b) to fulfill our project-specific section 7 responsibilities for this species, a discussion of the impacts (and beneficial effects) of this permitting action to the northern long-eared bat is included in order to be comprehensive with our effects analysis. **The Service has determined that the proposed action may affect northern long-eared bats using the permit boundary and on areas adjacent to the permit boundary as summer roosting areas. However, the northern long-eared bat 4(d) rule exempts incidental take for activities that fit the three criteria listed above. Therefore, there is no incidental take authorized for northern long-eared bat as a result of this BO.**

Athough valid bat surveys indicate that no Indiana bat hibernacula or summer roosts are known on or near the permit boundary, OSMRE has prudently assumed presence of Indiana bat on or adjacent to the permit boundary and has included protective measures in the permit that would prohibit any potential lethal take. Therefore, in an abundance of caution, t**he Service has determined that the proposed action could result in incidental take of up to 180 Indiana**

bats bats potentially using a 7,476-acre area (including the 1,496 acres within the permit boundary and an additional 5,980 acre area adjacent to and surrounding the mine permit boundary) for summer roosting areas. The Indiana bat is the only species covered under this formal consultation for which critical habitat has been federally designated. However, there is no federally designated critical habitat (DCH) in the vicinity of the proposed action.

Permanent (persistent) blackside dace populations are not known to occur in streams affected by the proposed action. However, OSMRE prudently assumed presence of blackside dace, as a consideration of the potential for elevated conductivity to persist in affected stream reaches and therefore to affect blackside dace migrating through stream reaches not typically occupied, or blackside dace attempting to colonize presently unoccupied streams on the permit boundary.

The action being considered in this current BO is anticipated to have overlapping effects in several stream reaches considered as “travel corridors” in a recent consultation with OSMRE (U. S. Fish and Wildlife Service 2016a). In that previous consultation, blackside dace incidental take related to surface mining and stream water quality was expected to result from activities associated with OSMRE’s issuance of a permit for the Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296). The Service has determined that the incremental effects of the Cooper Ridge Surface Mine, subject of this BO, in those stream reaches that would receive runoff and drainage from the Sterling and Strays Surface Mine Number 1 are not expected to be different from those effects already evaluated in the Sterling and Strays Surface Mine Number 1 BO. However, several other stream reaches on the Cooper Ridge Surface Mine permit boundary would be affected by the proposed activities. While these streams are not presently occupied by blackside dace, the Service considers that migrating blackside dace could attempt to colonize these streams and these individuals could be affected.

Therefore, in addition to an incidental take expected to result from blackside dace migrating through "travel corridors” the Service has determined that, for the length of time during which OSMRE maintains regulatory authority (e.g., through Phase III Bond Release) for this Cooper Ridge permit, the currently proposed action could result in the incidental take of up to 80 blackside dace individuals per year potentially occupying 3.1 miles of streams in the watersheds where water and habitat quality would be affected by mining, with the number of individuals diminishing over time with successful reclamation.

Reasonable and prudent measures (RPMs) to minimize the take, and their implementing terms and conditions (T&Cs), that must be observed when implementing those RPMs, have been included in this biological opinion. RPMs include ensuring that the proposed action (including avoidance, minimization, and other conservation measures) is implemented as descrsibed in OSMRE’s BA and in this BO. T&Cs include: inspecting for strict compliance with all permit commitments, analysis of stream water quality and biological monitoring data to verify accuracy of predicted stream water and habitat quality impacts resulting from the proposed mining; collaborating with partners and assisting in assessment of trends in stream water quality and blackside dace populations in the affected watersheds in an effort to estimate the accuracy of the predicted blackside dace incidental take and to identify if/when the authorized level may be approached; implementing adaptive management actions as needed (as identified by the

compliance, monitoring, and blackside dace population trends); and annually reporting these results.

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List of acronyms, abbreviations, or symbols used in this document:

303d	Under section 303(d) of the Clean Water Act, states are required to submit lists of impaired waters, or waters that are too polluted or otherwise degraded to meet water quality standards.
%	percent
µS/cm	microsiemens per centimeter, the unit of measurement of specific conductivity (or specific conductance)
°C	temperature measurement, degrees Celsius
°F	temperature measurement, degrees Fahrenheit
Act	References to "Act" in this BO are to the Endangered Species Act
AOC	Approximate Original Contour
BA	Biological Assessment
BISO	Big South Fork National River and Recreation Area (NRRA), a unit of the National Park Service in the U. S. Department of the Interior
BMP	Best Management Practice
BO	Biological Opinion
BSCI	Biological Systems Consultants, Inc.
cm	centimeter
cm^2	square centimeter
dBA	decibels adjusted, a measurement of noise level
DCH	designated critical habitat
DBH	diameter at breast height, a standard method of expressing the diameter of the trunk of a standing tree; in the United States this measurement is made at a distance of 1.4 meters (m) above the ground
DBNF	Daniel Boone National Forest
DMR	Discharge monitoring reports

DNA	Division of Natural Areas (a division within the Tennessee Department of Environment and Conservation)
DOI	Department of the Interior
DOJ	Department of Justice
EA	Environmental Assessment
EPA	(U. S.) Environmental Protection Agency
ESA	Endangered Species Act (also referred to in this BO as "Act")
et al.	latin term meaning "and others"
FRA	Forestry Reclamation Approach
ft^2	square feet
g	gram(s)
GIS	Geographic Information System
ha	hectare(s)
HUC	Hydrologic Unit Code
IPac	Information for Planning and Conservation, an online tool to identify endangered species occurrences and critical habitat designations in for environmental review or project planning located at https://ecos.fws.gov/ipac/
IUCN	International Union for the Conservation of Nature (evaluates extinction risk of species and subspecies of plants and animals and produces lists (Red List) with estimate of extinction risk
KFO	Knoxville Field Office (of the Office of Surface Mining Reclamation and Enforcement OSMRE)
km	kilometer
LIWA	Local Interagency Working Agreement
m	meter(s)
m^2	square meters

n	number of observations
MOP	Mining Operations Plan
MSHA	Mine Safety and Health Administration
NLAA	Not likely to adversely affect
NPDES	National Pollutant Discharge Elimination System; permits for discharges are required by the Clean Water Act (CWA) and, in Tennessee, issued by the Tennessee Department of Environment and Conservation (TDEC)
OSMRE	Office of Surface Mining Reclamation and Enforcement
oz	ounce
P1, P2	Priority 1, 2, etc. Used in identification of hibernacula considered higher in importance for conservation of Indiana bats (lower numbers are higher priority for conservation)
Pd	*Pseudogymnoascus destructans*, the causative agent of white-nose syndrome that affects bats
PEP	Protection and Enhancement Plan
RPMs	Reasonable and prudent measures
SD	standard deviation
SL	standard length; refers to the length of a fish measured from the tip of the snout to the posterior end of the last vertebra or to the posterior end of the midlateral portion of the hypural plate (it excludes the caudal fin)
SMCRA	Surface Mining Control and Reclamation Act of 1977
SOP	standard operating procedure
spp.	Species, plural (the species included in the statement are not identified)
SSPM	Species Specific Protection Measure
T&Cs	Terms and conditions
TDEC	Tennessee Department of Environment and Conservation
TDS	Total dissolved solids

TMDL	Total Maximum Daily Load; regulatory term in the U. S. Clean Water Act, describing a value of the maximum amount of a pollutant that a body of water can receive while still meeting water quality standards.
TSS	Total suspended solids
TWRA	Tennessee Wildlife Resources Agency
USACE	U. S. Army Corps of Engineers
WNS	white-nose syndrome

Consultation history

Informal consultation for the Cooper Ridge Mine permit began during the fall of 2012 with a meeting of the agencies working together to review pending coal mining permits under the Local Interagency Working Agreement (LIWA, see Office of Surface Mining Reclamation and Enforcement 2010). At that LIWA meeting, representatives from Kopper Glo Mining, LLC (met with personnel from OSMRE, the Service, Tennessee Division of Conservation and Environment (TDEC), the U. S. Army Corps of Engineers (USACE), and the Environmental Protection Agency (EPA) to discuss data and information needs that would help these agencies review an application package for the proposed surface mining permit. An application package requesting a permit from OSMRE under SMCRA for the proposed Cooper Ridge Surface Mine would be submitted by Kopper-Glo Mining, LLC, as SMCRA permit application number 3270.

Kopper Glo Mining, LLC began discussing the permit application with OSMRE and other regulators in 2012 (see detailed list of events below). Subsequently, OSMRE, the Service, and the applicant engaged in various forms of informal technical assistance. Also, since that time OSMRE, the Service, and Kopper Glo Mining, LLC have discussed numerous issues relating to the review of the permit application and potential effects of permit issuance on species included on the Federal Endangered Species List. A summary of the review and permitting process for this permit, and events relevant to ESA coordination are included below.

May 25, 2012	The applicant's consultant (Mark V Mining and Envineering, Inc.) sent a request to the Service for information about resources of concern to the Service for use in developing an application for the proposed coal mining.
June 28, 2012	The Service sent a response to the May 25 request from the applicant's consultant. The letter mentioned potential impacts to blackside dace, Indiana bat, and gray bat. Golden-winged warbler (not listed) was also mentioned.
September 17, 2012	LIWA meeting held to discuss the preliminary mining proposal for the Kopper Glo Mining, LLC permit. Attendees included representatives for the applicant and personnel from OSMRE, the Service, TDEC, USACE, and EPA.
January 23, 2013	OSMRE received a permit application from Kopper-Glo, LLC for Kopper Glo Mining, LLC requesting issuance of a SMCRA permit for surface mining at Cooper Ridge Surface Mine, Claiborne County, Tennessee.
February 12, 2013	Meeting between Service staff and the applicant's consultant to discuss surveys and other biological data needs to prepare permit 3270 application. Another SMCRA permit application was also discussed at this meeting.
March 4, 2013	Meeting to discuss blackside dace survey requirements for two SMCRA permit applications (3264 and 3270). TDEC, OSMRE, the Service,

Tennessee Wildlife Resources Agency (TWRA), and the applicant's mining consultant.

November 29, 2016 OSMRE sent the Service a Biological Assessment (BA) and request for consultation formal consultation with them pursuant to section 7 of the ESA 16 U.S.C. § 1536 for Cumberland Gap Tennessee Coalfield Permits in Tennessee. Thirty two permits for existing and proposed facilities were included in this consultation request, including the Cooper Ridge (OSMRE permit 3270).

December 1, 2016 The Service sent a response to OSMRE's request for formal project-specific consultation for permits in the Cumberland Gap Tennessee Coalfield, indicating that OSMRE should receive the BO and incidental take statements, as appropriate, for the permits included in this consultation request, including the Cooper Ridge permit, by April 13, 2017.

December 20, 2016 A LIWA Public Hearing took place at TDEC's office facility in Knoxville. In addition to the public participants, representatives from TDEC, OSMRE, USACE, EPA, and the Service were also present.

January 23, 2017 An email between OSMRE and the Service documented the mutual agreement of both agencies that resulted from discussion on a conference call that date. The conference call topic was about separating the Cooper Ridge permit (OSMRE permit 3270) from the Cumberland Gap Tennessee Coalfield consultation request and proceeding with a separate consultation for the Cooper Ridge permit. The agencies agreed to temporarily suspended work on the Cumberland Gap Tennessee Coalfield consulation, which also suspends the afore-mentioned April 13, 2017 conclusion date for the Cumberland Gap Tennessee Coalfield.

1. Proposed action

1.1. Description of the proposed action

The proposed Kopper Glo Mining, LLC, Surface Mining at Cooper Ridge Surface Mine (OSMRE permit 3270) is located in Claiborne County, Tennessee within an 1,496-acre permit boundary (see Figure 1), approximately five miles southwest from the junction of Tennessee State Highway 90 and Valley Creek Road, and the Valley Creek community. The proposed mine site is located in the Clear Fork watershed in the Upper Cumberland River system and is drained

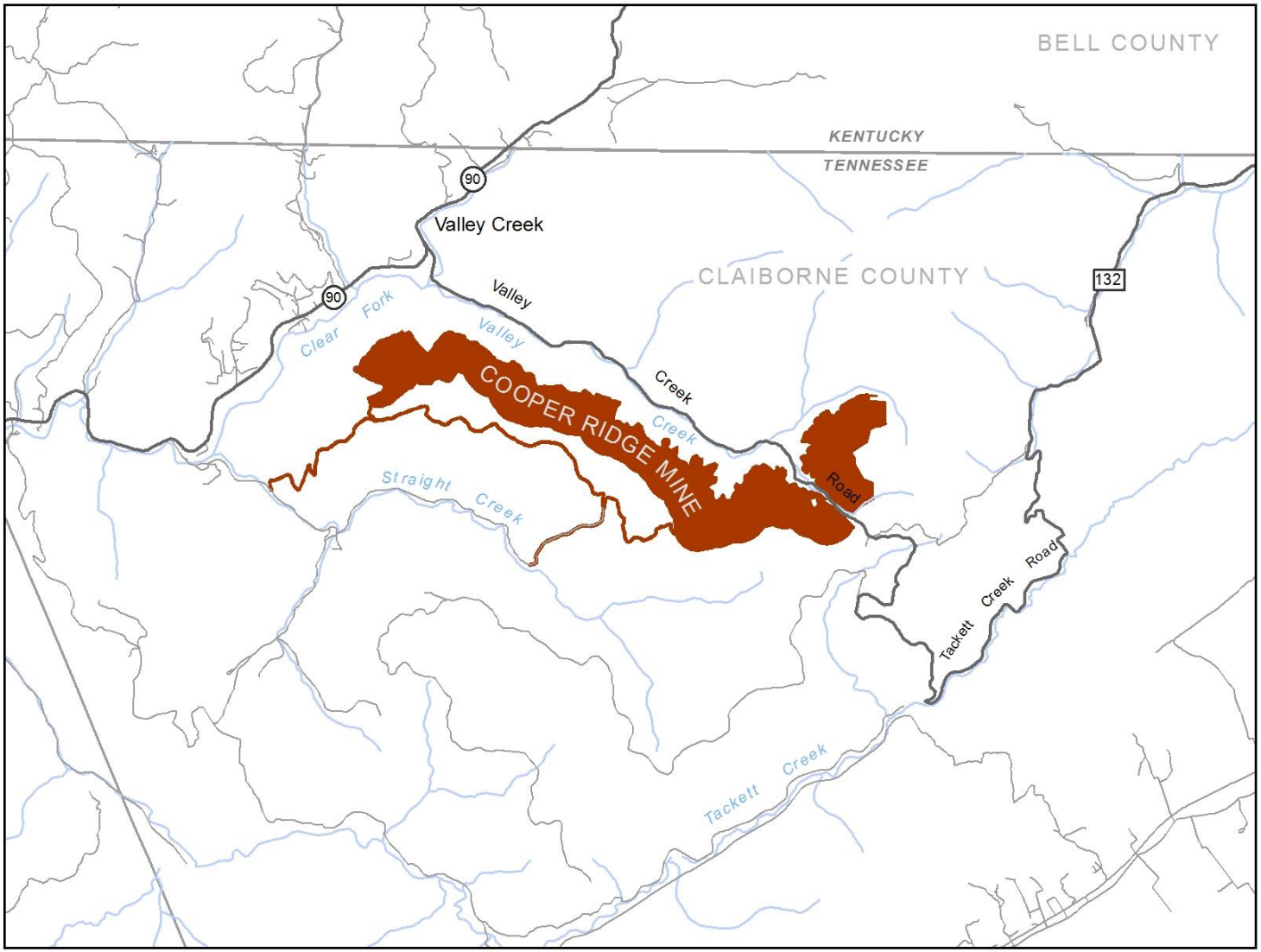

Figure 1. Location of Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270), Claiborne County, Tennessee.

locally by Clear Fork tributaries including: Nolan Branch, Clear Fork, Valley Creek, and Hurricane Creek (Figure 2). As identified in the *Consultation history* section (page 1, above) of this BO, the original application for a SMCRA permit was submitted to OSMRE by Kopper Glo Mining, LLC in January 2013.

The mining authorized by the issued permit would result in removal of coal from the Jellico (locally known as the Mingo or Mason) coal seam. The mining and associated activities would disturb approximately 752.2 acres within a 1496.3-acre permit boundary, of which 472.5 acres have been previously mined, over the five year operation authorized by the issued permit and included in the mine plan. The entirety of the 472.5-acre area has been previously mined, some of which took place prior to implemtation of SMCRA, and the mining would reclaim portions of the Jellico (Mason) mine bench (highwall) that remains unreclaimed, including an estimated 5.4 miles of highwalls (Tennessee Department of Environment and Conservation 2016a),. Information included in the OSMRE permit application estimated existing, unreclaimed highwall at 5.1 miles (Middleton 2017).

Highwall is a term used to note the cliff-like remnant of previous mining conducted before the passage of SMCRA. Depending on safety issues, environmental site stability, and economics, the mine plan (Office of Surface Mining Reclamation and Enforcement 2013) includes “re-mining” these abandoned, previously mined areas to salvage coal that was left behind, and using the remaining overburden to reclaim as many highwalls as possible. Reclaiming the highwalls would result in blending the mined highwall into the surrounding terrain to approximate the original contour of the land prior to mining.

The mine site is proposed to be bonded in three increments. Increment one (1) will include the sediment basins and roads, and consists of 139.2 acres (29.1 acres of sediment basins, 23.0 acres of bench access roads, and 87.1 acres of haul roads). Increment two (2) will include the surfaced and auger mine areas minus the on-bench sediment basins and bench acess road (included in increment 1). Increment 2 consists of 1,357.1 total acres (333.3 acres of surface mine area, 253 acres of mine management area, and 770.8 acres of auger mine area mine road overlaps). Increment three (3) will include the face-up and temporary spoil storage areas, and consists of 13.7 acres, all of which is included within the boundaries of Increment 2.

The mining would be accomplished using combinations of contour mining, auger/highwall mining. The mine is expected to annually produce approximately 360,000 tons and the life-of-mine (9 + years) production is estimated at approximately 3,329,724 tons. Existing roads would be used to access the site by equipment and personnel. These existing roads currently meet SMCRA standards, so no upgrades are required.

As identified above, all 472.5 acres of the permit area that will be surface mined was mined prior to the enactment of SMCRA, beginning in the 1950’s and continuing through the early 1980’s. As a result, reclamation that is now required by SMCRA, has not been accomplished on these areas, and approximately 5.1 – 5.4 miles of un-reclaimed highwall remain exposed (Tennessee Department of Environment and Conservation 2016a).

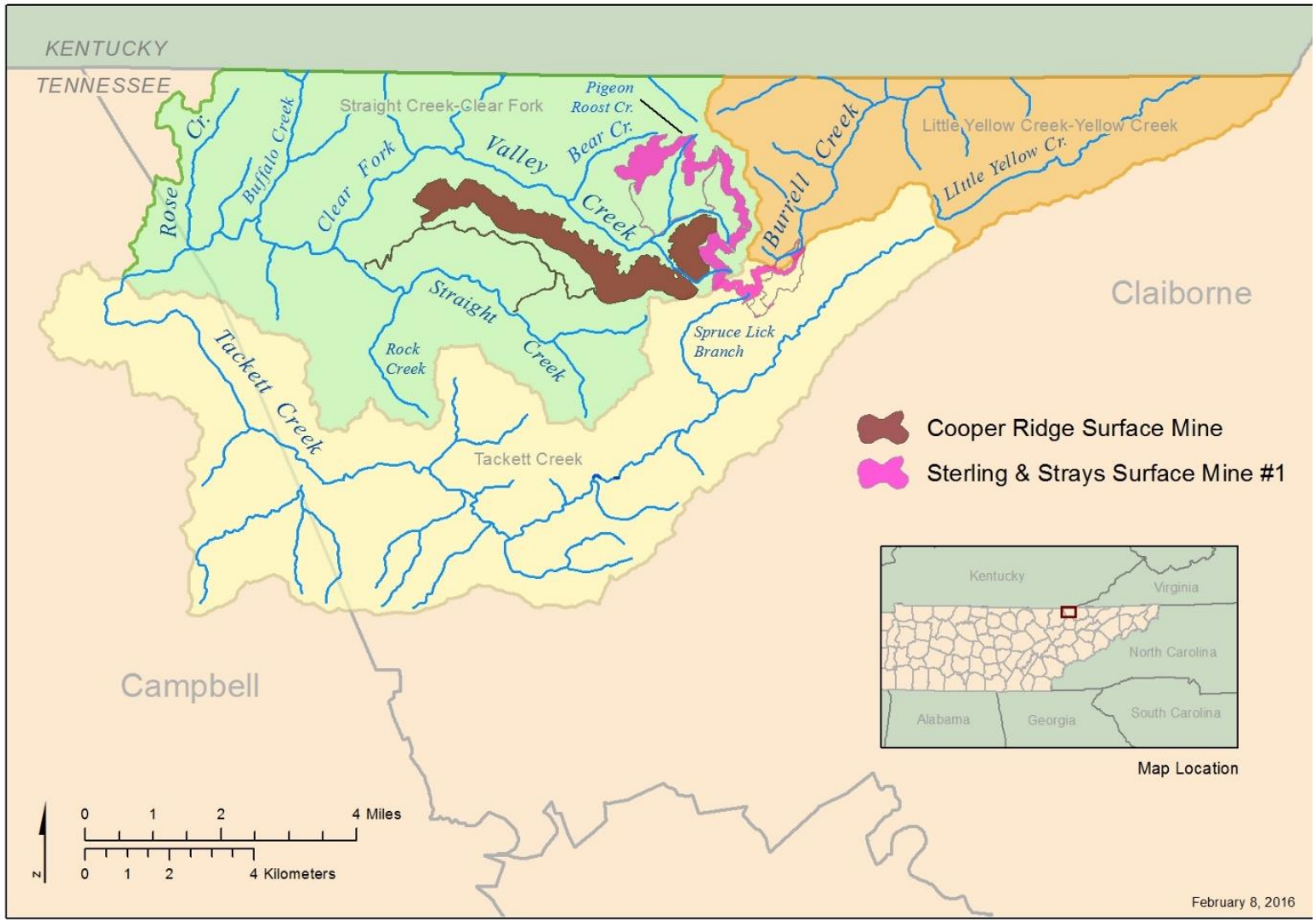

Figure 2. Location of Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270, Claiborne County, Tennessee) in the context of streams in the Clear Fork system of the Upper Cumberland River drainage.

1.2. Project components

1.2.1. Haul roads

The permit area is accessed via Claiborne County roads, Tackett Creek Road and Valley Creek Road (Figure 1). Coal will be hauled on roads within the permit boundary and offsite for further processing via Tackett Creek Road or to Valley Creek Road.

Portions of haul roads A, C and H (Mining Operations Map [MOP], included in Office of Surface Mining Enforcement and Reclamation 2013) are existing structures that were upgraded by Mountainside Coal Company as provided for in previous SMCRA permits (OSMRE permits 3052, 3058, and 3127). In addition, portions of haul roads A, C, D, and E are shared structures with Mountainside Coal Company. Mountainside Coal Company is currently in reclamation and is no longer actively using the haul roads. Therefore, Kopper Glo Mining, LLC has included these roads with this current Cooper Ridge permit to allow Mountainside Coal Company to apply for bond release for the permit(s) that included these haul roads.

After mining has been completed, all roads will remain as permanent facilities. After mining has been completed, road ditches and culverts will remain to simplify future maintenance, promote positive post mining drainage, and to reduce erosion and sediment loads to the receiving streams. As a result, within the permit boundary an estimated 85 acres of access roads will be retained as permanent facilities in the post-mining environment.

1.2.2. Pre-mining site preparation

Bulldozers, front-end loaders, trucks, and graders will be used to clear timber and other vegetation in advance of mining and reclamation operations within the proposed permit area. With the exception of the haul roads, the disturbed areas within the proposed permit area, will be reclaimed utilizing all reasonably available spoil. However, if maximum cuts are not taken for safety, environmental site stability, or economic reasons, portions of the pre-law highwall on the Jellico (Mason) coal seam may remain. The mine is expected to have an average annual production of approximately 360,000 tons and life-of-mine production of approximately 3,329,794 tons.

1.2.2.1. Tree and vegetation removal

Clearing and grubbing will be conducted in advance of the mining operations. Timber which can be salvaged and marketed will be harvested and transported off site. All other woody vegetation encountered by mining or in construction of related facilities will be burned in a controlled manner, windrowed along the coal outcrop in a manner that will not cause instability, or placed into the final layer of growth medium before reclaiming.

Although no Indiana bats or northern long-eared bats are known to use trees on the permit boundary for maternity roosts (see *Indiana bat summer roosting habitat, female*, section 2.1.3.2 and *Northern long-eared bat staging, spring migration, and summer roosting*, section 2.2.5 in

this BO) OSMRE has conservatively assumed females of either bat species may use trees within the permit boundary for maternity roosts. For the first two years of the permit (from permit issuance through March 31, 2019) OSMRE and the Service have agreed to allow tree removal during any month of the year **except during June and July of those two years**, when any young bats in maternity roosts within the permit boundary would not be volant (able to fly). However, for the remaining duration of the permit no tree removal to facilitate mining will be allowed between March 31 and November 15, as specified in the Rangewide Indiana bat Protection and Enhancement Plan Guidelines (U. S. Fish and Wildlife Service and Office of Surface Mining Reclamation and Enforcement 2013).

1.2.2.2. Sediment pond construction

Sediment ponds are dugout structures designed and placed to control runoff and collect sediment throughout the mine permit boundary. Where possible, the structures are located in areas that will discharge into natural drainways. Twenty-five sediment ponds and a network of drainage ditches are included in this mine plan. All 25 sediment ponds will be newly constructed for this mining project. All ponds associated with the Cooper Ridge permit (OSMRE 3270) will be removed no sooner than two years after the reclamation has been satisfactorily accomplished within the watershed, with approval from the regulating authority

New ponds will be constructed during dry, low flow periods whenever possible, and during construction temporary sediment control measures consisting of two rows of straw bales and silt fences will be placed downstream of ponds or ditches during construction when the drainways are flowing. Topsoil and other material (rocks, gravel, boulders, sand, etc.) excavated during this construction will be stored nearby for later use in reclamation. Disturbance to vegetated areas around pond locations during construction will be minimized as much as possible. Sediment ponds and diversion ditches will be lined with claylike material, if available, or synthetic liners.

1.2.2.3. Other mine facility placement and use

With the exception of the haul roads described in *Haul roads* section (1.3.1), no permanent support facilities are planned for this operation. Within the permit boundary temporary facilities associated with the mining operation and equipment support will include a mine office (a portable trailer), two explosive storage magazines (10 feet by 10 feet structures), two 3,000 gallon fuel storage tanks, four 500-gallon oil tanks, eight drop trailers, and an anfo bin. Equipment and employee parking will generally be near the working area.

The explosive storage magazines will be located at distances prescribed by the Mine Safety Health Administration (MSHA) from active mining operations. Otherwise, all structures will be portable structures that will be relocated across the mine site, as necessary during mining. All structures will be removed from the mine site when mining is completed and permanently ceased.

1.2.3. Mining operations

As mentioned in the *Description of the proposed action* (section 1.1) above, the surface mining includes remining of the Jellico (Mason) coal seam. Kopper Glo Mining, LLC will use surface contour (or excavation on the sides of the mountain) and auger/highwall mining methods for the mining.

Surface contour methods excavate coal from the sides of the mountain following the coal along the contour and moving around the mountain. Auger/highwall methods extract coal by drilling horizontal holes rotary shafts and hydraulic rams into the exposed coal outcrops along the highwalls, moving along the length of the highwall.

The remining portion of the project will remove the maximum amount of coal economically, environmentally, and safely feasible over approximately 472.5 acres where previously unreclaimed mine operations have taken place. Remining these areas could result in reclaiming up to an estimated 5.1 – 5.4 miles of pre-SMCRA highwall (Tennessee Department of Environment and Conservation 2016a). These pre-SMCRA highwalls will be reclaimed using all available spoil material generated as the mining progresses. However, portions of highwall may remain on the Jellico coal seam, and the maximum projected cuts may not occur, for safety, environmental, and economic considerations, as mentioned above.

Although such areas are unknown, if the mining results in landslides in unstable areas associated with previously unreclaimed highwalls, vegetated buffers will be established and maintained, and drainage control features will be designed to route water away from the slide areas to prevent further movement in these unstable areas. Otherwise, if unstable areas are identified that might cause landslides, these areas may be completely avoided, or solid berms may be established around other unreclaimed areas in order to minimize the likelihood of future slides.

Dozers, drills, loaders, trucks, graders, excavators, and augers will be used to mine the coal, construct, upgrade, and maintain roads and sediment control structures, to transport spoil and growth medium, and to reclaim all surface disturbances. It is expected that one spread of equipment will be used on the mine site. Mining will advance along the crest of the ridge with half of the entire ridge being excavated and coal removed. After excavation is completed and coal is removed from one “cut” area, mining will proceed on the opposite side of the ridge.

1.2.3.1. Blasting

Blasting and/or equipment will be used to push spoil material into the previously mined area for reclamation. Federal regulations (30 CFR 816.97[b]) require that noise associated with blasting may not exceed 129 to 133 db (depending on the type monitoring equipment used to measure the blast noise). These standards have been set to protect public health and safety and were not intended to preserve the highest levels of aesthetic qualities in an area. Under Federal regulation, blasting will occur only between sunrise and sunset. The blasting plan for the proposed operation indicates that blasts will occur no more than twice in a given day, with no more than six blasts in any given week, and will not occur on Sunday. As such, blasting noise will occur infrequently

(once or twice per day), will last only for a very brief period (i.e., a few seconds), will occur only during daylight hours, and will not occur on Sundays.

1.2.3.2. Excavation and coal removal

Mined coal will be temporarily stockpiled at various locations on the proposed permit area prior to being loaded onto trucks and hauled away. These coal stockpiles will be located on or near the mine pit active at the time and within areas providing drainage control. Coal will then be removed by truck, using the internal mine roads and proceeding to Claiborne County roads.

1.2.4. Expected future conditions

With the exception of retaining the existing road system, no further development within the permit boundary is intended and the postmining land use within the permit boundary is expected to be undeveloped forest that will facilitate restoration of fish and wildlife habitat. A secondary postmining land use of industrial/commercial may be implemented for oil and gas well development, as the oil and gas rights to portions of the property have been leased out, as listed in item 17 of the permit application.

A variety of native shrubs, trees, and herbaceous ground cover species will be planted. However, for several years after mining, the ground cover species will provide the principal component of the postmining habitats. The conservation measures included in the mine plan, and the Indiana and northern long-eared bat Protection and Enhancement Plan (PEP) included as SMCRA 3270 permit conditions, specifies tree species composition, density, and other planting requirements intended to establish eventual forest habitats appropriate for use by these and other bats. However, until the planted trees and other naturally invading trees become established and the forest canopy shades the ground cover, the reclaimed areas will be dominated by herbaceous species and woody shrubs. Therefore, it may take 15 to 30 years to establish terrestrial habitats similar to pre-mine habitats present within the mine permit boundary, and 50 to 75 years or more, following reclamation, before mature forests are established on the reclaimed mine site.

Mining and reclamation will alter the existing vegetative communities present on approximately 752 acres of the 1,496 acres within the mine permit boundary. Vegetation on an estimated 85 acres within the permit boundary was previously eliminated to construct the existing haul roads for the mining, logging, and oil and gas extraction that has previously taken place within the permit boundary (Office of Surface Mining Reclamation and Enforcement 2013).

1.2.5. Reclamation methods

Backfilling and grading will be done in conjunction with active mining. The spoil taken from the initial cuts will be placed in an unreclaimed mine pit and will be used to backfill the second cut highwall. The final cuts will be backfilled with the material that has been back stacked on preceding cuts. However, of the acreage that will be disturbed by the proposed mining and reclamation, approximately 472 acres of the mine permit area include abandoned pre-SMCRA surface mines and associated unreclaimed highwalls. Any topsoil present on these acres prior to the previous mining on these areas was likely pushed downslope and then buried by the

overburden that was left behind when coal was mined. Since little or no native topsoil is available in these areas, the applicant will use a blend of topsoil, woody vegetation, and weathered sandstone or shale as the plant growth medium.

Stockpiles for growth medium will be clearly marked for identification, located on stable areas within the permit area, and graded to ensure positive drainage. Diversion swales, windbreaks and/or other measures will be used to protect the stockpiles from wind and water erosion. If storage is to exceed 30 days these stockpiles will be seeded and mulched.

All of the disturbed acreage (752.5 acres) not already permanently altered by roads that will remain on the permit boundary after mining has been completed will be reclaimed with tree plantings according to the FRA. The FRA, a five step reclamation approach based on forestry research used to reclaim coal mine lands (U. S. Department of Interior 2015), includes placement of at least four to six feet of loosely graded growth medium over the backfilled slopes. However, in areas where potentially toxic-forming materials have been identified, the loose compaction component of the FRA will not be utilized. In those areas, backfill and growth medium will be compacted per normal backfilling and grading practices.

Within thirty days after growth medium has been distributed as described above, these areas will be planted with a mixture of grasses and legumes in order to accelerate establishment of vegetative cover and minimize erosion. Once the grasses have been sown, trees will be planted on the reclaimed mine site to facilitate the establishment of wildlife habitat.

1.2.6. Conservation measures

1.2.6.1. Sediment control

Sediment ponds

Sediment ponds are dugout structures whose engineering and placement is specifally designed to control runoff and collect sediment throughout the mine permit boundary. Sediment ponds designed for mining permits in Tennessee are designed with higher storage capacity than many other states in the Appalachian region (Office of Surface Mining Reclamation and Enforcement unpublished data, and see Appendix B in this BO). This storage capacity reduces the need for frequent sediment removal from the ponds, actions which can also result in temporary sediment releases. Where possible, these sediment pond structures are located in areas that will discharge into natural drainways.

Twenty-five sediment ponds and a network of drainage ditches are included in the Cooper Ridge Surface Mine mine plan (Office of Surface Mining Reclamation and Enforcement 2013). These ponds will be constructed during dry, low flow periods, when possible. Temporary sediment control measures consisting of two rows of straw bales and silt fences will be placed downstream of ponds or ditches during construction when drainways are flowing. Topsoil and other material (rocks, gravel, boulders, sand, etc.) excavated during this construction will be stockpiled nearby for later use in reclamation. Disturbance to vegetated areas around pond locations will be minimized as much as possible during construction. Sediment ponds and diversion ditches will

be lined with claylike material, if available, or synthetic liners if clays are not present. Diversion ditches used to direct or divert runoff to these sediment ponds may include sumps and/or rock check dams to provide additional sediment storage volume.

When a pond reaches approximately 80% of its designed storage volume accumulated sediment will be removed (and see Appendix B); water in the structure will be sampled, and treated (according to National Pollitant Discharge Elimination System, NPDES, permit requirements for maintaining appropriate pH for discharges) if necessary, prior to de-watering of the pond by pumping or siphoning. Non-acidic/non-toxic sediment will be transported to an area identified for such storage, allowed to dry, and later used as spoil or topsoil material, as appropriate. Any acidic/toxic sediment will be disposed according to the approved acid/toxic materials handling plan and covered with a minimum of four feet of clean non-toxic material (more detail regarding this topic is provided below).

Once the ponds have been built they will be regularly inspected for structural weakness, design failure, compliance with permit discharge requirements, erosion or other hazardous conditions. OSMRE is required (mandated) to inspect the permit four times per quarter, and these inspections must include at least one complete inspection and three partial inspections. All ponds are inspected during complete inspections, but they can also be inspected during partial inspections. In addition, the SMCRA permit requires basin annual certifications for each pond, and the TDEC NPDES permit associated with the Cooper Ridge Surface Mine will also require quarterly monitoring and reporting, by sumitting Discharge Monitoring Reports (DMRs). Additional reporting is also required after certain size rain events (see Tennessee Department of Environment and Conservation 2016b).

Stream crossings

Standard buffer zones included in SMCRA regulations are 100 feet on each side of a stream. However, the Cooper Ridge Surface Mine application package (Office of Surface Mining Reclamation and Enforcement 2013). However, all 15 of the stream reaches included in the area proposed to be surface mined (totaling 7,488 linear feet of stream impact) have been previously altered, with many of the stream channels moved, by pre-SMCRA mining. Therefore, since these stream channels had been relocated by the previous mining, and some have cut channels through existing mine benches that were not reclaimed, Kopper Glo Mining, LLC requested stream buffer zone variances that will allow them to mine through the existing stream channels. After mining has been completed in these areas, the surface water flow will be restored to the original stream channels (Tennessee Department of Environment Reclamation and Enforcement 2016a, Office of Surface Mining Reclamation and Enforcement 2013). The streams included in this variance and proposed restoration include six perennial streams and seven wet weather conveyances, and the reaches consist of: three sections of stream channel in an unnamed tributary to Nolan Branch; a total of 11 sections of stream channels in unnamed tributaries to Valley Creek; and one section of stream channel in an unnamed tributary of Hurricane Creek (Figures 2 and 3).

1.2.6.2. Toxic material handling plan/waste disposal

In accordance with the approved Hydrologic Reclamation Plan and the Toxic Material Handling Plan (Office of Surface Mining Reclamation and Enforcement 2013), if strata that might produce acid or toxic drainage are encountered during mining, measures to avoid adverse impacts from acid or toxic runoff include:

- Removing the rider seam,
- Selective handling of toxic material, or
- Blending the thin and discontinuous potentially acid producing strata with alkaline materials.

In addition, the Hydrologic Reclamation Plan and the Toxic Material Handling Plan includes the following commitments:

- The auger development waste, coal cleanings, and sediment cleanings will be tested prior to final disposal in the above described high and dry manner. This testing will determine the amount of lime required to effectively neutralize the acid content of this material. The needed lime will be broadcast onto the toxic material after final placement, prior to covering with a minimum of four feet of clean non-toxic material; the remaining backfill material will then be placed.

Geologic sampling conducted prior to permit issuance identified zones where acid or toxic (selenium) producing strata could be encountered when the Jellico (Mason) coal seams are mined Office of Surface Mining Reclamation and Enforcement 2013). Although no areas with potentially acid-forming materials occurring were identified, the application indicated that small, potentially acid-bearing zones, left behind from the previously unreclaimed mining could be encountered.

If toxic or acid materials are encountered, approximately five feet of strata above and below each coal seam will be segregated and isolated within pods in the backfill. These isolation pods will be located well above the pit floor and final highwalls and a minimum of four feet below the surface in order to minimize contact with ground water and to minimize oxidation, respectively, that could result in release of toxic forms of these materials into the environment. Any bottom strata below a coal seam that is not disturbed will remain in place (not removed and stored elsewhere). Pit cleanings from the coal or auger/highwall wastes will be assumed to be potentially acid-forming, and will also be segregated and placed high in the backfill to prevent contact with ground water. Other waste material, such as road sump, drainage ditch and sediment pond cleanings will also be tested for toxicity, and if found to be toxic or potentially acid forming, these materials will also be segregated and placed high in the backfill away from anticipated ground water levels and covered by at least four feet of non-acid and non-toxic spoil or soil material. Other thin and discontinuous acid-forming zones, which may randomly occur throughout the proposed permit area and not directly associated with a coal seam, will be neutralized by mixing with alkaline materials during mining and backfilling.

1.2.6.3. Protection and enhancement measures for Indiana bat and northern long-eared bat

Kopper Glo Mining, LLC incorporated a northern long-eared bat and Indiana bat protection plan into the Fish and Wildlife PEP that is a component of the mine permit application required by SMCRA. Actions that would avoid impacts to Indiana bat and northern long-eared bat. These include actions that avoid and minimize impacts to individual bats, and actions that enhance or improve habitats for Indiana bat and northern long-eared bat. These actions are described in more detail below.

Avoidance

- Avoidance of Caves and Underground Mines – According to information provided by the applicant in the SMCRA permit application (Office of Surface Mining Reclamation and Enforcement 2013), a winter habitat assessment of the permit boundary was performed and no open caves or underground mines were found. Any adjacent auger holes had been covered prior to applying for this permit, and they were covered when the OSMRE application was received.
- For the first two years of the permit, tree removal to facilitate the mining will be allowed at any time except during June and July, until March 31, 2019. The June and July restriction would ensure no (unknown) maternity roost trees would be disturbed during the time when bat pups would not be yet be able to fly (non-volant).
- After the first two years of the permit (after March 31, 2019), tree cutting restrictions will only allow tree removal during the time period when bats would be hibernating and not present on the forest landscape (e.g., between November 15 and March 31). This condition is in accordance with the *Rangewide Indiana Bat Protection and Enhancement Plan Guidelines for Surface Coal-Mining Operations* (U. S. Fish and Wildlife Service 2013).

Minimization

- Tree Clearing Restrictions – The mine project boundary is not within a five-mile radius of known hibernacula for gray bat, northern long-eared bat, or Indiana bat, nor is any winter habitat for these species present within the permit boundary (see bullet above). Therefore, the permit boundary contains only potential summer roosting bat habitat for northern long-eared bat or Indiana bat and potential foraging habitat for gray bat. All trees within the surface mining portion (approximately 472 acres) of the permit boundary could be removed. This includes potential northern long-eared bat or Indiana bat maternity/roosting trees, (i.e., any dead trees, live trees with dead snags, or live trees with diameter at breast height [DBH] of three inches or greater, with exfoliating bark) or male summer roosting trees, will be removed.
- Stream sediment control measures will minimize impact of sediment runoff to stream habitats used by bats for water supply and foraging (see sections 2.1.3.5 and 2.2.7).
- Although no hibernacula are known within a five-mile radius of the permit boundary, the blasting plan is intended to limit the amount of ground vibration

that could disturb hibernating bats (in unknown hibernacula nearby), and also to prevent blasted rock being thrown that could destroy bat habitat or harm bats in adjacent areas beyond the permit boundary where bats may be roosting and/or foraging.

Enhancement

- Tree Planting - During reclamation the portion of the permit area that was surface mined will be planted with species of trees that will, when mature, provide potential summer roost habitat for forest-dwelling bats.
- Tree Girdling – Girdling trees along the permit area perimeter, on the undisturbed berm within the permitted area, or along the perimeter of roadways within the permitted limits of roads will create roosting habitat for forest-dwelling bats. In these areas, one tree that is at least nine inches diameter at breast height (DBH) will be girdled approximately every linear 500 feet (for an estimated total of 85 girdled trees, Office of Surface Mining Enforcement and Reclamation 2013). The trees species selected for girdling will include those with exfoliating bark. Trees with smooth, tight bark will not be girdled.
- Reclamation of the sediment basins will result in wetlands, providing water sources for bats that could forage on the area after mining has been completed.
- As soon as possible after coal has been removed and backfilling and grading operations are completed, the mine site will be revegetated with a seed mixture selected for rapid establishment of herbaceous ground cover tolerant of post mining soil conditions. The herbaceous ground cover will be compatible with tree growth and provide benefits to wildlife. With the exception of permanent roads, ponds, and other existing structures, all disturbed areas will be planted in plots of trees and shrubs when mining has been completed. In order to satisfy the criteria for final bond release, the required tree stocking success is at least 400 stems per acre (including volunteer colonizers).

Forested acres suitable for roosting and foraging habitat are located adjacent to and in close proximity to the permit area. If large open areas are present within the permit area after the site is reclaimed and trees are planted, travel corridors linking roosting and foraging habitats will be planted. These travel corridors will be composed of at least four rows of trees and will be at least 50 feet in width (Per SMCRA Permit Item Number 59-Revegetation Plan and Item Number 60-Fish and Wildlife Protection and Enhancement Plan (Office of Surface Mining Reclamation and Enforcement 2013).

In summary, the post-mining land-use includes wildlife habitat that will consist of trees and shrubs being planted in the surface-mined portion of the permit boundary, and will result in the reforestation of approximately 472 acres of the project area with species of trees that could be suitable for bat use within 40 to 80 years of planting, depending upon the specific habitat (tree size) required by each bat species.

1.2.6.4. Protection and enhancement measures for blackside dace

OSMRE required Kopper Glo Mining, LLC to incorporate a blackside dace protection plan into their Fish and Wildlife PEP. That PEP and additional measures include:

- Actions that would avoid impacts to blackside dace and their habitats;
- Actions that would minimize impacts, including construction and maintenance Best Management Practices (BMPs);
- Actions that enhance or improve habitats for blackside dace; and
- Physical, chemical, and biological, monitoring programs, accompanied by adaptive management plans to address monitoring results suggesting mine-related problems that might need addressing. These actions are described in more detail below.

Avoidance

- A plan for handling and disposing of toxic materials (see section 1.3.6.2) will be developed using existing information that identifies areas where rock or soils that would expose acid or other toxic materials could be exposed during mining. Carefully implementing that plan as needed when acid or toxic materials were encountered during mining would avoid ground and surface water contamination.

Minimization

- The potential for erosion that could degrade stream habitats in watersheds receiving runoff from the mine would be minimized by incorporating and effectively installing and implementing BMPs and proper engineering techniques to prevent erosion to control sediment runoff at stream crossings and across the mine permit.
- Erosion potential would also be minimized by placing engineered structures in locations where erosion is likely to occur or where a change in flow direction or force is anticipated.
- The potential for hazardous substances to contaminate downstream portions of the mined watersheds in stormwater runoff from the mine site would be minimized by installing and maintaining berm buffers around facilities where hazardous substances could be present.
- Water quality and habitat degradation to streams receiving drainage from mined areas would be minimized by surfacing haulroads with durable material and constructing sediment sumps to collect run-off from roads not located in areas where drainage would be collected in sediment ponds.
- Water quality and habitat degradation to streams receiving drainage from mined areas would be minimized by maintaining the required sediment clean-out level and pond storage capacity.
- Water quality and habitat degradation to streams receiving drainage from mined areas would be minimized by maintaining discharges from the mine site within the National Pollutant Discharge Elimination System (NPDES) effluent limitations for total suspended solids (TSS) to ensure compliance with a Total Maximum Daily Load (TMDL). (Tennessee Department of Environment and Conservation 2016). This would be

accomplished by constructing sediment basins designed so that effluent from the site will not exceed 0.5 milliliters per liter peak suspended solids during a 10-year/24 hour precipitation event.

- Properly lining all drainage structures with either grass or rock and splash pads would control erosion and minimize impacts to water quality and habitat in streams receiving drainage from mined areas. Alternate sediment control in the form of hay bale dikes or silt fencing would be utilized during basin construction and removal to minimize the increase in solids in both the receiving streams and groundwater.
- Increases in suspended and dissolved solids in streams draining the mine site would be minimized by promptly seeding and mulching road, ditch, and embankment outslopes after construction has been completed.
- Disturbances to streams would be minimized by restricting impacts to crossings of existing roads, where existing culverts will be upgraded or replaced.

Enhancement

- A total of 9,296 feet of stream channels (3,871 feet of stream and 5,425 feet of wet weather conveyances) that had been altered and not restored by the prior earlier mining will be restored to original stream channels (Tennessee Department of Environment and Conservation 2016a).

Monitoring and adaptive management

In order to identify conditions that would require specific management actions to correct a problem, OSMRE analyzed pre-mine biological and water quality survey data. These analyses identified baseline water quality conditions and described the biological community in watersheds receiving drainage from mined areas. In addition, the predictive modeling (using Johnson et al. 2010) described anticipated changes that might occur during and following completion of mining (Office of Surface Mining Reclamation and Enforcement 2016a). The data used for this analysis were collected via continuously recording conductivity meters that had been deployed as part of a multi-agency sampling plan and partially in support of the OSMRE's request for ESA Section 7 consultation for the proposed approval of surface mining at Middlesboro Mining Operations, Inc. Surface Mining at Sterling and Strays Surface Mine Number 1, OSMRE Permit Number 3296 (U. S. Fish and Wildlife Service 2016a, and see Figure 3).

Those data included some of the same streams that would receive runoff and discharges from the proposed Cooper Ridge surfaced mine, and therefore were also useful in identifying daily and seasonal fluctuations and trends for specific conductance. If increases in sediment or specific conductance (or other parameters of concern) over the levels predicted by these analyses are encountered or reported, OSMRE and the Service will investigate the likely source(s) with appropriate sampling and analysis and, as appropriate, and measures to ameliorate adverse water quality trends will be implemented. At a minimum, stream monitoring survey data will be analyzed during the permit mid-term review (after mining has been ongoing for 2 ½ years), at any time the permittee (Kopper Glo Mining, LLC) requests revision of the existing permit, and at

permit renewal. Any unexpected trends that are associated with the mining occurring under this permit could result in an action plan to address the problem(s).

The blackside dace PEP for the Cooper Ridge Surface Mine permit also includes:

- An analysis of potential water quality impacts to receiving watersheds, using the method presented by Johnson et al. (2010) was used to predict parameters (e.g., specific conductance) known to be important predictors of blackside dace occurrence. This analysis was included in OSMRE's Draft CHIA, (Office of Surface Mining Reclmation and Enforcement 2016a) used in preparation of this BO.

1.3. Species considered in the consultation

Section 1.6 below lists the species originally included by OSMRE in their assessment of potential project impacts. The rationale for eliminating some of these species from further consideration is provided below. This BO assesses the potential for the project to adversely affect Indiana bat, northern long-eared bat, and blackside dace.

1.3.1. Species and designated critical habitat originally considered but eliminated from further consideration in the BO

OSMRE reviewed information provided by the Service's, Cookeville Ecological Services Field Office and the Service's IPaC database (http://ecos.fws.gov/ipac/, and see U. S. Fish and Wildlife Service 2016b) to identify 23 species federally threatened (T) or endangered (E) species and critical habitat for nine species considered for possible inclusion in this consultation. Refer to Figure 4 for general locations of the identified watersheds for aquatic species listed in Table 1.

Most of the species listed in Table 1 are only known to inhabit the Powell, upper Clinch River and upper Cumberland watersheds (Figure 4). Because any effects of this proposed mine project will affect only the Clear Fork watershed of the upper Cumberland River system and mine drainage would not affect any streams in the Tennessee River system (including the Upper Clinch River or the Powell River, see Figure 4). OSMRE (Office of Surface Mining Reclamation and Enforcement 2016b) concluded that species found only in these areas would not be affected by the proposed mining. In addition, while the Cumberland darter's range includes the upper Cumberland River system, its occurrence is in the Jellico Creek watershed, which would not be affected by effluent from any streams draining the action permit boundary.

Species, for which OSMRE made a determination of a "No Effect" include: Appalachian monkeyface, cracking pearlymussel, Cumberlandian combshell, Cumberland darter, fanshell, finerayed pigtoe, fluted kidneyshell, oyster mussel, dromedary pearlymussel, pink mucket, purple bean, rough pigtoe, rough rabbitsfoot, sheepnose mussel, shiny pigtoe, slabside pearlymussel, spectaclecase, pygmy madtom, and spotfin chub. OSMRE (Office of Surface Mining Reclamation and Enforcement 2016b) also concluded that the proposed project would not adversely modify designated critical habitat for the following species: Cumberlandian combshell, Cumberland darter, fluted kidneyshell, oyster mussel, purple bean, rough rabbitsfoot,

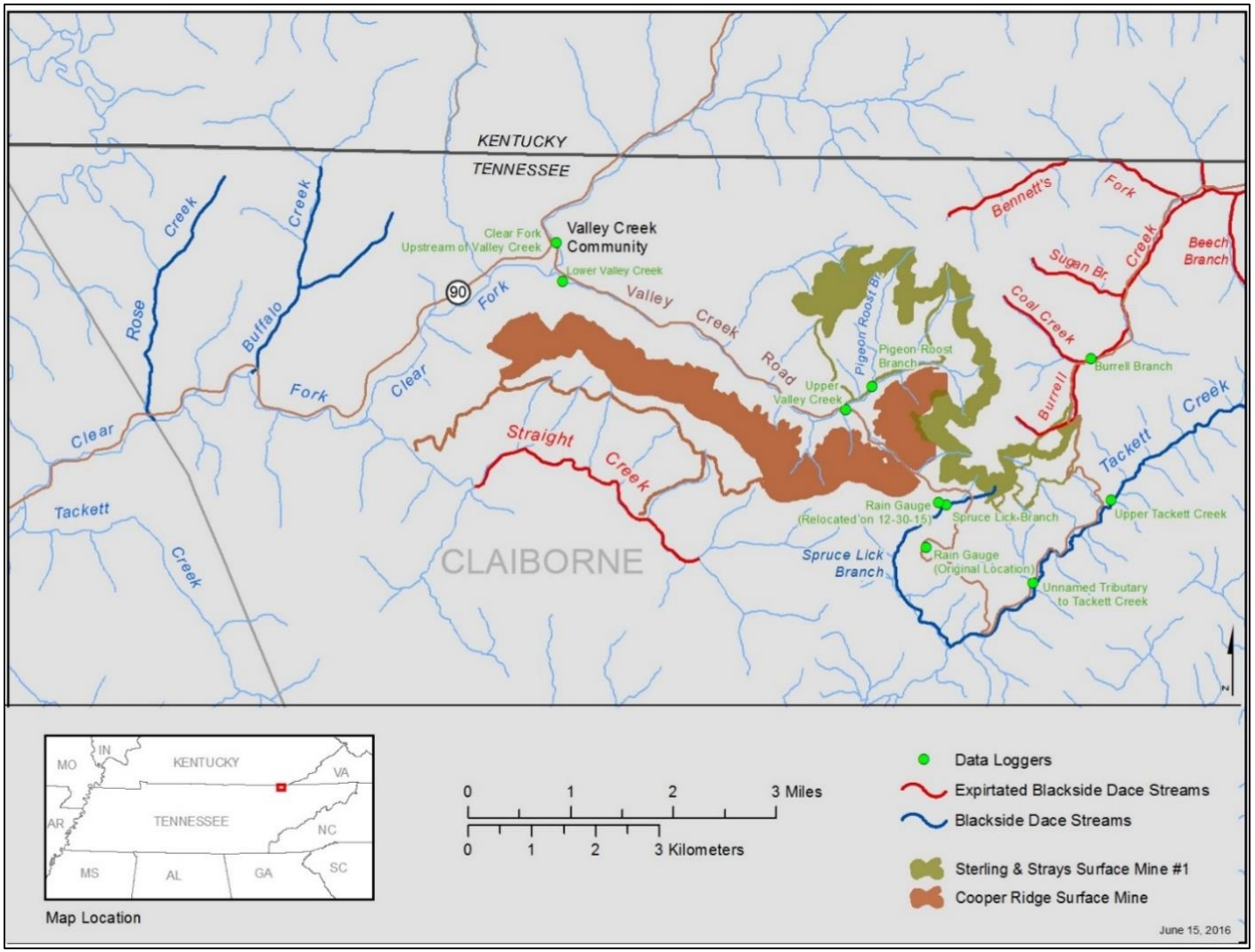

Figure 3. Locations where data loggers have been continuously recording baseline data on conductivity (May 2015 to present) in streams associated with the Middlesboro Mining Operations, Inc. Sterling and Strays Surface Mine Number 1, OSMRE permit 3296, Claiborne County, Tennessee (U.S. Fish and Wildlife Service 2016a), and also used for evaluating potential effects of the Cooper Ridge Surface mine (OSMRE permit 3270).

Table 1. Species and critical habitat considered in OSMRE's assessment of potential impacts from issuing a SMCRA permit (3270) to Kopper Glo Mining, LLC Cooper Ridge Surface Mine, Claiborne County, Tennessee. Species included in Table 1 were identified based on OSMRE's review of data on species' occurrences for the mine permit and vicinity, vailable in the Service's IPaC database (http://ecos.fws.gov/ipac/).

Common Name	Watershed	Present in Action Area?	May Affect?	Likely to Adversely Affect?	Critical Habitat Designated/Affected?
Blackside Dace	Clear Fork	YES	YES	YES	NO/NO
Cumberland elktoe	Powell River	NO	NO	NO	YES/NO
Northern riffleshell	Powell River	NO	NO	NO	NO/NO
Fanshell	Upper Clinch River	NO	NO	NO	NO/NO
Gray bat	NA	YES	YES	NO	NO/NO
Indiana bat	NA	YES	YES	YES	NO/NO
Northern long-eared bat	NA	YES	YES	YES	NO/NO

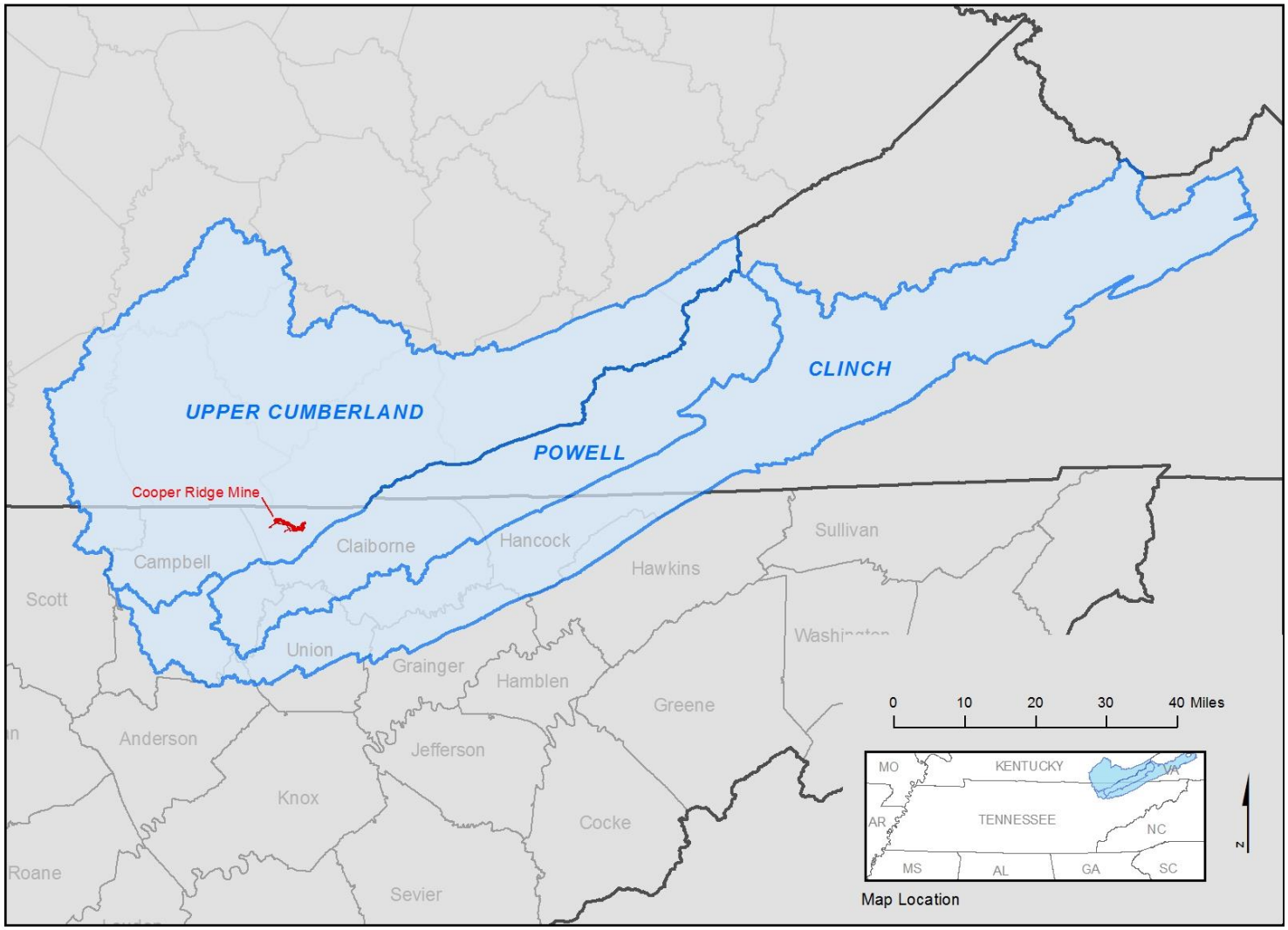

Figure 4. Watersheds associated with species and critical habitat initially considered in OSMRE's assessment of potential impacts from issuing a permit (3270) for surface mining to Kopper Glo Mining, LLC, Cooper Ridge Surface Mine, Claiborne County, Tennessee. See Table 1 for species and critical habitats considered.

slabslide pearlymussel, slender chub, and yellowfin madtom. OSMRE (Office of Surface Mining Reclamation and Enforcement 2016b) also identified potential adverse project effects to blackside dace, gray bat, Indiana bat, and northern long-eared bat (Table 1). They determined that the proposed mining may affect but would likely not adversely affect (NLAA) gray bat, but that the proposed mining would likely adversely affect (LAA) blackside dace, Indiana bat, and northern long-eared bat (Table 1). While, there is no statutory requirement for the Service to concur with a "no effect" determination, we agree with this determination and have no additional comments or concerns regarding these species. The species listed above and identified in Table 1 will not be discussed further in this Biological Opinion.

1.3.2. Summary, species included in this BO

In summary, this BO assesses the potential for the project to adversely affect Indiana bat, northern long-eared bat, and blackside dace.

1.4. Action area

By definition, the project action area encompasses an area where proposed activities can cause measurable or detectable changes in land, air and water or to other measurable factors that may elicit a response in the species or critical habitat addressed under the consultation. The project action area is not limited to the footprint of the action and should consider the chemical and physical impacts to the environment resulting from the action.

The 1,496-acre project boundary of the proposed Kopper Glo Mining, LLC Cooper Ridge Surface Mine is located in Claiborne County, Tennessee (see Figure 1) approximately five miles southwest from the junction of Tennessee State Highway 90 and Valley Creek Road and the Valley Creek community. The proposed mine site is located in the Clear Fork watershed in the Upper Cumberland River system and is drained locally by Clear Fork tributaries including: Valley Creek, Nolan Branch, Spar Branch, and Hurricane Creek, and various smaller unnamed tributaries of these streams (Figure 2).

The action area evaluated for this Biological Opinion includes all areas that the Kopper Glo Mining, LLC Cooper Ridge Surface Mine permit (OSMRE 3270) may directly or indirectly affect, including the 1,496 acres within the mine permit boundary where ground-disturbing mining activities will occur on a subset of this acreage, adjacent terrestrial areas around the permit boundary, and streams in the Clear Fork of the upper Cumberland River system 5,180 mi^2 8-digit HUC catalog unit 05130101 (Figure 5).

1.4.1. Action area for Indiana bat and northern long-eared bat

Indirect effects from noise and dust emissions from blasting, mining equipment, and vehicle traffic to may also affect the terrestrial animals (Indiana bat and northern long-eared bat) included in this BO. Therefore, the action area evaluated for this Biological Opinion includes an additional ½ mile buffer area around the permit boundary. The action area for bats includes the entirety of the mine permit boundary and a half-mile "buffer" around the mine permit boundary. This terrestrial action area polygon is identified on Figure 5.

1.4.2. Action area for blackside dace

The potential for indirect water quality impacts from sediment deposition, Total Dissolved Solids (TDS), and increases in dissolved ions (or salinity) that could result in elevated specific conductance downstream of the permit footprint, the action area also includes the Clear Fork of the Cumberland River, including the headwaters of Valley Creek, Hurricane Creek, Nolan Branch, and Clear Fork the mainstem, extending downstream in the Clear Fork mainstem downstream as far as the confluence with Tackett Creek (Figure 5).

2. Status of the species/critical habitat

2.1. Indiana bat (*Myotis sodalis*)

The Indiana bat (*Myotis sodalis*) was originally listed as in danger of extinction under the Endangered Species Preservation Act of 1966 (80 Stat. 926; 16 U. S. C. 668aa[c]) on March 11, 1967 (32[48]:4001). It was subsequently listed as endangered under the Act, as amended, which extended full protection to the species. Thirteen winter hibernacula (11 caves and two mines) in six states were designated as critical habitat for the Indiana bat on September 24, 1976 (41 FR 41914), including: Blackball Mine (LaSalle County, Illinois); Big Wyandotte Cave (Crawford County, Indiana); Ray's Cave (Greene County, Indiana); Bat Cave (Carter County, Kentucky); Coach Cave (Edmonson County, Kentucky); Cave 021 (Crawford County, Missouri); caves 009 and 017 (Franklin County, Missouri); Pilot Knob Mine (Iron County, Missouri); Bat Cave (Shannon County, Missouri); Cave 029 (Washington County, Missouri); White Oak Blowhole Cave (Blount County, Tennessee); and Hellhole Cave (Pendleton County, West Virginia) (U. S. Fish and Wildlife Service 2009a). A recovery plan, addressing the Indiana bat, was approved on October 14, 1983 (U. S. Fish and Wildlife Service 1983); the first revision of a draft recovery plan was issued on April 13, 2007 (U. S. Fish and Wildlife Service 2007).

2.1.1. Species description

The Indiana bat was first described as a distinct species based on museum specimens collected in 1904 from Wyandotte Cave in Crawford County, Indiana. Prior to that, specimens of the Indiana bat were often confused with those of other Myotis species, particularly the little brown bat (*Myotis lucifugus*) (Miller and Allen 1928). The Indiana bat is monotypic, indicating that there are no recognized subspecies.

The Indiana bat is a medium-size bat, having a wing span of 9 -11 inches and weighing only 0.25 oz. Its forearm length is 1.4 to 1.6 in, and its overall body length ranges from 1.6 to 1.9 in. It has brown to dark-brown fur, and the facial area often has a pinkish appearance. The species closely resembles the little brown bat and the northern long-eared bat. The northern long-eared bat is separated easily from the other two species by its long, pointed, symmetrical tragus (fleshy projection which covers the opening of the ear). The Indiana bat usually has a distinctly keeled calcar (spur-like projection on each wing), whereas the little brown bat does not (Barbour and Davis 1969).

Consultation Action Area

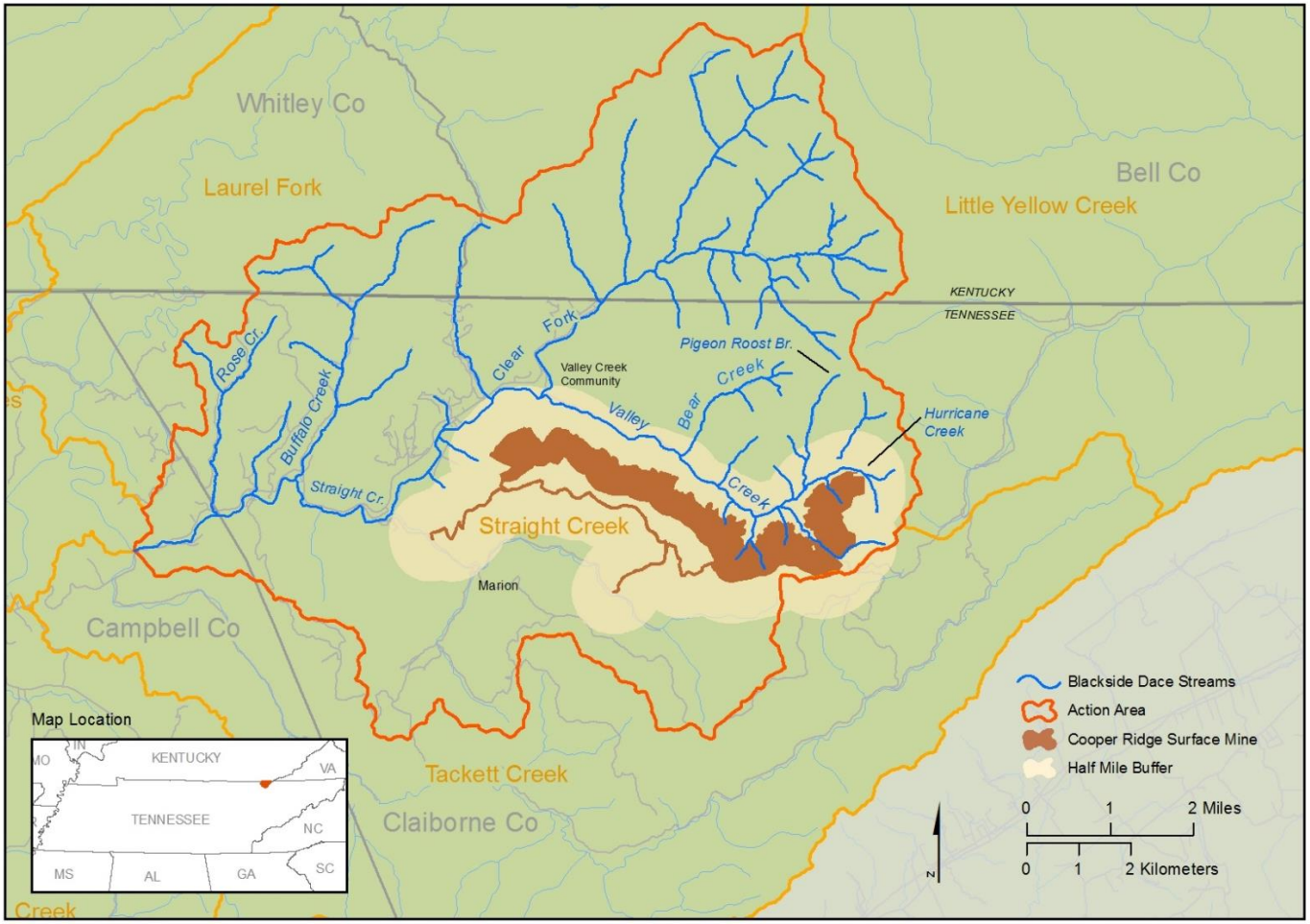

Figure 5. Action area within which the potential for direct and indirect effects to blackside dace, Indiana bat, and northern long-eared bat were analyzed for this consultation resulting from activities associated with Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270), Claiborne County, Tennessee. Action area for blackside dace includes streams within the orange polygon. Action area for Indiana and northern long-eared bats includes the mine permit boundary and the half-mile buffer around the permit boundary.

2.1.2. Life history

2.1.2.1. Life span

The average life span of the Indiana bat is five to ten years, but capture of banded individuals has verified that they can live as long as 14 to 15 years (Thomson 1982). Survivorship of females in an Indiana population was 76 percent (%) for Indiana bats from ages up to six years old and 66% for Indiana bats over six years of age, but younger than ten years. Male survivorship was 70% for ages one through six and 36% for ages six to ten (Humphrey and Cope 1977).

2.1.2.2. Diet

Indiana bats feed exclusively on flying aquatic and terrestrial insects. Moths (Lepidoptera), beetles (Coleoptera), and midges and flies (Diptera) constitute the bulk of the Indiana bat's diet (Brack and LaVal 1985). Moths have been identified as major prey items that may be preferentially selected (Brack and LaVal 1985), but beetles and flies were also found to be significant (Brack and Tyrell 1990). Aquatic insect prey is most often midges and other species that congregate over water, but are seldom mosquitoes (Diptera). Other prey include wasps and flying ants (Hymenoptera), caddisflies (Trichoptera), brown leafhoppers and treehoppers (Homoptera), stoneflies (Plecoptera) and lacewings (Neuroptera) (Brack and LaVal 1985, U. S. Fish and Wildlife Service 2007).

Diet varies seasonally and variations exist among different ages, sexes and reproductive status (U. S. Fish and Wildlife Service 2007). It is probable that Indiana bats use a combination of both selective and opportunistic feeding to their advantage (Brack and LaVal 1985). Reproductively active females and juveniles show greater dietary diversity, perhaps due to higher energy demands. Studies in some areas have found that reproductively active females eat more aquatic insects than juveniles or adult males (U. S. Fish and Wildlife Service 2007), but this may be the result of habitat differences (Brack and LaVal 1985). Male Indiana bats summering in or near a hibernation cave primarily eat moths and beetles, feeding on other terrestrial insects in lower percentages (U. S. Fish and Wildlife Service 2007).

2.1.2.3. Staging, spring migration and summer roosting

The period following winter hibernation, but prior to spring migration to summer habitat, is typically referred to as "staging". During this short time period, bats begin to gradually emerge from hibernation, exit the hibernacula to feed, but re-enter the same or alternative hibernacula to resume daily bouts of torpor (Whitaker and Hamilton 1998). Variation in timing (onset and duration) of staging for Indiana bats was based on latitude and weather (U. S. Fish and Wildlife Service 2007); similarly, timing of staging for northern long-eared bats is likely based on these same factors.

Staging for Indiana bats occurs in late March or early April; following hibernation, most Indiana bats emerge and forage for a few days or weeks near their hibernaculum before migrating to their traditional summer roosting areas. Female Indiana bats emerge first from hibernation, followed

by males. Most individuals have completely left their hibernacula by late April. However, the timing of annual emergence may vary depending upon latitude and annual weather conditions across the species' range. Shortly following emergence, the females become pregnant via delayed fertilization from sperm stored in their reproductive tracts throughout the winter (U. S. Fish and Wildlife Service 2007).

Spring migration to summer roosting areas is stressful for Indiana bats due to fat reserves and food supplies being low. As a result, adult mortality may be highest during late March and April. In the past, Indiana bats were thought to generally migrate north for the summer, but recent studies in Tennessee have documented migration in other directions (Gardner and Cook 2002, U. S. Fish and Wildlife Service 2007, Pelren, personal communication, 2014). A stronger homing tendency in tagged Indiana bats has been observed along a north-south axis, than in east-west directions. Male Indiana bats disperse throughout their range and roost individually or in small groups. In contrast, reproductive females form larger groups, referred to as maternity colonies, where they raise offspring (U. S. Fish and Wildlife Service 2007).

Females can arrive at summer roosting areas as early as April 1. Temporary roosts are often used until maternity roosts with large numbers of adult females are established. Indiana bats were found to arrive at maternity roosts in April and early May in Indiana, with substantial numbers being in residence by mid-May. Most documented maternity colonies have 50 to 100 adult bats (U. S. Fish and Wildlife Service 2007).

Fecundity is low in Indiana bats; females produce only one young per year, which is born in late June to early July. Young bats begin to fly between mid-July and mid-August, at about 4 weeks of age. Mortality between birth and weaning (when they begin to fly) has been determined to be about 8% (Humphrey et al. 1977).

2.1.2.4. Fall migration, swarming, mating and hibernation

The later part of the summer is spent accumulating fat reserves for fall migration (U. S. Fish and Wildlife Service 2007). Most Indiana bats arrive at their traditional hibernacula in August or September and begin to swarm; although, some males may begin to arrive at hibernacula as early as July. Females typically arrive later, and by September, the number of males and females is nearly equal. Swarming assists with mating and foraging until sufficient fat reserves have been deposited to sustain the bats throughout the winter (Cope and Humphrey 1977, U. S. Fish and Wildlife Service 1983). During swarming, most bats will continue to roost in trees during day light hours. Body weight may increase by 2 grams (g [approximately 0.7 oz]) within a short amount of time during this period, primarily in the form of fat. Swarming continues for several weeks, and copulation occurs on cave ceilings, near cave entrances, during the latter part of the period (U. S. Fish and Wildlife Service 2007). In Indiana and Kentucky, the time of highest swarming activity has been documented as early September (Cope and Humphrey 1977). Most females enter hibernation by late September, but males may continue swarming well into October, in what is believed to be an attempt to breed with late arriving females.

Tracking of Indiana bats during the swarming period has indicated that the majority forage within two to three miles of the hibernacula; however, some have been found up to five miles or

further from hibernacula (Rommé et al. 2002). Therefore, not only is it important to protect caves where the bats hibernate, but also to maintain and protect the quality and quantity of roosting and foraging habitat within at least five miles of each Indiana bat hibernaculum. Additional studies of fall swarming behavior are warranted to gain a better understanding of Indiana bat behavior and habitat needs during this part of their annual life cycle (Rommé et al. 2002).

Female Indiana bats may mate during their first autumn, whereas males may not sexually mature until the second year (U. S. Fish and Wildlife Service 2007). Females enter into hibernation shortly after mating. Most Indiana bats are in hibernation by the end of November when prey (i.e., insects) are no longer available; however, hibernacula populations may continue to increase (U. S. Fish and Wildlife Service 2007). Indiana bats cluster and hibernate on cave ceilings in densities of approximately 300-485 bats/ft^2 from approximately October through April. However, the hibernation season may vary by latitude and annual weather conditions. Clustering may protect individuals from temperature change and reduce their sensitivity to disturbance. As with other cave bat species, the Indiana bat naturally arouses during hibernation (Sealander and Heidt 1990). Arousals are more frequent and periods of activity last longer at the beginning and end of the hibernation period (Sealander and Heidt 1990). Limited mating occurs throughout the winter and into early April as bats emerge (U. S. Fish and Wildlife Service 2007).

2.1.3. Habitat characteristics and use

2.1.3.1. Winter hibernacula habitat

Indiana bats roost in caves or mines with configurations that provide suitable temperatures and humid microclimates (Brack et al. 2003, U. S. Fish and Wildlife Service 2007). In caves that provide appropriate temperatures and microclimates for Indiana bats, roosts are usually located near the cave entrances. Required cave and mine temperatures in October and November need to be approximately 50 degrees Fahrenheit (ºF) or less for Indiana bats at their time of arrival at hibernacula (U. S. Fish and Wildlife Service 2007), and mid-winter temperatures in these caves and mines should range from 39 to 46º F (U. S. Fish and Wildlife Service 1983). Only a small percentage of caves and mines meet these temperature requirements (Brack et al. 2003, U. S. Fish and Wildlife Service 2007). Such temperatures allow bats to maintain low metabolic rates and conserve fat reserves to survive the winter (U. S. Fish and Wildlife Service 2007). Relative humidity of roosts usually ranges from around 74% to just below saturation (although readings as low as 54% have been recorded) and may be an important factor for successful hibernation (U. S. Fish and Wildlife Service 2007). Hibernacula often contain large assemblages of several species of bats. Other bat species found in Indiana bat hibernacula include little brown bats, northern long-eared bats, eastern pipistrelles (*Pipistrellus subflavus*), gray bats, big brown bats (*Eptesicus fuscus*) and silver-haired bats (*Lasionycteris noctivagans*) (Brack et al. 2003).

2.1.3.2. Summer roosting habitat, female

Female Indiana bats generally migrate northward from hibernacula to summer roosting areas. However, recent telemetry work with Indiana bats has discovered that some migrate in other directions (southward, etc.) (Pelren 2014). Indiana bats exhibit strong site fidelity to their

traditional summer colony areas and foraging habitat and annually return to the same sites in the summer to bear their young (Garner and Gardner 1992, Kurta et al. 2002, U. S. Fish and Wildlife Service 2007). Traditional summer sites that maintain a variety of suitable roosts are essential to the reproductive success of local populations. It is not known how long or how far female Indiana bats will search to find new roosting habitat if their traditional roosting habitat is lost or degraded during winter months. If they are required to search for new roosting habitat in the spring, it is assumed that this effort places additional stress on pregnant females at a time when fat reserves are low or depleted, and they are already stressed from the energy demands of migration and pregnancy.

Summer roosting and foraging habitat for the Indiana bat is often in floodplain or riparian forests, but may also occur in upland areas. Indiana bat maternity colonies have often been found within forests that are streamside ecosystems. A telemetry study in Illinois found most maternity roosts within 1,640 feet of a perennial or intermittent stream (Hofmann 1996). In the Southern Appalachians, the primary roosts of Indiana bats are on the upper portions of south-facing slopes (O'Keefe 2013).

The Indiana bat may also persist in highly altered and fragmented forest landscapes. Instances have been documented of bats using forests altered by grazing, swine feedlots, row-crops, hay fields, residential developments, clearcut timber harvests and shelterwood cuts (Garner and Gardner 1992). Roosts have been found near lightly traveled, low maintenance roads, as well as higher disturbances areas, such as the Indianapolis Airport, Indiana, in the vicinity of Interstate 70 (I-70). These instances may indicate that Indiana bats might be more adaptable than previously thought, but it still is not known how a maternity colony's stability and reproductive success responds to increasing levels of habitat alteration and fragmentation.

Indiana bat maternity colonies typically occupy multiple roosts in riparian, bottomland and upland forests. Suitability of trees for roosting is determined by condition (dead or alive), loose bark, solar exposure, spatial relationship to other trees, and spatial relationship to water sources and foraging areas. Roost trees generally have exfoliating bark (which allows the bat to roost between the bark and bole of the tree), a southeast or south-southwest solar exposure and an open canopy. Indiana bats in the Southern Appalachians roost under the sloughing bark of tall, decaying, large conifer snags with good solar exposure on south-facing slopes (O'Keefe 2013). Tree cavities, hollow portions of tree boles or limbs, and crevice and splits from broken tops are also used as roosts on a very limited basis, usually by individual bats. Roost trees are often located on forest edges or openings with open canopy and open understory (U. S. Fish and Wildlife Service 2007).

A variety of trees are used by Indiana bats for roosts, including both conifers and hardwoods with defoliating bark. Many maternity colonies have been associated with oak (*Quercus* spp.) - hickory (*Carya* spp.) and elm (*Ulmus* spp.) - ash (*Fraxinus* spp.) - cottonwood (*Populus* spp.) forest types. A landscape-scale study in the Southern Appalachians found that optimal Indiana bat summer roosting habitat occurred near ridgetops in a south-facing, mixed pine forest at elevations ranging from 853 to 2,297 feet (Hammond 2013). Roost tree structure is probably more important than the tree species in determining whether a tree is a suitable roost site; trees which develop loose, exfoliating bark as they age and die are likely to provide roost sites.

Indiana bat roosts are transient and frequently associated with dead or dying trees. Roost longevity is variable due to many factors such as the bark sloughing off or the tree falling down. Some roosts may only be habitable for one to two years, but species with good bark retention, such as slippery elm (*Ulmus rubra*), eastern cottonwood (*Populus deltoides*), green ash (*Fraxinus pennsylvanica*), oaks and hickories may provide habitat for four to eight years. In the Southern Appalachians, heavily decayed yellow pines (*Pinus* spp.) were the most abundant type of snag on the landscape, although most yellow pine snags were too decayed to provide suitable roosts for Indiana bats (O'Keefe 2013). Gardner et al. (1991) evaluated 39 roost trees and found that 31% were no longer suitable the following summer, and 33% of those remaining were unavailable by the second summer.

A variety of suitable roosts are needed within a colony's traditional summer range for the colony to continue to exist. Indiana bat maternity sites generally consist of one or more primary maternity roost trees, which are used repeatedly by large numbers of bats, and varying numbers of alternate roosts, which may be used less frequently and by smaller numbers of bats. Primary roosts are often located in openings or at the edge of forests, while alternate roosts can be in either openings or the interior of forests. Primary roosts are usually surrounded by open canopy and are warmed by solar radiation. Alternate roosts may be used when temperatures are above normal or when it rains. Shagbark hickories (*Carya ovata*) provide good alternate roosts because they are cooler during periods of high heat, and their tight bark shields bats from precipitation (U. S. Fish and Wildlife Service 2007). Bats move among roosts within a season and when a particular roost becomes unavailable from one year to the next. It is not known how many alternate roosts must be available to assure retention of a colony within a particular area, but large, nearby, forest tracts improve the potential for an area to provide adequate roosting habitat (Callahan 1993, Callahan et al. 1997). Trees in excess of 15.7 inches DBH are considered optimal for maternity colonies (U. S. Fish and Wildlife Service 2002). Trees in excess of 8.6 inches DBH are used as alternate roosts by Indiana bats (U. S. Fish and Wildlife Service 2002). However, females have also been documented using roost trees as small as 5.5 inches (Kurta 2005).

Weather has a profound influence on bat behavior and habitat use (Humphrey et al. 1977). Exposure of trees to sunlight and location, relative to other trees, are important to Indiana bat roosting suitability. Cool temperatures can delay fetal and juvenile development, and selection of appropriate maternity roost sites may be critical to reproductive success. Dead trees with southeast and south-southwest exposures allow warming solar radiation. Some living trees may provide a thermal advantage during cold periods (U. S. Fish and Wildlife Service 2007). Therefore, maternity colonies use multiple roosts in groups that contain both living and dead trees. Extent and configuration of a use area is probably determined by availability of suitable roost sites. Distances between roosts can vary from a few yards up to a few miles. Maternity colony movements among multiple roosts seem to depend on climatic changes, particularly solar radiation (Humphrey et al. 1977). Kurta et al. (1993) suggested Indiana bat movement between roosts may be because of the temporary nature of a roost site due to loose bark (which eventually breaks loose from the tree). Presumably, bats that are aware of alternate roost sites are more likely to survive sudden, unpredictable destruction of their roosts, than bats that have not identified alternative sites.

Humphrey et al. (1977) observed that each night after sunset (the peak period of foraging activity), Indiana bats left their foraging areas without returning to their day roosts, which indicated the use of separate "night" roosts. Kiser et al. (2002) found three concrete bridges on Camp Atterbury, 25 miles south of Indianapolis, Indiana, used by Indiana bats as night roosts and, to a limited extent, as day roosts. Other bat species using the bridges included the big brown bat, northern long-eared bat, little brown bat and eastern pipistrelle. The Indiana bat was the most common species found roosting under the bridges, representing 51% of all bats observed, and the big brown bat was the second most abundant at 38% observed. The clusters of Indiana bats, observed night-roosting under the bridges, were lactating, post-lactating and newly volant juveniles. The bridges were comprised of concrete-girders (multi-beams) with deep, narrow expansion joints, and ranged from 46 to 223 feet in length and 26 to 39 feet in width. Average daily traffic ranged from less than ten vehicles/day to almost 5,000 vehicles/day. All the bridges were located over streams bordered by forested, riparian corridors that connected larger tracts of forest. The riparian vegetation did not shade the bridges, allowing solar input to warm them; however, trees were present within 9 to 16.5 feet of each bridge. Bat clusters under the bridges were located over terrestrial areas (on shore) near the end of the bridge spans, rather than over active flow channels. The bridges apparently acted as thermal sinks; mean ambient temperatures at night were consistently higher and less variable under the bridges than external ambient temperatures. The warmer, more stable environments presumably decreased the energetic cost of maintaining high body temperatures, thus promoting fetal development, milk production and juvenile growth. Three individuals were radio-tracked to their day roosts within 0.6- to 1.2 miles of their night roosts (Kiser et al. 2002).

2.1.3.3. Summer roosting habitat, male

Many male Indiana bats appear to remain at or near the hibernacula in summer with some fanning out in a broad band around the hibernacula (Whitaker and Brack 2002). Males roost singly or in small groups in two to five roost trees, similar to those used by females. Males may occasionally roost in caves during summer. Suitable roost trees typically have a large diameter, exfoliating bark and prolonged solar exposure, with no apparent importance in regard to the tree species or whether it is found in upland or bottomland forests (Whitaker and Brack 2002). However, because males typically roost individually or in small groups, the average size of their roost trees tends to be smaller than the roost trees used by female maternity colonies; males have been observed roosting in trees as small as 2.5 inches DBH (Gumbert et al. 2002, U. S. Fish and Wildlife Service 2007). Males have shown summer site fidelity and have been recaptured in the same foraging areas as they had used in prior years (U. S. Fish and Wildlife Service 2007). At Camp Atterbury in Indiana, male bats were observed using the same bridges as females for night roosts, but they roosted singly (Kiser et al. 2002).

2.1.3.4. Autumn swarming/spring staging habitat

Spring and autumn habitat use is variable due to the proximity and quantity of roosts, weather conditions and prey availability. Several studies support the idea that during autumn and spring, Indiana bats primarily use habitat within five miles of their hibernacula (Rommé et al. 2002). However, due to low sample sizes and difficulties with telemetry research techniques, additional

studies of fall and spring Indiana bat habitat use are warranted before this concept can be accepted (U. S. Fish and Wildlife Service 2007).

Indiana bats use roosts in spring and fall that are similar to those used in summer. During fall, when they swarm and mate at their hibernacula, the males roost in trees nearby during the day, when roost trees are often exposed to sunshine, and fly to caves during the night. Studies have found Indiana bat males roosting in dead trees on upper slopes and ridgetops within a few miles of the hibernacula (U. S. Fish and Wildlife Service 2007). Research conducted in Jackson County, Kentucky, found that fall roost trees tend to be located in canopy gaps created by disturbance (logging, windthrow [trees uprooted or broken by wind], prescribed burning, etc.) and along edges (Gumbert et al. 2002). Within-year Indiana bat fidelity to fall roosts has been observed, where an individual bat uses an individual roost for an average of two to three days before moving to a new tree (Gumbert et al. 2002).

2.1.3.5. Foraging habitat

Indiana bats forage between dusk and dawn and feed exclusively on flying insects, as mentioned under *Diet* (section 2.1.2.2). They typically forage in and around tree canopies and within floodplain, riparian and upland forest openings (U. S. Fish and Wildlife Service 2007). Ideal foraging habitat would have 50% to 70% canopy closure (Rommé et al. 1995), with relatively open understory (typically, less than 40% of the trees are 2 to 4.7 inches DBH) (Rommé et al. 1995). Excellent foraging habitat has been characterized as a strip of woody vegetation at least 100 feet wide along a stream.

Brack and Tyrell (1990) found that in early summer, foraging was restricted to riparian habitats. Foraging also occurs over clearings with successional vegetation, along cropland borders and fencerows, and over farm ponds. Indiana bats have routinely been documented flying at least 1.25 miles from their roosts to forage, and some have been tracked up to three miles from their roosts (U. S. Fish and Wildlife Service 2002). Foraging Indiana bats generally fly 6–100 feet above the ground (U. S. Fish and Wildlife Service 2007).

Female Indiana bats typically utilize larger foraging ranges than males (Garner and Gardner 1992). A study in Illinois found that streams associated with floodplain forests and impounded water bodies were preferred foraging habitats for pregnant and lactating Indiana bats, some of which flew up to 1.5 miles from upland roosts; the maximum distance that any female bat, regardless of reproductive status, flew from her daytime roost to a capture site was 2.5 miles (Gardner et al. 1991).

2.1.4. Population dynamics/status distribution

The Service compiles winter hibernacula survey data bi-annually from odd calendar years to determine the most current rangewide population estimates. The 2015 rangewide population estimate is 523,636 Indiana bats (U. S. Fish and Wildlife Service 2015c).

Because the vast majority of Indiana bats form dense aggregations or “clusters” on the ceilings of a relatively small number of hibernacula (i.e., caves and mines) each winter, conducting

standardized surveys of the hibernating bats is the most feasible and efficient means of estimating and tracking population and distribution trends across the species' range. Collectively, winter hibernacula surveys provide the Service with the best representation of the overall population status and relative distribution (U. S. Fish and Wildlife Service 2012).

As will be discussed further, information specific to the "reproductive unit" (i.e., maternity colony) of the Indiana bat is limited. While winter distribution of the Indiana bat is well documented, relatively little is known about location, number and sizes of maternity colonies. As described below, the locations of more than 90% of the estimated maternity colonies remain unknown. Additionally, the relationship between wintering populations and summering populations is not clearly understood. For example, while it is known that individuals of a particular maternity colony typically originate from one to many different hibernacula, the source hibernacula of individuals in a maternity colony is not usually known. The county distribution of hibernacula appears to be better represented and more complete than that of the species' summer distribution (U. S. Fish and Wildlife Service 2012).

There is limited information on the historic distribution and abundance of Indiana bats. However, paleontological evidence suggests that prehistoric abundance of Indiana bats may have exceeded our recent population estimates by an order of magnitude. A summary of prehistoric and historic distribution and abundance can be found in the revised draft recovery plan (U. S. Fish and Wildlife Service 2007).

There was a declining trend in Indiana bat population size between 1981 and 2001. The declining trend in population size from 1981 to 2001 was reversed between 2003 and 2007, with an estimated 635,349 Indiana bats in 2007 (U. S. Fish and Wildlife Service 2015e). However, since 2007, Indiana bats have declined overall by 17.6 % in all four recovery units (Ozark-Central Recovery, Midwest, Appalachia, and Northeast), some of which is attributed to white-nose syndrome (WNS) (discussed in Section 2.1.5).

2.1.4.1. Categorization of hibernacula

The revised draft recovery plan (U. S. Fish and Wildlife Service 2007), assigned Indiana bat hibernacula priority numbers based on winter population sizes and the importance of these hibernacula in the context of the species' range. Priority numbers are defined below:

Priority 1 (P1) – Essential to recovery and long-term conservation of the Indiana bat. *P1* hibernacula typically have (1) a current and/or historically observed winter population greater or equal to 10,000 Indiana bats and (2) currently have suitable and stable microclimates. *P1* hibernacula are further divided into one of two subcategories, "A" or "B," depending on their recent population sizes. Priority 1A (*P1A*) hibernacula are those that have held 5,000 or more Indiana bats during one or more winter surveys conducted during the past 10 years. In contrast, Priority 1B (*P1B*) hibernacula are those that have sheltered greater than or equal to 10,000 Indiana bats at some point in their past, but have consistently contained fewer than 5,000 bats over the past 10 years.

Priority 2 (P2) – Contributes to recovery and long-term conservation of the Indiana bat. *P2* hibernacula have a current or observed historic population of more than 1,000 Indiana bats, but less than 10,000 and an appropriate microclimate.

Priority 3 (P3) – Contributes less to recovery and long-term conservation of the Indiana bat. *P3* hibernacula have current or observed historic populations of 50 to 1,000 Indiana bats.

Priority 4 (P4) – Least important to recovery and long-term conservation of the Indiana bat. *P4* hibernacula typically have current or observed historic populations of fewer than 50 Indiana bats.

2.1.4.2. Current winter distribution

There are approximately 467 known hibernacula located in 19 states (King 2013). There are total of 16 *P1A* hibernacula; 8 *P1B* hibernacula; 35 *P2A* hibernacula; 20 *P2B* hibernacula; 154 *P3* hibernacula; and 234 *P4* hibernacula. Winter surveys in 2014-2015 found hibernating Indiana bats dispersed across 16 states. However, 94% of the estimated rangewide population hibernated in four states, Indiana (35%), Missouri (35%), Kentucky (13%) and Illinois (11%) (U. S. Fish and Wildlife Service 2015c).

2.1.4.3. Current summer distribution

The summer distribution of Indiana bats covers a broader geographic area than their winter distribution. Most of the known summer occurrences are from the upper Midwest, including southern Iowa, northern Missouri, much of Illinois and Indiana, southern Michigan, Wisconsin, western Ohio and Kentucky. In the past decade, many summer maternity colonies have been found in the northeastern states of Pennsylvania, Vermont, New Jersey, New York, West Virginia and Maryland. Maternity colonies extend south to northern Arkansas, southeastern Tennessee and southwestern North Carolina (Britzke 2003, U. S. Fish and Wildlife Service 2007) and have been found as far south as northern Mississippi, Alabama and Georgia. Summer occurrences have also been documented in eastern Oklahoma (U. S. Fish and Wildlife Service 2012, Pelren, personal communication, 2014).

Male Indiana bats are found throughout the range of the species, but in summer are most common in areas near hibernacula (Gardner and Cook 2002). Because they typically roost solitarily in the summer, they are less likely to be detected by mist-netting than adult females, which tend to occur in high-density maternity colonies.

2.1.4.4. Maternity colonies

The first documented Indiana bat maternity colony, located in east-central Indiana, was not discovered until 1971 (Cope et al. 1974). When the revised draft recovery plan was completed in 2007 (U. S. Fish and Wildlife Service 2007), 269 maternity colonies in 16 states were considered extant. Of those 269 colonies, 54% (146) of them had been found within ten years (i.e., since 1997), primarily through mist-netting surveys. Several maternity colonies have been located in northeastern and southeastern states via radio-telemetry, as females have been tracked from hibernacula to summer habitat. Because Indiana bat maternity colonies are widely

dispersed during the summer and difficult to locate, it is presumed that all of the summer survey efforts have found only a small fraction of the maternity colonies that likely exist (U. S. Fish and Wildlife Service 2012).

The total number of maternity colonies that exist rangewide is not known, but can be estimated based on population estimates derived from winter hibernacula surveys. Based on a rangewide population estimate of 467,947 Indiana bats, and assuming a 50:50 sex ratio and average maternity colony size of 50 to 80 adult females (Whitaker and Brack 2002), there were an estimated 3,802 (± 877) maternity colonies in 2007. Using the same set of assumptions, there were an estimated 3,450 (± 797) Indiana bat colonies in 2011, representing a loss of about 352 colonies over that two-year period. However, this simple mathematical approach fails to incorporate regional variations in the decline, the effects of WNS to Indiana bats, and the social structure of maternity colonies. A decline in hibernating Indiana bat populations due to WNS may manifest itself first as a reduction in the size of maternity colonies, then the loss of whole colonies if the number of surviving colony members is too small to allow the colony to persist. In areas where WNS is just becoming evident, it is likely that Indiana bat maternity colonies will be affected, at least initially, only by the loss of members from WNS-affected hibernacula. However, in areas where WNS has affected Indiana bat populations for multiple years, resulting in very high mortality rates, entire maternity colonies have probably been eliminated because all of the hibernating populations that supported those colonies have been decimated. If the resulting reduction in colony size is substantial, the colony may collapse because so few females remain to form the social clustering that is characteristic of the species and likely contributes to its survival and successful recruitment of young. Regardless of how numbers of maternity colonies are estimated, declining hibernating Indiana bat populations translate to declining summer populations (U. S. Fish and Wildlife Service 2012).

2.1.5. Threats

From 1965-2001, there was an overall decline in Indiana bat populations, with winter habitat modifications having been linked to changes in populations at some of the most important hibernacula (U. S. Fish and Wildlife Service 2007). Most of those modifications were human-induced and involved either commercialization of caves, control of access to caves or were related to mining operations. Improper gating and other structures have rendered many historical hibernacula unavailable to Indiana bats. Other documented threats, involving human disturbances of hibernacula, have included vandalism, flooding of caves during reservoir creation, destruction from quarrying limestone, and indiscriminate collecting, handling and/or banding of hibernating bats. Natural alterations of hibernacula include flooding, entrance and passage collapses, and blocked sinkholes, all of which can alter the temperature regime within a cave, and potentially prevent entry by Indiana bats. Both natural and human-induced changes to hibernacula can alter the climate required by Indiana bats, in turn, adversely affecting populations.

Although it is difficult to quantify the resultant impacts, summer habitat modifications are also suspected in contributing to the decline of Indiana bat populations. Forests used by foraging and roosting Indiana bats during spring, summer and autumn have changed dramatically from pre-settlement conditions; fragmentation of forests (forests converted to agriculture, etc.) and fire

suppression have eliminated or altered many native plant communities. Summer habitat can include extensive forests or small woodlots connected by hedgerows. The removal of such habitats is occurring rapidly in some portions of the Indiana bat's range due to residential and commercial development, mining, oil and gas development, and infrastructure development, including roadways and utility corridors. Even in areas of relatively abundant habitat, permanent and temporary impacts to forest habitat pose mortality risks to Indiana bats during tree felling activities. Furthermore, the ongoing, permanent loss of forests and woodlots may have a significant cumulative effect on the species, as habitat is lost, fragmented and degraded, and as maternity colonies are displaced from habitat to which they exhibit fidelity.

In addition, chemical exposure and contamination to bats outside of hibernacula has been suggested as a cause in the decline of Indiana bats. The degree to which acute or chronic toxicity may be contributing to population declines is still unknown. However, additional research should improve our knowledge of the effects of chemical contaminants on bats.
Due to the species' low reproductive potential, threats that increase mortality or decrease recruitment are of particular concern. In cases where threats have been reduced (i.e., hibernacula have been properly gated to preclude human disturbance), increases in Indiana bat population sizes have been noted. However, any increases in the overall population are expected to be gradual because the species is biologically incapable of responding via an increased reproductive rate (e.g., in response to low population densities or the amelioration of threats.

Indiana bats also appear to be vulnerable to collisions with fast-moving, man-made objects, such as wind turbines and vehicles (Arnett et al. 2008, Russell et al. 2009). While such fatalities seem counterintuitive due to the species' ability to echo-locate, it appears bats are not always able to detect and evade objects that are moving at a high rate of speed. Indiana bat and little brown bat fatalities have been documented along a heavily-traveled highway in central Pennsylvania and at wind farms in Indiana, West Virginia, Ohio and Pennsylvania (Good et al. 2011). Collision-related fatalities probably far exceed those documented, as there is an extensive road network within the Indiana bat's range, as well as hundreds of operating turbines, presenting a perpetual risk to bats. Furthermore, dead bats are not likely to be found or documented because Indiana bat carcasses are small, cryptic and readily scavenged, and few mortality surveys are conducted. Collision-related fatalities are typically considered an additive source of mortality.

WNS is a malady of unknown origin that is killing cave-dwelling bats in unprecedented numbers. WNS was first documented at four sites in eastern New York State during the winter of 2006-2007. Data suggests that a recently identified fungus (*Pseudogymnoascus destructans*) (Pd) (previously known as *Geomyces destructans*) is responsible, at least in part, for the impacts and mortality associated with WNS (Blehert et al. 2009).

The most obvious symptom of WNS is the presence of a white fungus on the face, wing or tail membranes of many, but not all, affected animals. Behavioral changes are also indicative of WNS affliction, characterized by a general shift of animals from traditional winter roosts to colder areas, or to roosts unusually close to hibernacula entrances. Affected bats are generally unresponsive to human activity in the hibernaculum, and may even fail to arouse from torpor when handled. However, bats at affected sites are regularly observed flying across the mid-winter landscape (when they typically would be expected to be hibernating). Affected animals

appear to be dying as a result of depleted fat reserves, presumably because of this winter activity, and mortalities are first apparent months before bats would be expected to emerge from hibernation.

As of August 4, 2016, WNS has been confirmed in 29 states (Alabama, Arkansas, Connecticut, Delaware, Georgia, Illinois, Indiana, Iowa, Kentucky, Maine, Maryland, Massachusetts, Michigan, Minnesota, Missouri, New Hampshire, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Rhode Island, South Carolina, Tennessee, Vermont, Virginia, Washington, West Virginia and Wisconsin), as well as five Canadian provinces (New Brunswick, Nova Scotia, Ontario, Prince Edward Island and Quebec) (White-Nose Syndrome.org 2015a). Also, the fungus that causes WNS, Pd, has been confirmed in three additional states (Mississippi, Nebraska and Oklahoma) (White-Nose Syndrome.org 2015a). The number of confirmed states changes frequently as WNS contintues to spread.

Research and field observations over the past few years have led to a better understanding of WNS and how it may be transmitted. The temporal presentation of WNS among bats in a single New York cave in 2006 to numerous sites in 26 states and five Canadian providences by 2014 suggests that WNS is spread from bat-to-bat, and very likely, from bat-to-hibernaculum and hibernaculum-to-bat (Hicks et al. 2010). This means of transmission is consistent with the rate of WNS spread observed since 2006, and is based on assumptions from available tracking data for local bat movements and from knowledge of inter- and intra-specific bat contact at spring, summer and fall roosting and staging sites. However, an equally plausible, additional mode of transport for Pd, the likely causative agent for WNS, is by anthropogenic sources. Fungal spores and other microscopic organisms can easily become attached to skin, hair, clothing and equipment with which they come in contact, and it is possible that such elements could remain viable for weeks or months after leaving a subterranean environment. Hard evidence that people are, or have been, responsible for transporting WNS to hibernacula is currently not available. However, the occasionally discontinuous nature of the spread of WNS, especially to sites in West Virginia and Virginia, does suggest that something other than bat-to-bat transmission is responsible. Another piece of supporting evidence for anthropogenic spread is the coincidental observation that many of the affected sites are also popular destinations for recreational users of caves and mines.

Another question regarding the effects of WNS is the degree to which susceptibility may vary by species within and among caves (due to differences in cave microclimates, bat densities, etc.), or if observed symptoms are expressed differentially by species. For example, the New York State Department of Environmental Conservation has reported that observed mortality rates may differ between Indiana bats and little brown bats, even within the same site. While susceptibility may be influenced by cave micro-climate and other factors, varying levels of susceptibility by species have emerged over the past few years. Within a five-state area affected by WNS for multiple years (New York, Pennsylvania, Vermont, Virginia and West Virginia), population monitoring at 42 hibernacula documented a 98% decline in northern long-eared bats, 91% decline in little brown bats, 75% decline in tri-colored bats (*Perimyotis subflavus*), 72% decline in Indiana bats, 41% decline in big brown bats and 12% decline in eastern small-footed bats (*Myotis leibii*) (Turner et al. 2011).

It is unclear how long symptoms take to manifest after exposure to the WNS causative agent(s), but field observations indicate the time lapse between initial detection of the visible fungus and mass mortality of bats ranges from a few weeks to over a year (Turner et al. 2011). Recent captive inoculation trials at the U. S. Geological Survey National Wildlife Health Center have demonstrated bat-to-bat transmission of Pd, and data analyses are currently on-going which will advance knowledge about the disease process. While it appears that bats are highly vulnerable to WNS during their hibernation periods, it is not known to what degree or under what conditions WNS may be spread during other periods, such as during fall swarming and summer communal roosting.

Considering WNS has been affecting hibernating bat populations for the longest in New York (since February 2006), data from that state may provide the best indication of the effects of this disease on bats, including Indiana bats. By 2010, all known Indiana bat hibernacula in New York had been documented with WNS. However, the apparent effects of WNS on Indiana bats varied between affected hibernacula. Some Indiana bat hibernating populations had declined by 92 to 100%, while counts of Indiana bats at other WNS-affected New York hibernacula had declined to a lesser extent (e.g., there had been a 21% decline at the Barton Hill Mine and a 77% decline at Glen Park Cave) (Turner et al. 2011).

The degree to which climate or other environmental factors may influence the spread of WNS, or the severity of its impact on affected bats, is unknown. At this time, there is no concrete evidence of resistance to WNS among survivors, although some affected hibernacula continue to support low numbers of bats five years into WNS exposure, and a few hibernacula have substantially lower mortality levels than most. If current trends for spread and mortality at affected sites continue, WNS threatens to drastically reduce the abundance of many species of hibernating bats in much of North America in what may only be a matter of years. Population modeling indicates a 99% chance of regional extinction of the little brown bat from 2010 through 2026 due to WNS (Frick et al. 2010). The closely related Indiana bat is just as vulnerable to regional extinction (if not more so) due to its smaller rangewide population and social behavior traits that increase the risk of bat-to-bat transmission. The declining mortality rates at some New York hibernacula and the apparent resistance of European *Myotis* species to Pd suggest that some level of resistance may exist or develop within North American *Myotis* species.

In partnership with several other state, Federal, and tribal agencies, the Service developed "A National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-Nose Syndrome in Bats" (https://www.whitenosesyndrome.org/national-plan/white-nose-syndrome-national-plan). Canada has developed a comparable plan, allowing for a broader coordinated response to the disease in both countries. The multi- agency, multi-organization WNS response team, under the U. S. National Plan and in coordination with Canadian partners, has developed and continues to update recommendations, tools, and strategies to slow the spread of WNS, minimize disturbance to hibernating bats, and improve conservation strategies for affected bat species. Some of these include:

- decontamination protocols to prevent human transport of fungal spores;
- cave management strategies and BMPs;

- forestry BMPs;
- nuisance wildlife control operator BMPs for removing bats from homes and businesses;
- transportation and bridge construction BMPs;
- hibernacula microclimate monitoring recommendations;
- wildlife rehabilitator BMPs; and
- a ranking of bat species in need of conservation actions.

The capacity of climate change to result in changes in the range and distribution of wildlife species is recognized, but detailed assessments of how climate change may affect specific species are limited (U. S. Fish and Wildlife Service 2016d). Climate change may affect bats since they are sensitive to changes in temperature, humidity and precipitation (Adams and Hayes 2008). Impacts from climate change may also indirectly affect bats due to changes in food (insects) availability or timing of emergence of insect prey (Sherwin et al. 2013, U. S. Fish and Wildlife Service 2016d), timing of hibernation, frequency and duration of torpor, rate of energy expenditure, reproduction, reproductive cycles and rates of juvenile bat development (Sherwin et al. 2013), all of which may contribute to a shift in suitable habitat. Surface temperature is directly related to cave temperature, so climate change will inevitably affect the suitability of hibernacula (U. S. Fish and Wildlife Service 2016d). Loeb and Winters (2013) modeled potential changes in Indiana bat summer maternity range within the United States; in their model, the area suitable for summer maternity colonies of Indiana bats was forecasted to decline significantly. Climate change has been suggested as a cause of population shift from southern to northern hibernacula in Indiana bats (Clawson 2002). Discerning effects from climate change may be difficult because the spread of WNS is occurring rapidly across the ranges of the Indiana bat, gray bat, and northern long-eared bat.

Although information suggests that climate change may affect Indiana bats, the Service does not have evidence suggesting that climate change in itself has led to population declines; furthermore, discerning effects from climate change may be difficult because the spread of WNS is occurring rapidly across the species' range.

2.1.6. Recovery

The overall population distribution has not changed over the past several years. However, the abundance of Indiana bats has declined significantly, particularly in the Northeast, and the threat to the species from WNS remains at a high level. Recovery efforts are primarily focused on the WNS investigation at this time because this poses a serious threat and immediate threat to the continued existence of the species throughout its range. When we consider the positive trends observed over the last several rangewide hibernacula counts (prior to WNS), along with the newly gathered information on WNS, we have concerns about the status of the species. The Service considers the population trend to be declining, with no expectation of a trend reversal in the foreseeable future (U. S. Fish and Wildlife Service 2015c).

Criteria for recovery included in the revised draft recovery plan (U. S. Fish and Wildlife Service 2007) include the following:

A. Reclassification to Threatened Status
   1. Permanent protection at 80% of all P1 hibernacula in each Recovery Unit, with a minimum of one P1 hibernaculum protected in each unit (In the Northeast and Appalachian Mountain recovery units, 80% protection would translate to 100% protection because these units have one and two P1 hibernacula, respectively.).
   2. A minimum overall population estimate equal to the 2005 population estimate of 457,000.
   3. Documentation using statistically reliable information that indicates important hibernacula within each recovery unit, on average, have positive annual population growth rates and minimal risk of population declines over the next ten-year period.
B. Removal from Endangered Species Act Protection
   1. Protection of a minimum of 50% of *P2* hibernacula in each recovery unit.
   2. A minimum overall population estimate equal to the 2005 population estimate of 457,000.
   3. Documentation using statistically reliable information that shows a positive population growth rate over an additional five sequential survey periods (i.e., ten years).

2.2. Northern long-eared bat (*Myotis septentrionalis*)

The northern long-eared bat (*Myotis septentrionalis*) was listed as a threatened species under a final rule on May 4, 2015 (80 FR 17974). A final rule under the authority of section 4(d) of the Act, providing measures that are necessary and advisable for conservation of the northern long-eared bat, also became effective on May 4, 2015 (80 FR 17974). Designation of critical habitat for the northern long-eared bat was determined not prudent.

2.2.1. Species description

The northern long-eared bat is a medium-sized bat species, weighing an average 5 - 8 g (0.18- to 0.28-oz), with females tending to be slightly larger than males (Caceres and Pybus 1997). Pelage colors include medium to dark brown fur on its back, dark brown, but not black, ears and wing membranes, and tawny to pale-brown fur on the ventral side (Nagorsen and Brigham 1993, Whitaker and Mumford 2009). As indicated by its common name, the northern long-eared bat is distinguished from other *Myotis* species by its large ears, that average 17 mm (0.67-in) (Whitaker and Mumford 2009) and, when laid forward, extend beyond the nose but less than 5 mm (0.20-in) beyond the muzzle (Caceres and Barclay 2000).

2.2.2. Life history

2.2.2.1. Life span

Adult longevity for the northern long-eared bat is estimated to be up to 18.5 years (Hall et al. 1957), with the greatest recorded age of 19 years based on banding records (Kurta 1995). Most

mortality for northern long-eared and many other species of bats occurs during the juvenile stage (Caceres and Pybus 1997).

2.2.2.2. Diet

The northern long-eared bat has a diverse diet including moths, flies, leafhoppers, caddisflies and beetles (Nagorsen and Brigham 1993, Brack and Whitaker 2001, Griffith and Gates 1985), with diet composition differing geographically and seasonally (Brack and Whitaker 2001). Feldhamer et al. (2009) noted close similarities of all *Myotis* diets in southern Illinois, while Griffith and Gates (1985) found significant differences between the diets of northern long-eared bats and little brown bats. The most common insects found in the diets of northern long-eared bats were moths and beetles (Brack and Whitaker 2001, Lee and McCracken 2004, Feldhamer et al. 2009, Dodd et al. 2012), with arachnids also being a common prey item (Feldhamer et al. 2009).

2.2.2.3. Staging, spring migration and summer roosting

Spring staging for the northern long-eared bat is the time period between winter hibernation and spring migration to summer habitat (Whitaker and Hamilton 1998). During this time, bats begin to gradually emerge from hibernation, exit the hibernacula to feed, but re-enter the same or alternative hibernacula to resume daily bouts of torpor (Whitaker and Hamilton 1998). The staging period for the northern long-eared bat is likely short in duration (Whitaker and Hamilton 1998, Caire et al. 1979). In Missouri, Caire et al. (1979) found that northern long-eared bats moved into the staging period in mid-March through early May. Variation in timing (onset and duration) of staging for Indiana bats was based on latitude and weather (U. S. Fish and Wildlife Service 2007); similarly, timing of staging for northern long-eared bats is likely based on these same factors. The spring migration period typically runs from mid-March to mid-May (Easterla 1968, Caire et al. 1979, Whitaker and Mumford 2009). In Michigan, Kurta et al. (1997) determined that by early May, two-thirds of the *Myotis* species, including the northern long-eared bat, had dispersed to summer roosting habitat.

The northern long-eared bat typically occupies summer roosting habitat from mid-May through mid-August each year. Female summer home-range size may range from 19 to 172 hectares (ha) (47 to 425 acres) (Lacki et al. 2007). Owen et al. (2003) estimated average maternal home range size to be 65 ha (161 acres). Northern long-eared bats actively form colonies in the summer (Foster and Kurta 1999) and exhibit fission-fusion behavior (Garroway and Broders 2007), where members frequently coalesce to form a group (fusion), but composition of the group is in flux, with individuals frequently departing to be solitary or to form smaller groups (fission) before returning to the main unit (Barclay and Kurta 2007).

Maternity colonies, consisting of females and young, are generally small, numbering from about 30 (Whitaker and Mumford 2009) to 60 individuals (Caceres and Barclay 2000); however, one group of 100 adult females was observed in Vermilion County, Indiana (Whitaker and Mumford 2009). Maternity colonies in two studies in West Virginia supported a range of 7 to 88 individuals (Owen et al. 2002) and 11 to 65 individuals, respectively, with a mean size of 31 (Menzel et al. 2002). Lacki and Schwierjohann (2001) found that the number of bats within a given roost declined as the summer progressed. Pregnant females formed the largest

aggregations (mean = 26) and post-lactating females formed the smallest aggregations (mean=4). Other studies have also found that the number of individuals roosting together in a given roost typically decreases from pregnancy to post-lactation (Foster and Kurta 1999, Lacki and Schwierjohann 2001, Garroway and Broders 2007, Perry and Thill 2007, Johnson et al. 2012).

Birthing within the colony tends to be synchronous, with the majority of births occurring around the same time (Krochmal and Sparks 2007). Northern long-eared bats generally give birth in late May or early June (Easterla 1968, Caire et al. 1979, Whitaker and Mumford 2009) to a single pup (Barbour and Davis 1969). However, birth may occur as late as July (Whitaker and Mumford 2009); Broders et al. (2006) estimated a parturition date of July 20 in New Brunswick. Lactating and post-lactating females were observed in mid-June in Missouri (Caire et al. 1979), July in New Hampshire and Indiana (Sasse and Pekins 1996, Whitaker and Mumford 2009), and August in Nebraska (Benedict et al. 2000). Juvenile volancy often occurs by 21 days after birth (Kunz 1971, Krochmal and Sparks 2007) and has been documented as early as 18 days after birth (Krochmal and Sparks 2007). Juveniles were captured in late June in Missouri (Caire et al. 1979), early July in Iowa (Sasse and Pekins 1996), and early August in Ohio (Mills 1971).

Northern long-eared bats switch roosts often (Sasse and Pekins 1996), typically every two to three days (Foster and Kurta 1999, Owen et al. 2002, Carter and Feldhamer 2005, Timpone et al. 2010). A 2004 study by Jackson tracked 30 northern long-eared bats over two years and found the mean number of different roost used by each bat to be 8.6 (with a range of 2 to 11). Consequently, they have a need for multiple, suitable roosts to be available within close proximity of each other.

2.2.2.4. Fall migration, swarming, mating and hibernation

Fall migration typically occurs between mid-August and mid-October (80 FR 17987). While the northern long-eared bat is not considered a long-distance migratory species, short regional migratory movements between seasonal habitats (summer roosts and winter hibernacula) have been documented between 56 km (35 miles) and 89 km (55 miles) (Griffin 1945, Caire et al. 1979, Nagorsen and Brigham 1993). Griffin (1940a) reported that a banded male northern long-eared bat had traveled from one hibernaculum in Massachusetts to another in Connecticut over the 2-month period of February to April, a distance of 89 km (55 miles).

Northern long-eared bats have shown a high degree of philopatry (tendency to return to the same location) for a hibernaculum (Pearson 1962), although they may not return to the same hibernaculum in successive seasons (Caceres and Barclay 2000). Banding studies in Ohio, Missouri and Connecticut documented return rates to hibernacula of 5% (Mills 1971), 4.6% (Caire et al. 1979), and 36% (Griffin 1940b), respectively. An experiment showed an individual bat returned to its home cave up to 32 km (20 mi) away after being removed three days prior (Stones and Branick 1969).

The swarming season fills the time between the summer and winter seasons (Lowe 2012), and the purpose of swarming behavior may include: introduction of juveniles to potential hibernacula, copulation and stopping over sites on migratory pathways between summer and winter regions (Kurta et al. 1997, Parsons et al. 2003, Lowe 2012, Randall and Broders 2014).

The swarming season for some species of the genus, *Myotis*, begins shortly after females and young depart maternity colonies (Fenton 1969). During this time, both male and female northern long-eared bats are present at swarming sites (often with other species of bats).

Heightened activity and congregation of transient bats around caves and mines is observed during swarming, followed by increased sexual activity and bouts of torpor prior to winter hibernation (Fenton 1969, Parsons et al. 2003, Davis and Hitchcock 1965). For the northern long-eared bat, the swarming period may occur between July and early October, depending on latitude within the species' range (Hall and Brenner 1968, Fenton 1969, Caire et al. 1979, Kurta et al. 1997, Lowe 2012). The northern long-eared bat may investigate several cave or mine openings during the transient portion of the swarming period, and some individuals may use these areas as temporary daytime roosts or roost in forest habitat adjacent to these sites (Kurta et al. 1997, Lowe 2012). Many of the caves and mines associated with swarming are also used as hibernacula for several species of bats, including the northern long-eared bat (Fenton 1969, Whitaker and Rissler 1992, Kurta et al. 1997, Glover and Altringham 2008, Randall and Broders 2014).

Little is known about northern long-eared bat roost selection outside of caves and mines during the swarming period. Lowe (2012) documented northern long-eared bats in the Northeast roosting in both, coniferous and deciduous trees or stumps as far away as 3 miles (7 km) from the swarming site. Although Lowe (2012) hypothesized that tree roosts used during the fall swarming season would be similar to summer roosts, there was a difference found between summer and fall in the variation in distances bats traveled from the capture site to roost, roost orientation and greater variation of roost types (e.g., roost species, size, decay class) in the fall. Greater variation among roosts during the swarming season may be a result of the variation in energy demands that individual northern long-eared bats exhibit during this time (Barclay and Kurta 2007, Lowe 2012).

Northern long-eared bats hibernate during the winter months to conserve energy from increased thermoregulatory demands and reduced food resources. To increase energy savings, individuals enter a state of torpor, when internal body temperatures approach ambient temperature, metabolic rates are significantly lowered and immune function declines (Thomas et al. 1990, Thomas and Geiser 1997, Bouma et al. 2010).

In general, northern long-eared bats arrive at hibernacula in August or September, enter hibernation in October and November, and emerge from the hibernacula in March or April (Caire et al. 1979, Whitaker and Hamilton 1998, Amelon and Burhans 2006). However, hibernation may begin as early as August (Whitaker and Rissler 1992). In Copperhead Cave (a mine) in west-central Indiana, the majority of northern long-eared bats enter hibernation during October (Whitaker and Mumford 2009). In northern latitudes, such as in upper Michigan's copper-mining district, hibernation may begin as early as late August and continue for eight to nine months (Stones and Fritz 1969, Fitch and Shump 1979).

Studies typically have found, northern long-eared bats were not abundant and composed a small proportion of the total number of bats observed hibernating in a hibernaculum (Barbour and Davis 1969, Mills 1971, Caire et al. 1979, Caceres and Barclay 2000). Although usually

observed in small numbers, the species typically inhabits the same hibernacula with large numbers of other bat species, and occasionally are found in clusters with these other bat species. Other species that commonly occupy the same habitat include little brown bat, big brown bat, eastern small-footed bat (*Myotis leibii*), tri-colored bat (*Perimyotis subflavus)* and Indiana bat (Swanson and Evans 1936, Griffin 1940b, Hitchcock 1949, Stones and Fritz 1969). Whitaker and Mumford (2009), however, infrequently found northern long-eared bats hibernating beside little brown bats, Indiana bats or tri-colored bats. Barbour and Davis (1969) found that the species was rarely recorded in concentrations of more than 100 in a single hibernaculum. Northern long-eared bats have been observed moving among hibernacula throughout the winter, which may further decrease population estimates (Griffin 1940b, Whitaker and Rissler 1992, Caceres and Barclay 2000). Whitaker and Mumford (2009) found that the species flies in and out of some mines and caves in southern Indiana throughout the winter. In particular, the bats were active at Copperhead Cave periodically all winter, with northern long-eared bats being more active than other species (such as little brown bats and tri-colored bats) hibernating in the cave. Though northern long-eared bats fly outside of hibernacula during the winter, they do not feed; therefore, the function of this behavior is not well understood (Whitaker and Hamilton 1998). It has been suggested, however, that such bat activity during winter could be due in part to disturbance by researchers (Whitaker and Mumford 2009).

Northern long-eared bats exhibit significant weight loss during hibernation (80 FR 17987). In southern Illinois, Pearson (1962) found an average weight loss of 20% during hibernation in male northern long-eared bats, with individuals weighing an average of 6.6 g (0.2 oz) prior to January 10, and those collected after that date weighing an average of 5.3 g (0.2 oz). Whitaker and Hamilton (1998) reported a weight loss of 41 to 43% over the hibernation period for northern long-eared bats in Indiana. In eastern Missouri, male northern long-eared bats lost an average of 3 g (0.1 oz), or 36%, during the hibernation period (late October through March), and females lost an average of 2.7 g (0.1 oz), or 31% (Caire et al. 1979).

2.2.3. Habitat characteristics and use

2.2.3.1. Winter hibernacula habitat

Northern long-eared bats predominantly overwinter in hibernacula that include caves and abandoned mines (80 FR 17984). Hibernacula used by northern long-eared bats vary in size from large, with large passages and entrances (Raesly and Gates 1987), to much smaller hibernacula (80 FR 17984). These hibernacula have relatively constant, cooler temperatures (32 to 48° F) (Raesly and Gates 1987, Caceres and Pybus 1997, Brack 2007), with high humidity and no air currents (Fitch and Shump 1979, Van Zyll de Jong 1985, Raesly and Gates 1987, Caceres and Pybus 1997). The sites favored by northern long-eared bats are often in very high humidity areas, to such a large degree that droplets of water are often observed on their fur (Hitchcock 1949, Barbour and Davis 1969). Northern long-eared bats, like eastern small-footed bats and big brown bats, typically prefer cooler and more humid conditions than little brown bats, but are less tolerant of drier conditions than eastern small-footed bats and big brown bats (Hitchcock 1949, Barbour and Davis 1969, Caceres and Pybus 1997).

Northern long-eared bats are typically found roosting in small crevices or cracks in cave or mine walls or ceilings, sometimes with only their noses and ears visible, and thus are easily overlooked during surveys (Griffin 1940b, Barbour and Davis 1969, Caire et al. 1979, Van Zyll de Jong 1985, Caceres and Pybus 1997, Whitaker and Mumford 2009). Caire et al. (1979) and Whitaker and Mumford (2009) commonly observed individuals exiting caves with mud and clay on their fur, also suggesting the bats were roosting in tighter recesses of hibernacula. Additionally, northern long-eared bats have been found hanging in the open, although not as frequently as in cracks and crevices (Barbour and Davis 1969, Whitaker and Mumford 2009). Whitaker and Mumford (2009) observed three northern long-eared bats roosting in the hollow core of stalactites in a small cave in Jennings County, Indiana, in 1968.

To a lesser extent, northern long-eared bats have also been observed over-wintering in other types of habitat that resemble cave or mine hibernacula, including abandoned railroad tunnels, (80 FR 17984). Also, in 1952, three northern long-eared bats were found hibernating near the entrance of a storm sewer in central Minnesota (Goehring 1954). Kurta et al. (1997) found northern long-eared bats hibernating in a hydroelectric dam facility in Michigan. In Massachusetts, northern long-eared bats have been found hibernating in the Sudbury Aqueduct (80 FR 17984). Griffin (1945) found northern long-eared bats in Massachusetts during December in a dry well and commented that these bats may regularly hibernate in ‘‘unsuspected retreats’’ in areas where caves or mines are not present. Although confamilial (belonging to the same taxonomic family) bat species (e.g., big brown bats) have been found using non-cave or mine hibernacula, including attics and hollow trees (Neubaum et al. 2006, Whitaker and Gummer 1992), to date, northern long-eared bats have only been observed over-wintering in suitable caves, mines or habitat with the same types of conditions as found in suitable caves or mines. Anecdotal reports indicate there may be other landscape features being used by northern long-eared bats during the winter that have yet to be formally documented.

2.2.3.2. Summer roosting habitat

During summer, northern long-eared bats roost singly or in colonies underneath bark or in cavities, crevices or hollows of both, live and dead trees and/or snags (Sasse and Pekins 1996, Foster and Kurta 1999, Owen et al. 2002, Carter and Feldhamer 2005, Perry and Thill 2007, Timpone et al. 2010). Males’ and non-reproductive females’ summer roost sites may also include cooler locations, including caves and mines (Barbour and Davis 1969, Amelon and Burhans 2006). Northern long-eared bats have also been observed roosting in colonies in human-made structures, such as in buildings, in barns, on utility poles, behind window shutters and in bat houses (Mumford and Cope 1964, Barbour and Davis 1969, Cope and Humphrey 1977, Burke 1999, Sparks et al. 2004, Amelon and Burhans 2006, Whitaker and Mumford 2009, Timpone et al. 2010, Bohrman and Fecske 2013, 80 FR 17984).

The northern long-eared bat appears to be somewhat flexible in tree roost selection, selecting varying roost tree species and types of roosts throughout its range. Northern long-eared bats have been documented in roost in many species of trees, including: black oak (*Quercus velutina*), northern red oak (*Quercus rubra*), silver maple (*Acer saccharinum*), black locust (*Robinia pseudoacacia*), American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), sourwood (*Oxydendrum arboreum*) and shortleaf pine (*Pinus echinata*) (Mumford and Cope 1964, Clark et

al. 1987, Sasse and Pekins 1996, Foster and Kurta 1999, Lacki and Schwierjohann 2001, Owen et al. 2002, Carter and Feldhamer 2005, Perry and Thill 2007, Timpone et al. 2010). Northern long-eared bats most likely are not dependent on certain species of trees for roosts throughout their range; rather, many tree species that form suitable cavities or retain bark will be used by the bats opportunistically (Foster and Kurta 1999). Carter and Feldhamer (2005) hypothesized that structural complexity of habitat or available roosting resources are more important factors than the actual tree species.

In the majority of northern long-eared bat telemetry studies, roost trees consist predominantly of hardwoods (Foster and Kurta 1999, Lacki and Schwierjohann 2001, Broders and Forbes 2004). Broders and Forbes (2004) reported that female northern long-eared bat roosts in New Brunswick were 24 times more likely to be shade-tolerant, deciduous trees than conifers. Of the few northern long-eared bat telemetry studies in which conifers represented a large proportion of roosts, most were reported as snags (Cryan et al. 2001, Jung et al. 2004). Overall, these data suggest that hardwood trees most often provide the structural and microclimate conditions preferred by maternity colonies and groups of females, which have more specific roosting needs than solitary males (Lacki and Schwierjohann 2001), although softwood snags may offer more suitable roosting habitat for both genders than hardwoods (Perry and Thill 2007, Cryan et al. 2001). One reason deciduous snags may be preferred over conifer snags is increased resistance to decay, and consequently, roost longevity of the former (80 FR 17984).

Many studies have documented the northern long-eared bat's selection of both live trees and snags, with a range of 10 to 53% selection of live roosts (Sasse and Pekins 1996, Foster and Kurta 1999, Lacki and Schwierjohann 2001, Menzel et al. 2002, Carter and Feldhamer 2005, Perry and Thill 2007, Timpone et al. 2010). Foster and Kurta (1999) found 53% of roosts in Michigan were in living trees, whereas in New Hampshire, 66% of roosts were in live trees (Sasse and Pekins 1996). The use of live trees versus snags may reflect the availability of such structures in study areas (Perry and Thill 2007) and the flexibility in roost selection when there is a sympatric bat species present (e.g., Indiana bat) (Timpone et al. 2010). Most telemetry studies describe a greater number of dead than live roosts (Cryan et al. 2001, Lacki and Schwierjohann 2001, Timpone et al. 2010, Silvis et al. 2012). A significant preference for dead or dying trees was reported for northern long-eared bats in Kentucky (Silvis et al. 2012), Illinois and Indiana; in South Dakota (Cryan et al. 2001) and West Virginia, northern long-eared bat roost plots contained a higher than expected proportion of snags (Owen et al. 2002). Moreover, most studies reporting a higher proportion of live roosts included trees that had visible signs of decline, such as broken crowns or dead branches (Foster and Kurta 1999, Ford et al. 2006). Thus, the tendency for northern long-eared bats (particularly large maternity colonies) to use healthy live trees appears to be fairly low.

Canopy coverage at northern long-eared bat roosts has ranged from 56% in Missouri (Timpone et al. 2010, to 66% in Arkansas (Perry and Thill 2007), to greater than 75% in New Hampshire (Sasse and Pekins 1996), to greater than 84% in Kentucky (Lacki and Schwierjohann 2001). Studies in New Hampshire and British Columbia have found that canopy coverage around roosts is lower than in available stands (Sasse and Pekins 1996). Females tend to roost in more open areas than males, likely due to increased solar radiation, which aids pup development (Perry and Thill 2007). Fewer trees surrounding maternity roosts may also benefit juvenile bats that are

starting to learn to fly (Perry and Thill 2007). However, in southern Illinois, northern long-eared bats were observed roosting in areas with greater canopy cover than in random plots (Carter and Feldhamer 2005). Roosts are also largely selected below the canopy, which could be due to the species' ability to exploit roosts in cluttered environments; their gleaning behavior suggests an ability to easily maneuver around obstacles (Foster and Kurta 1999, Menzel et al. 2002).

Results from studies have found the diameters of roost trees selected by northern long-eared bats vary greatly. Some studies have found that the DBH of northern long-eared bat roost trees was greater than random trees (Lacki and Schwierjohann 2001), and others have found both DBH and height of selected roost trees to be greater than random trees (Sasse and Pekins 1996, Owen et al. 2002). However, other studies have found that roost tree mean DBH and height did not differ from random trees (Menzel et al. 2002, Carter and Feldhamer 2005). Based on a consolidation of data from across the northern long-eared bat's range (Sasse and Pekins 1996, Foster and Kurta 1999, Lacki and Schwierjohann 2001, Owens et al. 2002, Schultes 2002, Carter and Feldhamer 2005, Perry and Thill 2007, Lacki et al. 2009, Timpone et al. 2010, Lowe 2012, Lereculeur 2013, 80 FR 17985), roost tree DBH most commonly used (close to 80% of over 400 documented maternity tree roosts) by northern long-eared bat maternity colonies ranged from 10 - 25 centimeter (cm), or 4 - 10 in.

As for elevation of northern long-eared bat roosts, Lacki and Schwierjohann (2001) have found that northern long-eared bats roost more often on upper and middle slopes than lower slopes, which suggests a preference for higher elevations, possibly due to increased solar heating. Silvis et al. (2012), found that selection of mid- and upper slope roost areas may also be a function of the landscape position, where forest stands are most subjected to disturbance (e.g., wind, more intense fire, more drought stress, higher incidence of insect attack) which, in turn, creates suitable roost conditions among multiple snags and trees within the stand.

Some studies have found tree roost selection to differ slightly between male and female northern long-eared bats. Some studies have found male northern long-eared bats more readily using smaller diameter trees for roosting than females, suggesting males are more flexible in roost selection than females (Lacki and Schwierjohann 2001, Broders and Forbes 2004, Perry and Thill 2007). In the Ouachita Mountains of Arkansas, both sexes primarily roosted in pine snags, although females roosted in snags surrounded by fewer midstory trees than did males (Perry and Thill 2007). In New Brunswick, Canada, Broders and Forbes (2004) found that there was spatial segregation between male and female roosts, with female maternity colonies typically occupying more mature, shade-tolerant deciduous tree stands, and males occupying more conifer-dominated stands. Data from West Virginia at the Fernow Experimental Forest and the former Westvaco Ecosystem Research Forest (both of which contain relatively unmanaged, older, mature stands; early successional/mid-age stands; and fire-modified stands) suggests that females choose smaller diameter, suppressed understory trees, whereas males often chose larger, sometimes canopy-dominant trees for roosts, perhaps in contrast to other tree-roosting myotids such as Indiana bats (Menzel et al. 2002, Ford et al. 2006, Johnson et al. 2009). A study in northeastern Kentucky found that males did not use colony roosting sites and were typically found occupying cavities in live hardwood trees, while females formed colonies more often in both hardwood and softwood snags (Lacki and Schwierjohann 2001). However, males and non-reproductively active females are found roosting within home ranges of known maternity colonies the majority

of the time (1,712 of 1,825 capture records or 94%) within Kentucky (80 FR 17985), suggesting little segregation between reproductive females and other individuals in summer.

2.2.3.3. Foraging habitat

Northern long-eared bats are nocturnal foragers and use hawking (catching insects in flight) and gleaning (picking insects from surfaces) behaviors in conjunction with passive acoustic cues (Nagorsen and Brigham 1993, Ratcliffe and Dawson 2003). Broders et al. (2006) and Henderson and Broders (2008) found foraging areas (of either sex) to be six or more times larger than roosting areas. The mean distance between roost trees and foraging areas of radio-tagged individuals in New Hampshire was 620 m (2,034.1 ft) (Sasse and Perkins 1996).

Emerging at dusk, most hunting occurs above the understory, 1 to 3 m (3 to 10 ft) above the ground, but under the canopy (Nagorsen and Brigham 1993) on forested hillsides and ridges, rather than along riparian areas (Brack and Whitaker 2001, LaVal et al. 1977). This coincides with data indicating that mature forests are an important habitat type for foraging northern long-eared bats (Caceres and Pybus 1997). Occasional foraging also takes place over small forest clearings and water, and along roads (Van Zyll de Jong 1985). Foraging patterns indicate a peak activity period within five hours (hr) after sunset followed by a secondary peak within eight hours after sunset (Kunz 1973).

2.2.4. Population dynamics/status and distribution

Most records of northern long-eared bats have been from winter hibernacula surveys (Caceres and Pybus 1997). More than 1,100 northern long-eared bat hibernacula have been identified throughout the species' range in the U. S., although many hibernacula contain only a few (one to three) individuals (Whitaker and Hamilton 1998). Northern long-eared bats are documented in hibernacula in 29 of 37 states (these states are identified in the Section 2.2.8) in the species' range (80 FR 17976). Known hibernacula (sites with one or more winter records of northern long-eared bats) include: Alabama (2), Arkansas (41), Connecticut (8), Delaware (2), Georgia (3), Illinois (21), Indiana (25), Kentucky (119), Maine (3), Maryland (8), Massachusetts (7), Michigan (103), Minnesota (11), Missouri (more than 269), Nebraska (2), New Hampshire (11), New Jersey (7), New York (90), North Carolina (22), Oklahoma (9), Ohio (7), Pennsylvania (112), South Carolina, (2), South Dakota (21), Tennessee (58), Vermont (16), Virginia (8), West Virginia (104), and Wisconsin (67) (80 FR 17976). Other states within the species' range have no known hibernacula (due to no suitable hibernacula present, lack of survey effort or existence of unknown retreats) (80 FR 17976). The species typically roosts in small crevices or cracks on cave or mine walls, or ceilings; therefore, they are easily overlooked during surveys and usually observed in small numbers (Griffin 1940b, Barbour and Davis 1969, Caire et al. 1979, Van Zyll de Jong 1985, Caceres and Pybus 1997, Whitaker and Mumford 2009).

In Tennessee, northern long-eared bats have been observed in both summer mist-net surveys and winter hibernacula counts. Summer mist-net surveys from 2002 through 2013 resulted in the capture of more than 1,000 individuals, including males and juveniles or pregnant, lactating or post-lactating adult females (80 FR 17981). During the winter of 2009–2010, the TWRA began tracking northern long-eared bat populations and has since documented northern long-eared bats

in 58 hibernacula, with individual hibernaculum populations ranging from 1 to 136 individuals (80 FR 17981). According to TWRA, Tennessee has over 9,000 caves and less than 2% of those have been surveyed, which led them to suggest that there could be additional unknown northern long-eared bat hibernacula in the state (80 FR 17981).

The northern long-eared bat is found in the U. S. from Maine to North Carolina on the Atlantic Coast, westward to eastern Oklahoma and north through the Dakotas, even reaching into eastern Montana and Wyoming. The species' current range includes the following 37 states: Alabama, Arkansas, Connecticut, Delaware, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Montana, Nebraska, New Hampshire, New Jersey, New York, North Carolina, North Dakota, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming, and the District of Columbia (Nagorsen and Brigham 1993, Caceres and Pybus 1997). In Canada, it is found from the Atlantic Coast, westward to the southern Yukon Territory and eastern British Columbia (Nagorsen and Brigham 1993, Caceres and Pybus 1997).

Historically, the species was found in greater abundance in the northeastern and portions of the midwestern and southeastern U. S., and the Canadian Provinces of Quebec and Ontario, with increased sightings during swarming and hibernation (Caceres and Barclay 2000). However, throughout the majority of the species' range, it was patchily distributed, and historically, less common in the western portions of the range (Amelon and Burhans 2006). A single historical record (winter 1954) from Jackson County, Florida, indicates that the species was observed in a cave in that locality. However, since that observation, historical and recent surveys at this cave and 12 other caves in Jackson County have not resulted in documentation of the northern long-eared bat (80 FR 17975 - 17976).

2.2.5. Categorization of caves/hibernacula

A species' recovery plan has not yet been developed for the northern long-eared bat. Therefore, at this date, the Service has not assigned northern long-eared bat cave or hibernacula priority levels based on biological significance, location, winter population sizes, vulnerability, etc.

2.2.6. Threats

The final listing rule for the northern long-eared bat (80 FR 17973 - 18033) described known threats to the species under each of the five statutory factors for listing decisions, of which disease/predation is the dominant factor (due to WNS).

Northern long-eared bats are believed to experience only a small amount of predation; therefore, predation does not appear to be a population changing cause of mortality (Caceres and Pybus 1997, Whitaker and Hamilton 1998). Diseases known or suspected to infect northern long-eared bats, include the rabies virus (Constantine 1979, Main 1979, Burnett 1989) and equine encephalitis (Main 1979), but are not known to have appreciable effects on the species. Northern long-eared bats are also known to carry a variety of pests including chiggers (Trombiculidae), mites (Acaridae), bat bugs (Cimicidae) and internal helminths (Coccidia, Nematoda and

Trematoda) (Caceres and Barclay 2000). However, the level of mortality caused by WNS far exceeds mortality from all other known diseases and pests of northern long-eared bats.

A discussion of WNS and its effects to bat populations has previously been discussed under *Indiana bat Threats* (section 2.1.5). Therefore, our discussion of WNS, here, will be pertinent to specific effects and impacts of the disease to the northern long-eared bat. The effect of WNS on northern long-eared bats has been especially severe and has caused mortality in the species throughout the majority of the WNS-affected range (80 FR 17996). This is currently viewed as the predominant threat to the species, and if WNS had not emerged or was not affecting northern long-eared bat populations to the level that it has, the Service presumes the species would not be declining to the degree observed. Microclimate inside the cave, duration and severity of winter, hibernating behavior, body condition of bats, genetic structure of the colony, and other variables may affect the timeline and severity of impacts at the site level. However, there is no evidence to date that any of these variables would greatly delay or reduce mortality in infected colonies.

WNS was first documented in Tennessee during the winter of 2009-2010 (Lamb and Wyckoff 2010), when the fungus was found on three species of bats including the little brown bat, eastern pipistrelle and northern long-eared bat in six caves within six different counties. The number of counties in Tennessee that have received WNS confirmations doubled from 6 to 12 during the 2011-2012 hibernation season (Holliday 2012). WNS-related mortality was documented (including northern long-eared bat mortality) in Tennessee in 2014 (80 FR 17979); however, there is no pre-WNS data from these sites prior to 2009 to draw any conclusions regarding population trend rates of northern long-eared bats. In comparison to 2009–2012, northern long-eared bats had declined 71 - 94% (across all sites) by 2014, based on unit of effort comparisons (80 FR 17981). Over 70% of the 185 northern long-eared bats tested for the presence of Pd in Tennessee hibernacula between 2011 and 2014 were found to have Pd (80 FR 17997).

The northern long-eared bat has apparently experienced a precipitous population decline, estimated at approximately 96% (from hibernacula data) in the northeastern portion of its range (data from at 103 sites across Quebec, New York, Massachusetts, New Hampshire, Vermont, Connecticut, New Jersey, Pennsylvania, Maryland, North Carolina, Virginia and West Virginia), due to the emergence of WNS. WNS has spread to approximately 60% of the northern long-eared bat's range in the U. S., and if the observed average rate of spread of Pd continues, the fungus will be found in hibernacula throughout the entire species' range within 8 to 13 years, based on the calculated rate of spread observed to date by the Service and the Committee on the Status of Endangered Wildlife in Canada (80 FR 18000).

Northern long-eared bats prefer to hibernate at temperatures between 0 and 9° C (32 to 48° F) (Raesly and Gates 1987, Caceres and Pybus 1997, Brack 2007), which falls within the optimal growth limits of Pd, at 5 to 16° C (41 to 61° F) (Blehert et al. 2009, Verant et al. 2012), making them susceptible to WNS infection once exposed to Pd, regardless of hibernaculum type. Northern long-eared bats also roost in areas within hibernacula that have higher humidity. Cryan et al. (2001) suggested this roosting preference may be due to the northern long-eared bat's high intrinsic rates of evaporative water loss during torpor. Langwig et al. (2012) suggested that these more humid conditions could explain why northern long-eared bats actually experience higher rates of infection than other species, such as Indiana bats. Northern long-eared bats have been

reported to enter hibernation in October or November, but sometimes return to hibernacula as early as August, and emerge in March or April (Caire et al. 1979, Whitaker and Hamilton 1998, Amelon and Burhans 2006). This extended period of time (in comparison to many other cave bat species that have been less impacted by WNS) may explain observed differences in fungal loads of Pd when compared to less susceptible species because the fungus has more time to infect bats and grow. Langwig et al. (2015) determined that nearly 100% of northern long-eared bats sampled in 30 hibernacula across six states (New York, Vermont, Massachusetts, Virginia, New Hampshire and Illinois) were infected with Pd early in the hibernation period, and that northern long-eared bats had the highest Pd-load of any other species in these sites. Similar patterns of high prevalence and fungal load in northern long-eared bats were reported by Bernard (80 FR 17998) for bats surveyed outside of hibernacula in Tennessee during the winter. Furthermore, the northern long-eared bat occasionally roosts in clusters or in the same hibernacula as other bat species that are also susceptible to WNS and are susceptible to bat-to-bat transmission of WNS.

There has been a sustained and coordinated effort between partners (e.g., Federal, state, Canada, non-government) to curtail the spread of WNS, and while these measures may reduce or slow the spread of WNS, these efforts are currently not enough to ameliorate the population-level effects on the northern long-eared bat. Also, research is under way to develop control and treatment options for WNS-infected bats and hibernacula; however, additional research is needed before potential treatments are implemented on a landscape scale.

We summarize the findings of the final listing rule (80 FR 17973 - 18033), regarding the other four factors that are relevant to this consultation, below.

Human and non-human modification of hibernacula, particularly altering or closing hibernacula entrances, is considered the next greatest threat (following WNS) to the northern long-eared bat. Some modifications (e.g., closure of a cave entrance with structures/materials other than bat-friendly gates) can cause a partial or complete loss of the utility of a site to serve as hibernaculum. Humans can also disturb hibernating bats, either directly or indirectly, resulting in an increase in energy-consuming arousal bouts during hibernation (Thomas et al. 1990, Thomas 1995).

During the summer, northern long-eared bat habitat loss is primarily due to forest conversion, and to a lesser degree, forest management. Throughout the range of the northern long-eared bat, forest conversion is expected to increase due to commercial and urban development, energy production and transmission, and natural changes. Forest conversion causes loss of potential habitat, fragmentation of remaining habitat, and if occupied at the time of the conversion, direct injury or mortality to individuals. Forest management activities, unlike forest conversion, typically result in temporary impacts to the habitat of northern long-eared bats, but like forest conversion, may also cause direct injury or mortality to individuals. The net effect of forest management may be positive, neutral or negative, depending on the type, scale and timing of various practices. The primary potential benefit of forest management to the species is perpetuating forests on the landscape that provide suitable roosting and foraging habitat. The primary potential impacts of forest management are greatly reduced with the use of various measures that avoid or minimize effects to bats and their habitat (e.g., limiting the size of

clearcuts, avoiding or minimizing timber harvest during the flightless period for bat pups, leaving sufficient numbers of snags and other trees suitable as roosts following harvests, etc.).

Wind energy facilities are known to cause mortality of northern long-eared bats. While mortality estimates vary between sites and years, sustained mortality at particular facilities could cause declines in local populations. Wind energy development within portions of the species' range is projected to continue.

Climate change may also affect this species, as northern long-eared bats are particularly sensitive to changes in temperature, humidity, and precipitation. Climate change may indirectly affect the northern long-eared bat via changes in food availability and the timing of hibernation and reproductive cycles.

Environmental contaminants, in particular insecticides, other pesticides, and inorganic contaminants, such as mercury and lead, may also have detrimental effects on northern long-eared bats. Contaminants may bio-accumulate (become concentrated) in the tissues of bats, potentially leading to a myriad of sub-lethal and lethal effects.

2.3. Blackside dace (*Chrosomus* [= *Phoxinus*] *cumberlandensis*)

The blackside dace (*Chrosomus cumberlandensis*) was listed as a threatened species on June 12, 1987 (52 FR 22580-22585). A recovery plan addressing the blackside dace was approved on August 17, 1988 (U. S. Fish and Wildlife Service 1988).

Critical habitat has not been designated for this species. At the time of its listing, the Service determined that designation of critical habitat was not prudent for the species (52 FR 22580-22585). The Service believed that the designation and publication of critical habitat areas would increase the species' vulnerability to illegal taking and/or vandalism, further threaten the species survival and increase law enforcement problems. The Service decided that protection of the species' habitat could best be accomplished by providing suitable habitat locations to all appropriate local, state and Federal agencies, and that additional protection of the species' habitat would be addressed through Federal recovery and formal consultation processes.

2.3.1. Species description

The blackside dace is a member of the minnow family (Cyprinidae) (Etnier and Starnes 1993). The species was probably first observed in 1883 by D. S. Jordan and J. Swain in Clear Fork River tributaries in Whitley County, Kentucky, and described based on color; they regarded it as a color variation of the southern redbelly dace (*Chrosomus erythrogaster*) (Starnes and Starnes 1978). The species was not recognized and described as a distinct species until 1978 (Starnes and Starnes 1978). In a recent phylogenetic analysis of all North American and Eurasian species, *Phoxinus phoxinus*, Strange and Mayden (2009) found that the genus, as currently recognized, is an unnatural group. To have a classification that is consistent with the monophyletic groups recovered in their phylogeny, they proposed a revised taxonomy that elevates the subgenus *Chrosomus* to genus rank. *Chrosomus* includes the blackside dace, among

six other North American species currently recognized in the genus *Phoxinus* (Strange and Mayden 2009).

The blackside dace reaches a maximum length of approximately three inches and is characterized by a wide, black lateral stripe or two stripes, converging on the caudal peduncle, an olive-colored dorsal surface with numerous dark spots/speckles, and scarlet and yellow coloration on the head and abdomen (most pronounced in the spring). The blackside dace is similar to the southern redbelly dace but can be distinguished by its single lateral stripe or two convergent stripes (the southern redbelly dace has two parallel stripes) and the shape of its opercular bone (Starnes and Starnes 1978, Etnier and Starnes 1993).

2.3.2. Life history

The biology of blackside dace is only partially understood. Feeding habits and reproductive characteristics were investigated by Starnes and Starnes (1981), who reported schools of 5 to 20 fish grazing on rocks and sandy substrates. Gut analyses revealed that sand comprised the largest portion of the species' gut (36 percent [%]). The remaining portions of the gut were composed of unidentified organisms (32%), algae and diatoms (12%) and macroinvertebrates (4.5%). However, during the winter, macroinvertebrates (mostly aquatic insect larvae) composed the entire diet (Starnes and Starnes 1981).

The spawning period for the species extends from April until July (Starnes and Starnes 1981), but spawning individuals have been observed in late March in the Rock Creek basin, a tributary of Jellico Creek in southeastern McCreary County, Kentucky. Eggs are typically deposited in fine gravel, primarily in nests constructed by other species such as creek chubs (*Semotilus atromaculatus*) (Cicerello and Laudermilk 1996) and central stonerollers (*Campostoma anomalum*) (Starnes and Starnes 1981). Creek chub nests appear to be used more often than stoneroller nests, as suggested by Cicerello and Laudermilk (1996). Interestingly, Lewis and Cashner (2015) reported that creek chubs will defend their nest mounds but they tolerate nest associates that exhibit bright red hues when they are spawning, including blackside dace.

In a study of blackside dace reproductive behavior, Mattingly and Black (2007) provided evidence that blackside dace rely heavily on creek chubs as a nest-building spawning associate, perhaps in an obligatory fashion. Mattingly and Black (2007) observed 25 spawning events, and all of these events took place over creek chub nests; there was no evidence that blackside dace spawned independently. It is suspected that the species takes advantage of other species' nests because these habitats provide the most abundant silt-free substrates in much of the species' current range and possibly afford the eggs some protection from predators by guarding these nests. Rakes et al. (2013) were successful in enticing captive blackside dace to spawn without any cues from other nest building fish species but they did construct a rock mound similar to the nest constructed by creek chubs. However, it remains unknown whether the species will construct nests independent of other species in natural streams if suitable substrates are available (Mattingly 2006).

Scherer & Santangelo (2013) studied a large, healthy, stable population of blackside dace in Big Lick Branch in the Daniel Boone National Forest (DBNF) in Kentucky, to quantify physical

stream characteristics associated with spawning habitat. They found 16 active nests during their surveys between early May and mid-June. Nests found were all shallow, but well-defined gravel pits, often located at the downstream ends of pools. More than half the nests observed by Scherer and Santangelo (2013) were at depths of 10-15 centimeter (cm), with flows between 0.06 – 0.14 meters per second (m/sec); 75% of the nests they observed were placed one third to halfway from stream bank, closer to the center of the stream channel than the bank. They observed at least five and up to 20 or more males over these nests during their observations.

While they were seen in the stream, but not observed constructing nests, Scherer and Santangelo (2013) assumed creek chub (*Semotilus atromaculatus*) was the nest-building associate in Big Lick Branch. These authors suggested that blackside dace benefit from nest selection choices made by other species, but they also commented that several inactive nests were observed, indicating that the dace may also discriminate between nests and choose ones that they consider to be better.

Starnes (1981) observed spawning in May at water temperatures of approximately 64 degrees Fahrenheit (ºF). Females deposited eggs on fine gravel at the edge of an existing stoneroller nest located in a run area. Adults are capable of spawning at age one (I) and have a lifespan of three to four years (Starnes and Starnes 1981); females appear to have greater survivorship (Starnes and Starnes 1981). Starnes and Starnes (1981) reported the sex ratio in September as 21 males-to-29 females and in April as 11 males-to-11 females. Based on length-frequency and scale data, growth rates were similar for males and females (age 0, 20 to 24 millimeters [mm] [approximately 0.8 to 1-in] standard length [SL]; age I, 39 to 57 mm [approximately 1.5 to 2.2 in] SL; and age II, 62 to 64 mm [approximately 2.4 to 2.5 in] SL). The fastest growth occurred during the first year and then gradually declined during the second and third years (Starnes and Starnes 1981).

Fish species commonly found in association with blackside dace include the creek chub, central stoneroller, white sucker (*Catastomus commersoni*), northern hogsucker (*Hypentelium nigricans*), green sunfish (*Lepomis cyanellus*), stripetail darter (*Etheostoma kennicotti*), arrow darter (*E. sagitta sagitta*) and rainbow darter (*E. caeruleum*) (Starnes and Starnes 1978, O'Bara 1990, Mattingly et al. 2005). Additional species that may occur along with blackside dace include the bluntnose minnow (*Pimephales notatus*), silverjaw minnow (*Notropis buccata*), striped shiner (*Luxilus chrysocephalus*), longear sunfish (*Lepomis megalotis*), and redbreast sunfish (*Lepomis auritus*). Based on published research by Eisenhour and Piller (1997), some blackside dace/creek chub hybrids result from this nest association.

Mattingly et al. (2005) and Detar and Mattingly (2013) studied movement patterns (frequency, spatial extent, directionality and environmental correlates) of the species by tagging 653 dace from Big Lick Branch (Pulaski County, Kentucky) and Rock Creek (McCreary County, Kentucky) with visible implant elastomer injections. Dace were recaptured from February 2003 through March 2004 using baited minnow traps. The majority of tagged dace (81% in Big Lick Branch and 58% in Rock Creek) were recaptured within the 200-meter (m) (approximately 656 ft) stream reach where tagging had occurred. However, several individuals moved considerable distances from the original tagging site, including the first documented inter-tributary movement for the species. Mean distances moved upstream in these two systems (148 ± 138 m [486 ± 453

ft] in Big Lick Branch; 733 ± 1,259 m [2,405 ± 4,131 ft] in Rock Creek) were not statistically different from mean distances moved downstream (77 ± 29 m [253 ± 95 ft] in Big Lick Branch; 314 ± 617 m [1,030 ± 6,640 ft] in Rock Creek). However, the mean overall distance moved was statistically greater in Rock Creek, a longer stream, than in Big Lick Branch; maximum distances moved in Big Lick Branch and Rock Creek were 1 kilometer (km) (0.6-mile [mi]) and 4 km (2.5 mi), respectively. These results were similar to other fish movement studies, suggesting that some stream fish populations are comprised of a relatively large sedentary group and a small mobile group (Freeman 1995, Smithson and Johnston 1999, Rodriguez 2002).

2.3.3. Habitat characteristics and use

Habitat for the blackside dace consists of small (generally 4 to 15 feet wide), cool (rarely exceeding 80 ºF), upland streams with moderate flows and generally, silt-free substrates (Starnes and Starnes 1978, 1981, O'Bara 1985, 1990, U. S. Fish and Wildlife Service 1988, 2015d, Mattingly et al. 2005, Black et al. 2013a, b, Mattingly and Black 2013). Streams inhabited by the species are generally those with good riparian vegetation that provide at least 70% canopy cover and numerous submerged root wads, undercut banks and large rocks (U. S. Fish and Wildlife Service 1988). Blackside dace rarely have been found in low-gradient streams or high-gradient tributaries (O'Bara 1985, U. S. Fish and Wildlife Service 1988, 2015d, Black et al. 2013a). A riffle to pool ratio less than 60:40 and elevations ranging from 300 to 500 m (984 to 1,640 ft) above mean sea level (msl) appear to be preferred by the species (Starnes and Starnes 1981, O'Bara 1990). Streams with higher riffle to pool ratios (above 60:40) harbor fewer populations of blackside dace and tend to be dominated by blacknose dace (*Rhinichthys atratulus*) and creek chubs.

These descriptions are also supported by the results of a blackside dace survey in a stream located partially on the DBNF in Kentucky reported by Leftwich et al. (1995). They surveyed all available habitat units in the stream and found blackside densities increased from downstream to upstream reaches, with most blackside dace (75% or more) found in pools, and the remainder in riffles. The stream contained a total of 11 species of fish, including non-native rainbow trout.

Also, Hitt et al. (2016) used range-wide collections of blackside dace to construct a model that identified the species' persistence in relation to important landscape and other variables. The model, included conductivity, stream temperature, slope, and watershed area (mean watershed areas of 20 km^2) as the four most important variables, explained 75.7% of the total variance in the data they analyzed. Blackside dace occurrence probability decreased as conductivity values increased from the 13 microsiemens per centimeter (µS/cm) minimum observation to approximately 400 µS/cm, and in watersheds smaller than approximately 20 km^2, above both levels which the species' occurrence probability was diminished. Stream temperature was nearly as important (less than 20°C) as conductivity.

2.3.4. Population dynamics/status and distribution

Originally the blackside dace was known only from small streams tributary to the Cumberland River upstream of Cumberland Falls, the recently understood distribution also includes streams tributary to the Cumberland River downstream of Cumberland Falls, upper Kentucky River

system, and the upper Tennessee River system (U. S. Fish and Wildlife Service 2015d and see Figure 6).

Rangewide surveys completed from 1982−1994, 2003−2006, and 2010−2012 demonstrate that the species has been extirpated from 31 streams (Laudermilk and Cicerello 1998, Black et al. 2013b, Compton et al. 2013, U. S. Fish and Wildlife Service 2015d). Populations of the blackside dace continue to persist in 125 streams, including: (1) the upper Cumberland River drainage in eight Kentucky counties (Bell, Harlan, Knox, Laurel, Letcher, McCreary, Pulaski and Whitley) and three Tennessee counties (Scott, Campbell and Claiborne) (Burr and Warren 1986, Mattingly et al. 2005, Black et al. 2013b, Skelton 2013), (2) the upper Kentucky River drainage in one Kentucky county (Perry County in the Maces Creek watershed) (U. S. Fish and Wildlife Service 2015d), and (3) the Powell and Clinch River drainages in two Virginia counties (Lee and Scott) (U. S. Fish and Wildlife Service 1988, Laudermilk and Cicerello 1998, Skelton 2007, Black et al. 2013b, U. S. Fish and Wildlife Service 2015d). See Figure 6 and Table 2.

Most land ownership within watersheds occupied by blackside dace is private, but portions of 61 of these watersheds are in public ownership. The majority (85%) of the watershed areas on public lands are located on the DBNF in Laurel, McCreary, Pulaski and Whitley counties, Kentucky (U. S. Fish and Wildlife Service 2015d). Most blackside dace populations are considered to be small and remnant in nature (i.e., less than ten individuals observed during surveys), an adequate understanding of population viability is lacking (Black et al. 2013).

In spite of several previous surveys (Shoup and Peyton 1940, Comiskey and Etnier 1972, Brazinski 1979, O'Bara et al. 1982, and Kirsch 1983), the blackside dace was unknown from the Big South Fork National River and Recreation Area (BISO) until Scott (2010) reported results from a 2003-2006 inventory of BISO fishes (and see Bivens et al. 2013). They reported results of 68 surveys of 41 tributaries (some tributaries were surveyed at more than one site, and some sites were surveyed more than once). These surveys identified blackside dace at two localities in a single headwater stream in the system, representing a new distributional record for the species. The extent of this population's range and size in the BISO is unknown, but Scott (2010) considered this species rare, comprising only 0.24% of the total individuals of all fish species he collected at the 82 total sites surveyed.

Genetic analyses of newly discovered blackside dace populations in the Kentucky River system are being examined to help identify which hypothesis (human stocking "bait bucket", recent natural dispersal, or genetically distinct populations) best explains the origin of these recently discovered populations (Bedal and Cashner 2016). Conclusions of this analysis are not yet available.

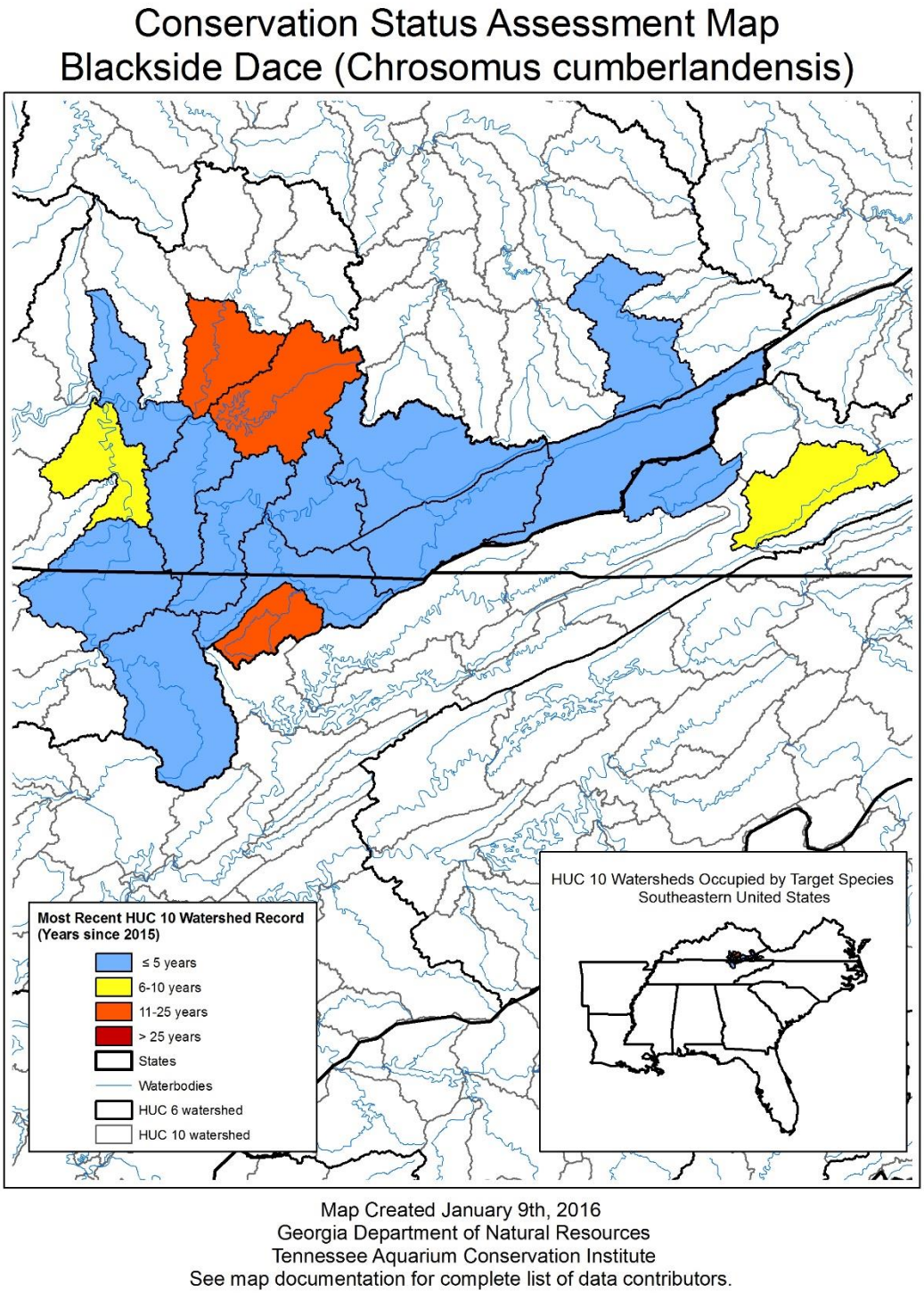

Figure 6. Geographic distribution of blackside dace (*Chrosomus* [= *Phoxinus*] *cumberlandensis*).

Table 2. Number of streams from the major river systems within blackside dace range considered to have extant blackside dace populations in 2015. Adapted from U. S. Fish and Wildlife Service (2015d).

River System	Number of Extant Populations
Upper Cumberland River system (upstream of Cumberland Falls)	90
Middle Cumberland River system (downstream of Cumberland Falls)	27
Upper Kentucky River system	1
Upper Tennessee River system	6

Genetic analyses of newly discovered blackside dace populations in the Kentucky River system are being examined to help identify which hypothesis (human stocking "bait bucket", recent natural dispersal, or genetically distinct populations) best explains the origin of these recently discovered populations (Bedal and Cashner 2015). To date, conclusions of this analysis are not available.

Mattingly et al. (2005), Detar (2004), and Black et al. (2013b) investigated population densities within the species' current range, and Leftwich et al. (1995) investigated blackside dace distribution and abundance within the Daniel Boone National Forest (DBNF) in Kentucky. Blackside dace were captured at 52 of 72 sites (25 of 28 streams) in the upper Cumberland River basin using single-pass electrofishing from June to August, 2003. The majority of sites (58%) had catch rates of ten or fewer dace per 200-m site, and 70 or more dace were captured in only 7 of the 72 sites. Single pass catch rates averaged 29 ± 37 (mean ± standard deviation [SD]; n = 52) dace per 200 m in occupied sites, while the median number of captured dace was 14/200 m. Using the Petersen mark-recapture method, population estimates were also conducted on nine sites within five different streams. Population estimates averaged 176 ± 133 dace per 200 m (range: 33 - 429), corresponding to density estimates of 27.7 ± 19.7 dace/100 square meters (m^2) (range: 2.7 - 55.3). Based on these data, a regression model was constructed to obtain population estimates for the other 43 sites in which dace were captured during single-pass electrofishing. Population estimates for these sites averaged 65 ± 82 dace/200 m (range: 4 – 321), corresponding to density estimates of 8.8 ± 13.6 dace/100 m^2 (range: 0.3 – 73.9). Overall, population estimates for the 52 sites in which dace were present averaged 84 ± 101 dace/200 m, corresponding to density estimates of 12.1 ± 16.3 dace/100 m^2.

Density estimates (56.8 - 73.1 dace per 1/100 m^2) reported by Starnes and Starnes (1981) for three sites in Youngs Creek (Whitley County, Kentucky), one of the healthiest known populations at that time, were consistent with the two highest densities identified by Mattingly et al. (2005) and Detar (2004). Density estimates for Big Lick Branch by Leftwich et al. (1997) and Middle Fork Beaver Creek by Leftwich et al. (1995) were 10 - 350 dace/100 m^2 and 130 dace/100 m^2 (one pool), respectively. These results were considerably higher than those calculated by Mattingly et al. (2005) and Detar (2004), but both studies conducted by Leftwich et

al. (1995, 1997) were based on habitat units (pools and riffles), rather than specific stream lengths. Consequently, they may have encountered elevated densities of blackside dace in certain pools.

Black and Mattingly (2007) conducted additional presence-absence surveys and performed population estimates on an additional 27 streams (47 200-m reaches) in Kentucky and Tennessee via single-pass backpack electrofishing. Seven sites were double-sampled to allow estimates of population size (Peterson mark-recapture). Blackside dace were found in 18 of 27 streams and 27 of 47 reaches, but most reaches (72%) had catch rates of ≤ 10 dace/200 m. Occupied reaches had single-pass catch rates of 1 to 96 (21 ± 25) individuals. Petersen mark-recapture population estimates at seven selected reaches ranged from 54 to 613 dace per 200-m reach and densities averaged 37.5 ± 27.2 dace per 100 m^2. These results were used to calibrate single-pass electrofishing results and provide population estimates in the remaining 19 reaches. The mean population estimate was 46 ± 84 (range: 2 to 360) dace per 200 m, and the associated mean density was 9.3 ± 17.6 (range: 0.3 to 73.5) dace/100 m^2. Population estimates for the 26 reaches harboring dace averaged 92 ± 150 dace/200 m, and associated densities averaged 16.9 ± 24.2 dace/100 m^2. Black et al. (2013b) summarized this and some additional information, reporting that dace inhabited 43 of 55 streams and 78 of 119 reaches surveyed. The additional data altered density estimates to 14.1 ± 19.4 dace/100 m^2. Black et al. (2013b) however, commented that electrofishing sampling efficiency for blackside dace was 30% (i.e., 70% of the blackside dace present in an area avoided capture).

Mattingly et al. (2005) and Black and Mattingly (2007) identified 12 streams as having the most robust populations of the species. Population estimates for these streams exceeded 100 individuals/200-m reach and ranged from a low of 104 (Fall Branch, Tennessee) to a high of 613 (Breedens Creek, Kentucky) individuals.

Blackside dace densities reported by Leftwich et al. (1995) for a stream partially located on the DBNF (Middle Fork Beaver Creek) are also similar to those included above. The dace were widely distributed in the stream, and the surveys reported by Letwich et al. (1995) included many habitat units (including pools and riffles), with densities ranging from near 0 to around 1.3 individuals per square meter. The majority of blackside dace (75% or more) were found in pools and the remainder in riffles. Population estimates for individual habitat units surveyed ranged from 0 to 16 individual blackside dace, and as would be expected for a headwater stream inhabitant, Leftwich et al. (1995) reported that densities increased from downstream to upstream reaches. Hitt et al. (2016) also described blackside dace habitat as pools of first and second-order streams where streambanks are mostly forested; population densities in their dataset were 1-153 individuals (mean 11.7 and 26.0 standard deviation).

Strange and Burr (1995) investigated the genetic variation and meta-population structure of the species. Their research revealed the presence of three or four meta-population units: one centered in the upper Poor Fork through Straight Creek stream systems (Group A), another unit comprised the stream systems from Stinking Creek to Youngs Creek (Group B), a third centered around Marsh and Jellico creeks (Group C), and a potential fourth comprised of streams below Cumberland Falls. A cladistic analysis of gene-flow indicated that Group B was the center of dispersal for blackside dace mitochondrial-DNA haplotypes (Figure 7).

In order to preserve blackside dace genetic diversity, Strange and Burr (1995) recommended that recovery plans treat the meta-populations as management units, employing carefully planned reintroductions and habitat protection. Translocation of the species between meta-populations was discouraged; rather, they recommended that translocations be made from sites geographically proximate to the site of the reintroduction and preferably within the same stream system. Their data further indicated that considerable gene-flow took place within meta-populations, suggesting that protection of dispersal corridors may be as important as protecting actual habitats. Priority should be given to corridor protection, allowing the species to reinvade formerly occupied habitats on its own.

More recently, Bedal collected tissue samples from streams in the Tennessee portion of the upper Jellico and upper Clear Fork watersheds for a project using mitochondrial DNA (cytochrome b gene) to identify blackside dace population genetic structure. Their prelimary work identified 20 polymorphic microsatellite loci that will be used for population genetics analyses to identify the degree of genetic isolation or mixing between blackside dace populations (Bedal et al. 2016, Bedal and Cashner 2016).

Milton and Ambruster (2015), in a larger genetic survey, generated an "All Cypriniformes Tree of Life" by incorporating phylogenetic and phylogenomic data for 2000 cypriniforme fishes in order to provide input to conservation priorities as related to genetic distinctiveness and global endangerment. In the southeastern United States, the authors identified laurel dace (*Chrosomus saylori*) and blackside dace with higher EDGE scores than slender chub (*Erimystax cahni*). All three of these minnow species are listed as endangered on the IUCN Red List, but the slender chub has not been collected in decades, and is possibly extinct (U. S. Fish and Wildlife Service 2014b). EDGE scores consider the International Union for the Conservation of Nature's (IUCN) Red List global ranking and evolutionary distinctiveness into comparative indices. Milton and Ambruster (2015) suggested that, in spite of lower relative extinction risk than other cypriniforme fishes, conservation actions should be prioritized towards protection of species that include the more evolutionarily distinct components of the fish fauna, including blackside dace and laurel dace.

2.3.5. Threats

The recovery plan attributed the loss of many blackside dace populations to impacts associated with the extraction of coal and timber resources in Kentucky and Tennessee (U. S. Fish Wildlife Service 1988). Coal mining-related problems were identified as the primary threat to the species, followed in order of importance by logging, road construction, agriculture, human development and naturally low stream flows (U. S. Fish Wildlife Service 1988). All of these threats remain, but the overall decline of blackside dace can be attributed to a variety of human-related activities in the upper Cumberland River drainage. Resource extraction (e.g., surface coal mining, logging, oil/gas well exploration), land development, rural residential land use, road construction and agricultural practices have all contributed to the degradation of streams within the species' range (Mattingly et al. 2005, Kentucky Division of Water 2010, 2011, 2013, Tennessee Department of Environment and Conservation 2016b).

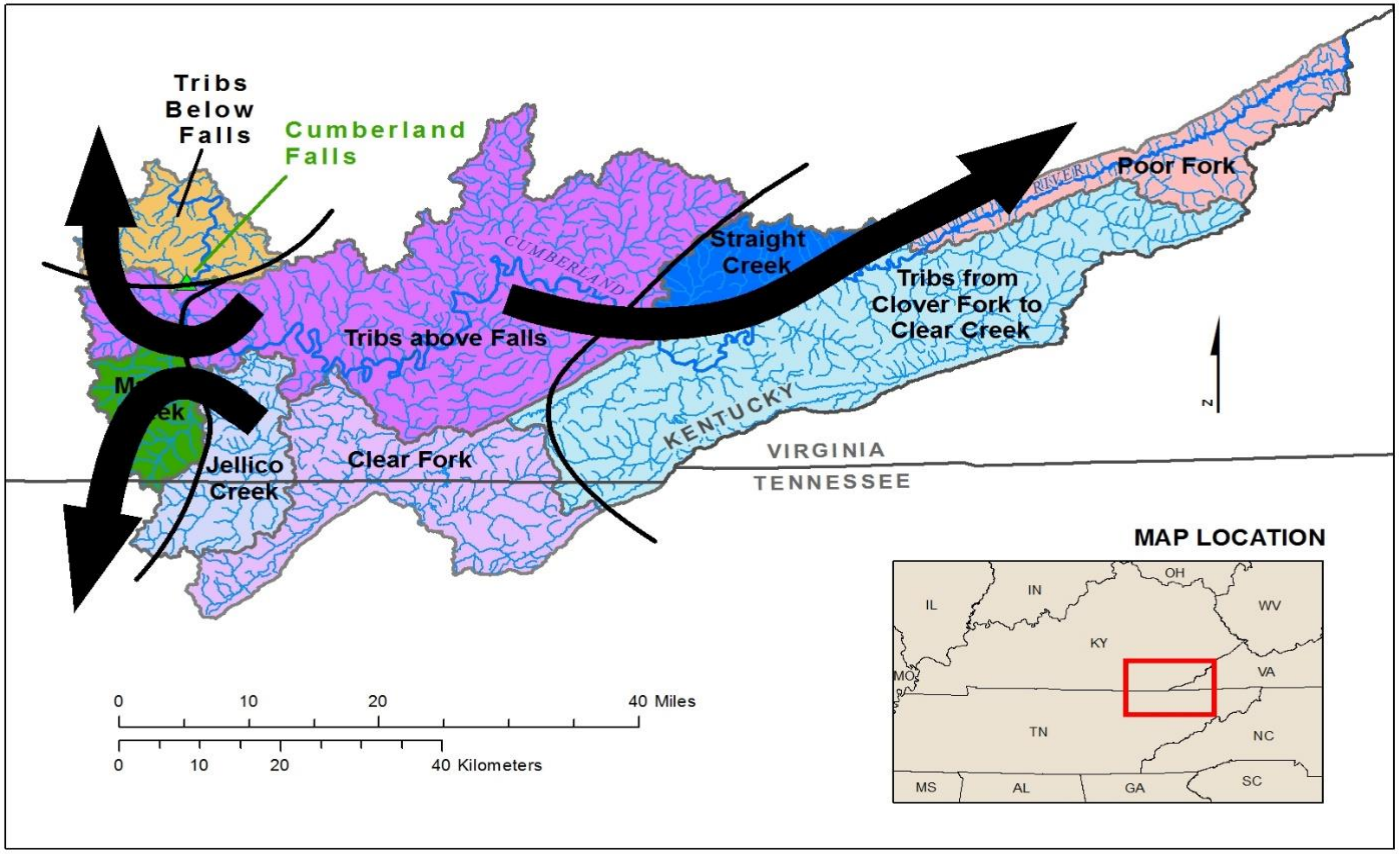

Figure 7. Blackside dace (*Chrosomus* [= *Phoxinus*] *cumberlandensis*) recovery units and direction of gene flow between meta-populations, as proposed by Strange and Burr (1995). Recvery units are indicated in various colors, as identified in the blackside dace recovery plan (U. S. Fish and Wildlife Service 1988), and are overlain by large arrows showing direction of gene flow by dispersal of blackside dace between meta-populations as proposed by Strange and Burr (1995). The thinner black lines separate the three metapopulations identified by Strange and Burr.

These land use activities have led to chemical and physical changes to stream habitats that have adversely affected the blackside dace and other fishes. Specific stressors have included inputs of dissolved solids and elevation of instream conductivity, inputs of nutrients and organic enrichment, sedimentation/siltation of stream substrates (excess sediments suspended or deposited in a stream), the removal of riparian vegetation, and the relocation or straightening of stream channels (Kentucky Division of Water 2011, 2013). Some of the most recently identified stream reaches where water or habitat quality is potentially affecting blackside dace populations include listings by the Kentucky Division of Water (2013) and the Tennessee Department of Environment and Conservation (2016c), who identified portions of 27 streams in Kentucky and seven streams in Tennessee as impaired on their respective 303(d) lists.

Black et al. (2013a) developed a model to predict occurrence of blackside dace. They found that specific conductivity (below 240 microsiemens per centimeter, µS/cm, during summer) was a good predictor of blackside dace presence and persistence. And, as described above in the *Habitat characteristics and use* (Section 2.3.3), Hitt et al. (2016) suggested 343 µS/cm as the blackside dace conductivity threshold. However, the mechanism of effect for impacts of elevated conductivity to blackside dace remains unknown. Preliminary, independent work by Yates et al. (2016) confirm the Hitt et al. (2016) threshold. Hitt et al. (2016) incorporated data from Kentucky streams only, and the Yates et al. (2016) analysis used data from Tennessee streams. While streams with high levels of conductivity may drain areas affected by various land uses (natural gas extraction, silviculture and urbanization), surface coal mining can also increase the conductivity of receiving streams (McAbee et al. 2013).

Blackside dace are undoubtedly consumed by natural predators; however, there is no evidence that predation is a significant threat to the species. The species has evolved with various predators over thousands of years and has continued to persist within the watershed. Disease is not known to be a threat to the species. Mattingly and Floyd (2013) summarized the status of blackside dace, and listed a variety of natural and anthropogenic impacts affecting blackside dace, alteration of habitat by beaver impoundment, including natural resource extraction, stream channelization, bridge or culvert construction that might prevent barriers to movement, and loss of, and alteration of, riparian vegetation. Eisenhour and Floyd (2013) described culvert barriers to blackside dace movement.

Climate change is expected to affect water quality, quantity in both inland and coastal areas, and alter the seasonal changes in stream flow. Specifically, precipitation is expected to occur more frequently via high-intensity rainfall events, causing increased runoff and erosion. More sediments and chemical runoff will therefore be transported into streams and groundwater systems, impairing water quality. Water quality may be further impaired if decreases in water supply cause nutrients and contaminants to become more concentrated. Rising air and water temperatures will also impact water quality by increasing primary production, organic matter decomposition and nutrient cycling rates in lakes and streams, resulting in lower dissolved oxygen levels. This suite of water quality effects will increase the number of water bodies in violation of today's water quality standards, worsen the quality of water bodies that are currently in violation, and ultimately increase the cost of meeting current water quality goals for both consumptive and environmental purposes (Adams and Peck 2008).

2.3.6. Recovery

When the recovery plan was completed in 1988, the species was known from a total of 35 streams in Kentucky and Tennessee. As indicated in section 2.3.4, currently, the blackside dace persists in 125 streams in Kentucky, Tennessee and Virginia. Considering the distribution of these streams and the species' maximum recorded movement of 4 km (2.5 mi), it is estimated the species is currently represented by 57 isolated groups (or populations) that are functionally separated from one another (U. S. Fish and Wildlife Service 2015d). Over the past 27 years, more streams have become occupied by the species in the eight sub-basins (recovery units, Figures 7 and 8) identified in the species' recovery plan (U. S. Fish and Wildlife Service 1988); however, more information is needed to evaluate the genetic diversity and viability of populations in these streams (U. S. Fish and Wildlife Service 2015d). The Clear Fork Recovery Unit is the only recovery unit affected by the actions included in this consultation.

Based on survey results and other observations regarding abundance, age-class structure and recruitment, the Service estimates that 76 streams contain stable populations, with the remaining 49 streams rated as vulnerable (U. S. Fish and Wildlife Service 2015d). As mentioned in section 2.3.4 (*Population dynamics/status and distribution)*, the species appears to have been extirpated from at least 31 streams in which it was previously documented.

Based on the best available scientific and commercial information available to the Service regarding the species' current status and past, present and future threats, the species continues to be impacted by poor water quality and habitat deterioration resulting from resource extraction activities, siltation caused by poor land use practices, reductions in riparian cover and by other nonpoint-source pollutants. The species' patchy distribution limits the natural genetic exchange between and within its populations. Because of its restricted distribution and continued vulnerability to these threats, and the Service's uncertainty with regard to the viability of individual populations across the range, the Service concluded that the species continues to meet the definition of threatened (likely to become endangered within the foreseeable future throughout all or a significant portion of its range) and should remain classified as such (U. S. Fish and Wildlife Service 2015d).

1. The ultimate goal of the blackside dace recovery plan (U. S. Fish and Wildlife Service 1988) is to restore viable populations of the species to a significant portion of its historic range and then remove the species from the Federal List of Endangered and Threatened Wildlife and Plants. The species will be considered for delisting upon achievement of the following criteria: Each of the eight subbasins identified in the recovery plan has a viable population comprised of at least three protected, inhabited stream reaches per sub-basin.
2. Each of the 24 stream reaches is protected in some manner, either through public agency or private conservation organization ownership or some form of permanent easement, and a management plan has been implemented for each stream that provides for the species' long-term protection.
3. No foreseeable threats exist that would threaten survival of the species in any of the subbasins.

4. Noticeable improvements in coal-related problems and substrate quality have occurred to the species' habitat throughout the upper Cumberland River basin, and the species has responded through natural means or with human assistance to successfully recolonize other streams and stream reaches within the upper Cumberland River basin.

3. Environmental baseline

3.1. Regional context

The 1,496-acre permit boundary of the Kopper Glo Mining, LLC Cooper Ridge Surface Mine is located in Claiborne County, Tennessee approximately five miles southwest from the junction of Tennessee State Highway 90 and Valley Creek Road and the Valley Creek community (Figure 1). Much of the permit boundary was mined in the 1950's, 1960's, and early 1970's, and although little or no reclamation occurred on these areas, they have largely naturally succeeded to secondary forests. In addition, logging has periodically occurred in the same time period, and oil and gas exploration and their associated disturbances are common in this area. Roadbeds remaining from these historic land uses in the area crisscross the ridges (Office of Surface Mining Reclamation and Enforcement 2017).

The current land use within the proposed permit area is best defined as undeveloped forest appropriate for fish and wildlife habitat. The existing roads and ponds remaining from previous mining would provide most of the needed access between the mine site and public roads and some of the sediment control basins needed for the proposed mining (Office of Surface Mining Reclamation and Enforcement 2017).

Precipitation and runoff from the proposed mine site drains to Nolan Branch, Hurricane Creek, Valley Creek, and Straight Creek in the Clear Fork system, and various smaller unnamed tributaries of these streams.

3.1.1. Bats

Because Indiana bat (*M. sodalis*) and gray bat (*M. grisecens*) are documented to use caves in the region surrounding the project site for winter hibernacula, and therefore habitats in the project area could be used for roosting or swarming during the non-hibernating seasons. Therefore, applications for four recent SMCRA permits in the vicinity of the proposed Cooper Ridge Surface mine permit boundary included winter habitat assessments for potential hibernating use of caves or mine portals by bats. These surveys also included assessment of forest habitat and acoustic and mist-net surveys to identify likelihood of non-hibernating use of these permit areas by bats, especially endangered or threatened species (Aquatic Resources Management LLC 2010, Biological Systems Consultants, Inc. 2013a, Copperhead Environmental Consulting 2013a, b).

The results of these mist-netting surveys are summarized below in Table 2. None of the acoustic or mist-netting efforts documented Indiana bat occurrence in the vicinity. However, one individual gray bats (male [Copperhead Environmental Consulting 2013b]) and five total northern long-eared bats were captured in mist nets (including two non-reproductive males, a single nonreproductive female, and two post-lactating females [Aquatic Resources Management 2010, Biological Systems Consulting, Inc. 2013a, Copperhead Environmental Consulting 2013b]).

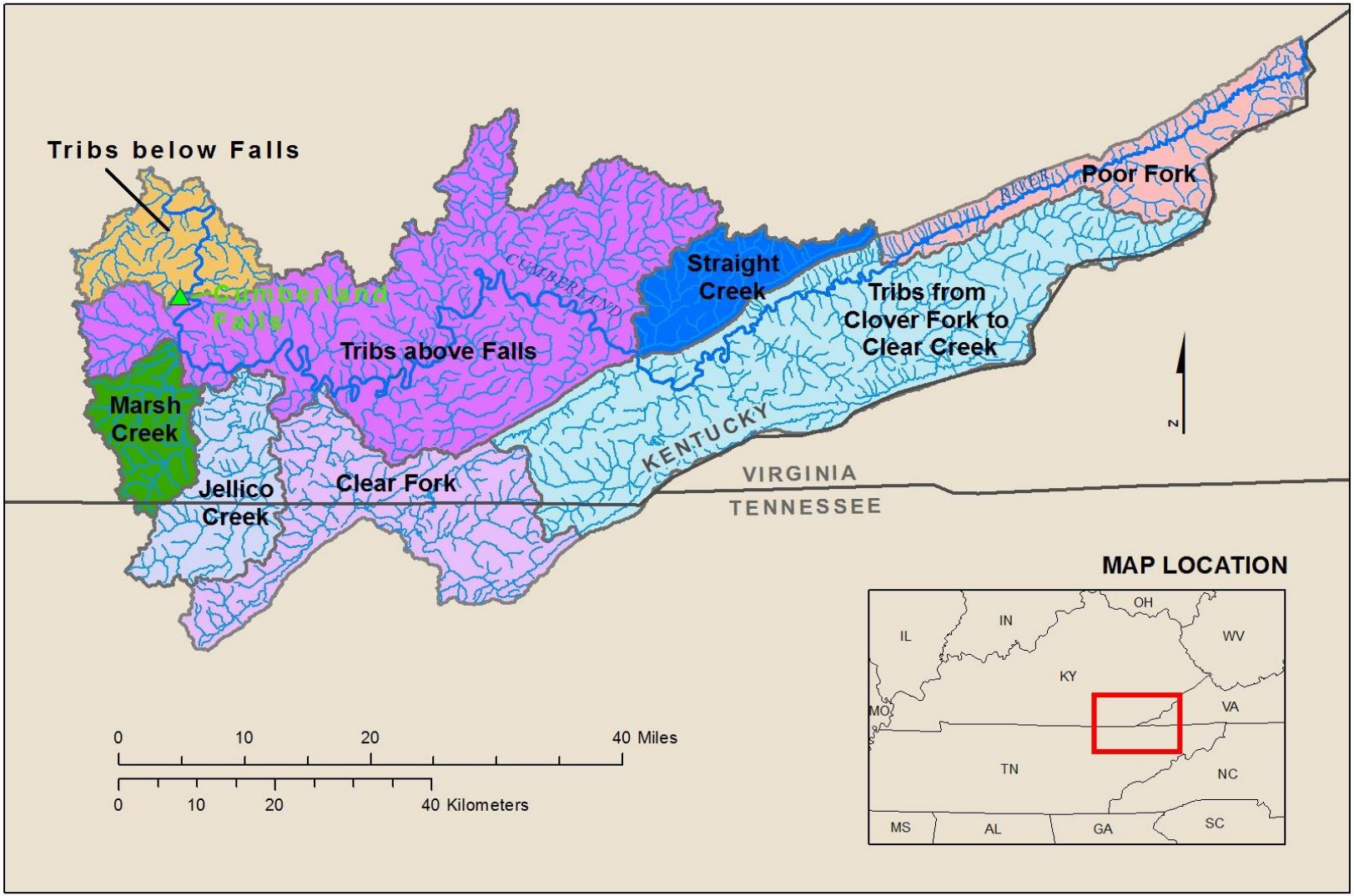

Figure 8. Blackside dace (*Chrosomus* [= *Phoxinus*] *cumberlandensis*) recovery units, as included in the blackside dace recovery plan (U. S. Fish and Wildlife Service 1988). Streams included in the action area for this are located in the Clear Fork Recovery Unit.

Table 3. Summary of recent bat surveys conducted in the vicinity of the Kopper Glo Mining, LLC Cooper Ridge Surface Mine, Claiborne County, Tennessee (OSMRE permit 3270).

Common Name	Scientific Name	Federal Status[a]	Total Captured	References[b]
Big brown bat	*Eptericus fuscus*		20	ARM 2010, BSCI 2013, CC 2013a, b
Eastern small-footed myotis	*Myotis leibii*		1	BSCI 2013
Eastern red bat	*Lasiurus borealis*		13	ARM 2010, BSCI 2013, CC 2013a, b
Gray bat	*Myotis grisescens*	E	2	CC 2013b
Hoary bat	*Lasiurus cinereus*		2	ARM 2010
Little brown bat	*Myotis lucifugus*		2	BSCI 2013, CC 2013b
Northern long-eared bat	*Myotis septentrionalis*	T	5	BSCI 2013, CC 2013b
Tri-colored bat	*Perimyotis subflavus*		3	ARM 2010, BSCI 2013
TOTAL	**8**		**47**	

[a]E – endangered, T – threatened

[b]ARM – Aquatic Resources Management, LLC. 2010.

BSCI – Biological Systems Consultants, Inc. 2013a.

Copperhead Environmental Consulting. 2013a, b.

The Aquatic Resources Management (2010) report noted that habitat appropriate for bats that use forests for summer roosts was relatively abundant in 2010. However, the other reports listed above and in Table 3 concluded that, while forested areas in the vicinity included some snags and tree species appropriate as summer roosting habitat for Indiana bat, the quality of these forested areas was considered to be marinal to poor, due to recent timber harvest.

3.1.2. Fish

Numerous fish surveys have been conducted on streams in the upper Clear Fork watershed of Campbell and Claiborne Counties over the last 45 years (Office of Surface Mining Reclamation and Enforcement unpublished data). OSMRE's data includes five surveys of the mainstem Valley Creek and tributaries beginning in 1989, with the most recent in 2013. Valley Creek would receive the majority of the discharge and runoff from the mine permit boundary.

In addition to documented occurrence of the Cumberland arrow darter (*Etheostoma sagitta*) listed by the TWRA as "deemed in Need of Management" the Tennessee Division of Natural Heritage notified OSMRE that the emerald darter (*E. baileyi*) and roseyface shiner (*Notropis rubellus*), both listed as "Deemed in need of management", and blackside dace and silverjaw minnow (*N. buccatus*), both listed by the TWRA as threatened were present within four miles of the proposed mine site (Table 4). However, recent surveys in the Valley Creek watersheds have not identified the presence of a persistent blackside dace population in the mainstem of Valley Creek.

3.2. Status of the listed species within the action area

3.2.1. Blackside dace

The blackside dace 5-year review (U. S. Fish and Wildlife Service 2015d) discussed population status based on the eight Recovery Units included in the blackside dace recovery plan (U. S. Fish and Wildlife Service 1988, and see Figures 7 and 8). This review identified 125 occupied streams in nine Kentucky, three Tennessee (Campbell, Claiborne, and Scott), and two Virginia counties occupied by the species. As discussed in *Population dynamics/status and distribution* (section 2.3.4), above, this includes blackside dace populations that have been recently discovered in three drainages that were not included in the Recovery Plan (Kentucky River and Tennessee River drainages, and tributaries to the Cumberland River downstream of Cumberland Falls, U. S. Fish and Wildlife Service 2015d).

Also, as described in *Population dynamics/status and distribution* (section 2.3.4), above, genetic analyses (Strange and Burr 1995) identified three or four blackside dace metapopulation units that are larger than the eight recovery units identified in the blackside dace recovery plan. Consequently, the blackside dace 5-year review considered the 125 occupied streams to consist of 58 isolated population "groups" that are functionally isolated.

Table 4. Summary of occurrences for fish species with protected status in streams within a four-mile radius of the Kopper Glo Mining, LLC, Cooper Ridge Surface Mine (OSMRE permit 3270). Information summarized from Office of Surface Mining Reclamation and Enforcement (2014) and provided to OSMRE by the Tennessee Department of Environment and Conservation's Division of Natural Areas (DNA). Specific information about identity of surveyors, localities, and fish species collected were not provided by DNA.

Scientific name	Common Name	State status
Arrow darter	*Etheostoma sagitta*	In Need of Management
Blackside dace	*Chrosomus cumberlandensis*	Threatened
Silverjaw minnow	*Notropis buccatus*	Threatened
Rosyface shiner	*Notropis rubellus*	In Need of Management
Emerald darter	*Etheostoma baileyi*	In Need of Management

The action area for this consultation is includes streams in the Recovery Plan's "Clear Fork" Recovery Unit (see Figures 9 and 10). And according to Strange and Burr's (1995) metapopulation analysis, action area streams in the "Clear Fork" Recovery Unit are included in Strange and Burr's Group B (see Figure 9). Strange and Burr's (1995) mitochondrial DNA analysis indicated that Group B is the center of dispersal for all blackside dace populations.

Recent surveys include 2015 surveys by students at Morehead State University and Austin Peay State University (Bedal and Cashner 2016, Bedal et al. 2016, Yates et al. 2016), specifically aimed at blackside dace populations and occurrences. Bedal (2016) surveyed streams in the Tennessee portion of the upper Jellico and upper Clear Fork watersheds for blackside dace tissue samples (Bedal et al. 2016) in order to identify blackside dace population genetic structure, using mitochondrial DNA (cytochrome b gene). Yates et al. (2016) surveyed several streams within the action area; Bedal's surveys were in the Elk Fork Creek system headwaters, and did not include any action area streams.

As mentioned in the *Threats* section (2.3.5) above, in 2015 Yates et al. (2016) surveyed some of the streams in the action area considered in this BO to independently confirm the blackside dace 343 µS/cm conductivity threshold reported by Hitt et al. (2016). Action area streams surveyed by Yates (2016) include Valley Creek (headwaters), two locations on Clear Fork (just downstream of the confluence with Valley Creek, and just upstream of the confluence with Buffalo Creek), Rose Creek, and an unnamed Valley Creek tributary downstream Rose Creek (Yates 2016).

3.2.1.1. Blackside dace status in the Clear Fork system

Jordan and Swain (1883) described blackside dace as "very abundant in the smaller streams" in the Clear Fork system. Starnes (1981) and O'Bara (1985, 1990) only observed the species in five streams (including Buffalo, Davis, Elk, and Louse creeks in the Tennessee portion of the watershed (Figure 9). Most of the Clear Fork populations observed by Starnes (1981) and

O'Bara (1985, 1990) during the late 1970's and 1980's were small, and these surveyors concluded blackside dace distribution in the drainage was limited by coal mining and logging activities. For example, the Buffalo Creek population that was considered one of the most robust Tennessee populations by Starnes (1981) was resurveyed by O'Bara (1990) and described as small, an observation which has been more recently confirmed by Biological Systems Consultants Inc. (2012a).

However, the blackside dace 5-year review (U. S. Fish and Wildlife Service 2015d) listed 17 occupied blackside dace streams in the Clear Fork watershed. Of these 17 streams, in the blackside dace 5-year review six were considered "stable", and the remaining 11 were considered "vulnerable". The headwaters of Tackett Creek (mainstem Tackett Creek) was included in the 17 identified streams, based on the collection of three individuals in 2015 by TWRA biologists (U. S. Fish and Wildlife Service 2015e). The current status and size of the recently discovered Tackett Creek headwater population is unknown, but additional surveys there are planned by TWRA and the Service (U. S. Fish and Wildlife Service 2015e). The best remaining populations within the drainage are considered to be those in Buck Creek (KY), Mud Creek (KY), Rose Creek (TN), and Terry Creek (TN) (Kentucky State Nature Perserves Commission 2010, Black et al. 2013b, U. S. Fish and Wildlife Service 2015d and unpublished data).

Action area streams in the "Clear Fork Recovery Unit" and Metapopulation Group B include the vulnerable population in Buffalo Creek and the stable Rose Creek population (Figure 9 and Table 5).

3.2.1.2. Summary of blackside dace occurrence and abundance in the action area

Parts of the Clear Fork system were identified as blackside dace protection zones in the Service's blackside dace minimum guidelines for protection and enhancement plans for blackside dace (*Phoxinus cumberlandensis*) (U. S. Fish and Wildlife Service and Office of Surface Mining Reclamation and Enforcement 2009). Physical habitat (riffle/pool ratios, watershed size, stream gradient), water quality parameters (including conductivity values identified by Black et al. 2013b, and Mattingly and Black 2005), and biological communities (fish and macroinvertebrate assemblages) within the action area have been assessed to identify streams or stream reaches that might support extant dace populations and that could be affected by the mining operations described in this BO. These surveys and assessments are documented by unpublished reports to OSMRE, U. S. Fish and Wildlife Service, or the Tennessee Department of Environment and Conservation, and include reports for a previous consultation (U.S. Fish and Wildlife Service 2016a), and include Biological Systems Consultants, Inc. (2011, 2012 a, b, c, 2013b, 2014), U. S. Fish and Wildlife Service (2015e), and unpublished data. In spite of these stream assessments and surveys, blackside dace have only been observed in two streams in the action area (Table 5).

The blackside dace occupied streams listed in Table 5 are located in the Clear Fork Recovery Unit (U. S. Fish and Wildlife Service 2015d). As noted in Table 5, the Buffalo Creek blackside dace population is considered "vulnerable" and the Rose Creek population is considered to be "stable" (U. S. Fish and Wildlife Service 2015d).

The Buffalo Creek population has been monitored since 2012 (Biological Systems Consultants, Inc. 2012b). Water quality parameters in Buffalo Creek mainstem appear appropriate for blackside dace (Biological Systems Consultants Inc. 2012b), but sediment deposition, from roads, agriculture, and beaver, has impacted physical habitat. Consequently, the majority of individuals in this Buffalo Creek population appear to be confined to upstream reaches. While the right fork of Buffalo Creek contains marginal physical habitat and water quality, the continued observation of isolated individuals indicates this area may be a refuge for the system (Biological Systems Consultants, Inc. 2012a).

Blackside dace populations in Rose Creek and Buffalo Creek would not be directly affected by effluents from the Kopper Glo Mining, LLC Cooper Ridge Surface Mine because no mine effluent drains to these streams. However, because of the potential for individuals in these populations to disperse from Rose Creek or Buffalo Creek and travel through larger affected streams (Valley Creek) to tributaries affected by mine drainage within the project area or to unaffected tributaries or mainstem areas upstream of the project area, the status of the Rose Creek and Buffalo Creek populations will be monitored over the lifetime of mining and reclamation.

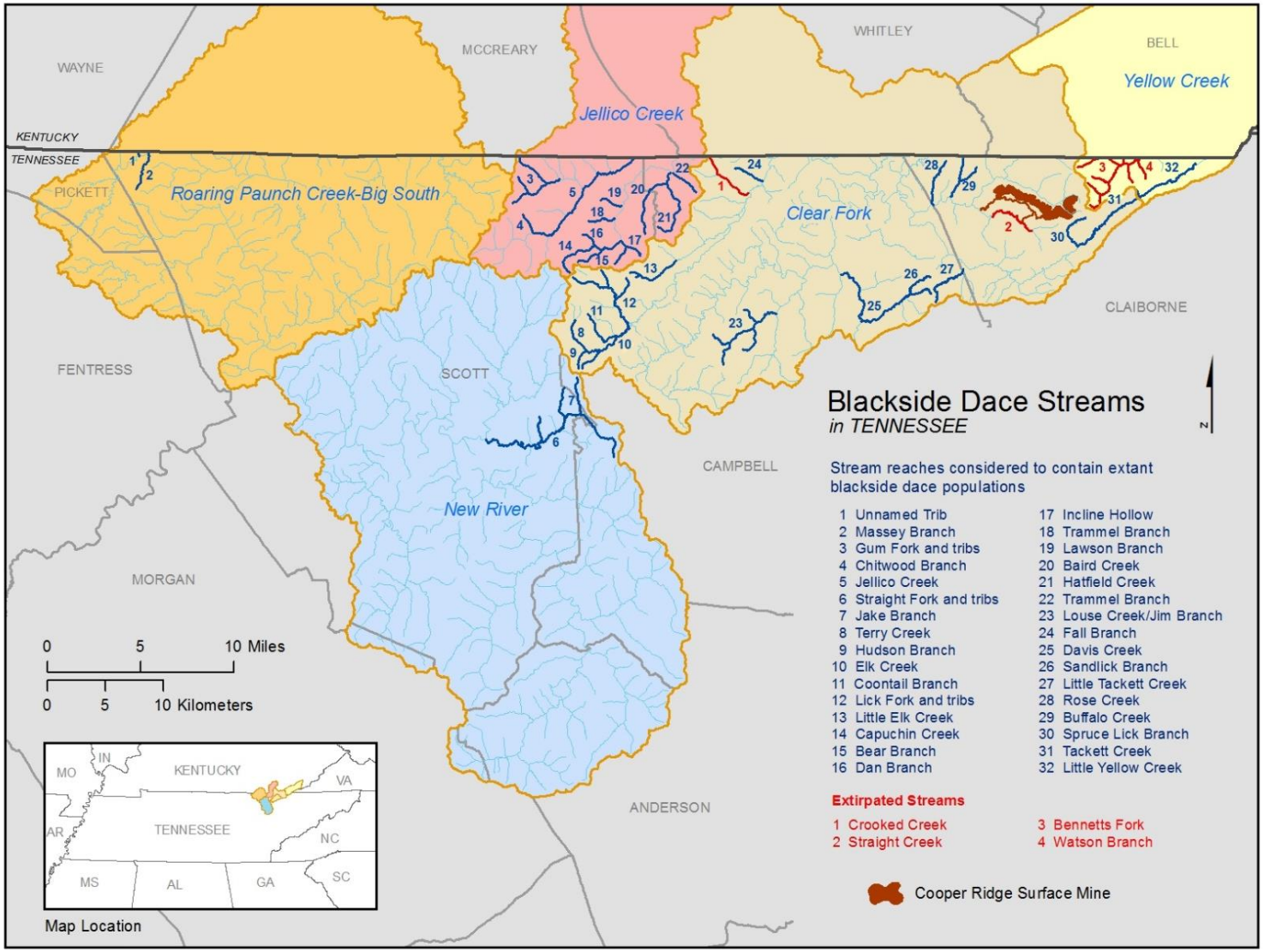

Figure 9. Tennessee streams historically and currently occupied by blackside dace (*Chrosomus* [= *Phoxinus*] *cumberlandensis*). Streams included in the action area for this consultation are located in the Clear Fork Recovery Unit (see Figures 7 and 8). Hydrologic units are identified by name and different units are indicated by different colored polygons.

Table 5. Blackside dace observations in the action area years 2000 to 2017.

Surveyors (Reference)	Date of Observation(s)	Occupied[a] Streams	Status in 2017[b]
BSCI[c] (2011, 2012a); Multi-agency[d] (U. S. Fish and Wildlife Service 2015b; Floyd 2016)	2011, 2012, 2015	Rose Creek	Stable
BSCI (2012a)	2012	Buffalo Creek	Vulnerable

[a]"Occupied" indicates at least one individual was collected or observed at some time between 2000 and present, and does not imply existence of an extant population or extent or size of any population
[b]According to *blackside dace 5-year review* (U. S. Fish and Wildlife Service 2015d)
[c]Biological Systems Consulting, Inc.
[d]U. S. Fish and Wildlife Service (KY and TN Ecological Services Field Offices), Office of Surface Mining Reclamation and Enforcement, Tennessee Wildlife Resources Agency

Table 6. Predicted water quality in action area streams receiving runoff and discharges from mining and reclamation activities. Analysis from Middlesboro Mining Operations, Inc. Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296) BO (U. S. Fish and Wildlife Service 2016a), and supplemented by more recent data from data loggers (see Appendix C) placed in Valley Creek headwaters, Valley Creek near the confluence with Clear Fork, and in Clear Fork upstream of the confluence with Valley Creek.

Stream	**Mean Specific Conductance (µS/cm)/Standard Deviation[a] Minimum – Maximum[b]**	**Predicted Specific Conductance Increase Within 95% Baseline Confidence Limit[c]?**
Valley Creek headwaters	415/97.9 **122.5** – 727.7	51.7% NA
Valley Creek below headwaters and upstream of Clear Fork confluence	407.3/85.3 **272.8** – 507.5	39.2% **NO**
Valley Creek near confluence with Clear Fork	538.2/116.3 **168.2** – 781.7	10.3% YES
Clear Fork	579.2/183.0 **112.0** – 1100.3	2.1% YES

[a]Values presented are expressed in microsiemens per centimeter, or µS/cm. Data in this column are from continuously recording conductivity data loggers placed at select locations by OSMRE May 2015 and February 2017, and include at least 30,000 individual data observations, depending upon locality and stream conditions. Periodic short-term lapses in data collection occurred as some probes were temporarily exposed by reduced flow, fouled probes, etc. See Figure 3 in this BO for locations of data loggers.
[b]Values included based on upper and lower 95% confidence intervals for each location.
[c]Conductivity threshold (343 µS/cm) from Hitt et al. (2016).
[d]Values in this column were calculated by OSMRE using a model described by Johnson et al. (2010) to quantify potential impacts of land use alterations to headwater streams. The model predicts potential changes in downstream conductivity at a fine scale, for the smaller streams receiving mine drainage. Predictions are based on an averaged effluent specific conductance value for each coal seam and mine drainage area that would be mined, and mixed with a specific conductance value from included stream reaches on a particular collection day. As a result, model predictions could change significantly, depending on conditions when specific conductance is measured. Therefore, the use of percentage increase is probably more meaningful than an absolute specific conductance values for projecting change. Highlighted values are below the 343 µS/cm change-point threshold above which blackside dace abundance is reduced (Hitt et al. 2016).

3.2.2. Indiana bat

While there have been no bat surveys specifically conducted for this proposed permit area, several acoustic and mist-net surveys have taken place in the vicinity. These include: Aquatic Resources Management, LLC (2010), who conducted mist netting two sites along Clear Fork for OSMRE 3224; Biological Systems Consultants, Inc. (Biological Systems Consultants, Inc. 2013a), who conducted acoustic and mist netting in 2013 for OSMRE 3264; and Copperhead Environmental Consulting (2013a, b), who conducted acoustic and mist netting for OSMRE 3231 and the Cooper Ridge deep mine on the 3270 permit boundary. No Indiana bats were captured during these surveys. The nearest documented occurrences of Indiana bats to the action area include a hibernaculum, Limestone Cave, in Whitley County, Kentucky, approximately three miles from the action area; several records in Lee County, Virginia, including summer roosting (non-maternity and maternity) and five hibernacula (Cudjo Cave, Cumberland Gap Saltpeter Cave, Grassy Springs Cave, and two unnamed caves, approximately 5.4 to 8.7 miles from the action area; and Jolly Saltpeter Cave in Union County, Tennessee, approximately 12 miles from the action area (Tennessee Department of Environment and Conservation 2016). In addition, Copperhead Environmental Consulting also tracked a radio-tagged female Indiana bat from New Mammoth Cave, (Kentucky) to this general area within the past decade, but they were unable to track her to her final summer roosting location (Pelren 2016).

Although the presence of Indiana bats in the action area has not been verified, OSMRE has assumed their presence near the mine permit site during the non-hibernating season (April 1 – September 30). Because suitable summer habitat for the species occurs throughout the 7,476-acre area included in this BO action area (1,496 acre mine permit boundary plus 5,980 acres included in a ½ mile buffer around the mine permit boundary), the Service believes that it would be presumptuous to conclude that the Indiana bat is not utilizing the action area for summer roosting and/or foraging; therefore, the Service assumes that the Indiana bat occurs within the action area.

3.2.3. Northern long-eared bat

Biological Systems Consultants, Inc. conducted acoustic and mist net surveys in the proposed project area in 2013 (Biological Systems Consultants, Inc. 2013a); four northern long-eared bats, three adult females (two post-lactating and one non-reproductive) and one adult male, were captured during these surveys over 20 net nights. Acoustical monitoring was also performed, which identified northern long-eared bat calls.

Because the presence of northern long-eared bats in the action area has been verified, OSMRE has assumed their presence near the mine permit site during the non-hibernating season (April 1 – September 30). Suitable summer habitat for the species (maternity roost trees and foraging habitat) occurs throughout 1,496-acre permit boundary.

3.2.4. Status of critical habitat in the action area

Of the two federally listed species covered by this consultation, the Indiana bat is the only species for which critical habitat has been designated. While the action area supports suitable

summer roosting habitat for the Indiana bat, no DCH for the species is located in the action area for this consultation. Therefore, DCH will not be discussed further in this biological opinion.

3.2.5. Factors affecting the species' environments in the action area

Mattingly and Floyd (2013) and U. S. Fish and Wildlife Service (2015d) summarized the threats which continue to affect blackside dace throughout its range in spite of state and Federal regulatory mechanisms that could potentially protect the species. Human-induced land and water use changes were included in these discussions (i.e. improper stream riparian zones, stream channel alterations, and culvert and dam passage barriers). However, lingering and persistent effects from legacy and current surface coal mining continue to affect the habitat and water quality in action area streams historically and currently occupied or transitionally used by blackside dace.

Approximately 32% (472 acres) of the total proposed project area (1,496-acre permit boundary) has been previously mined for coal 5.1 -5.4 miles of unreclaimed highwall. Spoil ridges and orphan pits, remaining from previous pre-SMCRA mining activities, have altered the surface water patterns in the action area. Surface runoff is detained in the orphan pits and is introduced into streams via road culverts, eroded windows and/or gullies. Old spoil banks, openings are often created by erosion, and water will move through the spoil bank or windows will deliberately be cut to facilitate water drainage. Surface runoff from above mined areas sheet flows across backfilled slopes into drainage ditches, which flow directly to streams. Valley Creek and portions of the Tackett Creek watershed currently receive the drainage from previous surface and underground mining activities, resulting in water quality issues, primarily elevated sulfates and specific conductance.

Previous land clearing activities have temporarily or permanently reduced the amount of suitable summer habitat available to the Indiana bat and northern long-eared bat. Much of the forestland has now gone through several successional stages and currently provides habitat for both bat species, but existing roads and other areas cleared for agriculture, residences, etc. remain and have contributed to habitat loss. In summary, historical coal mining activities and associated remnant disturbances, along with impacts caused by logging, agriculture, residential development, oil/gas disturbances (exploration and well development, maintenance and abandonment), and road building and maintenance activities, have impacted aquatic and terrestrial habitats within the action area.

4. Effects of the action

Analysis methodology

SMCRA provide a framework for reviewing applications for surface coal mining, and the 1996 nationwide programmatic BO (U. S. Fish and Wildlife Service 1996) provided additional Terms and Conditions (T&Cs) to ensure that authorizing surface coal mining by issuing SMCRA permits would not jeopardize the continued existence of listed species or result in adverse modification or destruction of designated critical habitats. In addition, the OSMRE Knoxville Field Office (KFO) established a LIWA in 2012 between state and Federal agencies that have

regulatory authority (OSMRE, the Service, USACE, EPA, and TDEC) related to surface coal mining in Tennessee (Office of Surface Mining Reclamation and Enforcement 2010). This LIWA established standard procedures for coordinated, interagency review of permit applications for proposed surface mining in Tennessee, and also attempted to simplify and streamline reviews for each agency's regulatory requirements (Office of Surface Mining Reclamation and Enforcement 2010).

The OSMRE provided to the Service a biological assessment (BA) that included the action by letter dated November 29, 2016. The scope of the BA describes the effects of the proposed surface mining and associated activities within the Kopper Glo Mining, LLC action permit boundary and various other proposed OSMRE actions to blackside dace, Indiana bat, and northern long-eared bat. We have relied on information in the EA (Office of Surface Mining Reclamation and Enforcement 2017), the permit applicaton package (Office of Surface Mining Reclamation and Enforcement 2013), the draft Cumulative Hydrologic Impact Assessment (CHIA) (Office of Surface Mining Reclamation and Enforcement 2016a) and draft Environmental Assessment (EA) (Office of Surface Mining Reclamation and Enforcement 2017) to identify and describe activities included in this consulation. In addition we used species listing packages (U. S. Fish and Wildlife Service 1987, 2015a, b), 5-year reviews and other status summaries (U. S. Fish and Wildlife Service 2009, 2015c, d), recovery plans (U. S. Fish and Wildlife Service 1983, 1988, 2007), published and unpublished scientific studies and reports (many referenced throughout this BO and others summarized in U. S. Fish and Wildlife Service 2016c), and agency unpublished data to analyze the effects of the proposed mining to these species

For each of the project components described in *Description of the proposed action* (section 1.1), we apply the following steps to analyze effects:

- **Literature review** – We review best available science and commercial information about how the activity may affect the blackside dace, Indiana bat and northern long-eared bat.
- **Stressor-exposure-response pathways** – Based on the literature review, we identify the stressor(s) (alteration of the environment that is relevant to the species) that may result from the proposed activity.
    - For each stressor, we identify the circumstances for an individual blackside dace, Indiana bat, and northern long-eared bat's exposure to the stressor (overlap in time and space between the stressor and these individuals).
    - Given exposure, we identify the likely individual response(s), both positive and negative. For this consultation, we group responses into one of five categories:
        - increased fitness (e.g., increased access to, or availability of, prey or other food);
        - annoyance (e.g., vehicle, blasting, or other disturbances in a foraging area, causing bats to forage elsewhere);
        - reduced fitness (e.g., reduced food resources, reduced suitable nesting sites for blackside dace or roosting sites for Indiana bat or northern long-eared bat);
        - harass (e.g., temporary increases in sediment in streams used for spawning causing blackside dace to move away from otherwise suitable spawning

areas or causing creek chub nest-builders to flee and not guard nests, increasing the likelihood of blackside dace egg or hatchling predation, day-time disturbance in a roosting area, causing bats to flee and increasing the likelihood of predation); and
        - harm (e.g., mine drainage water chemistry that causes death or injury to blackside dace migrating or spawning adults or eggs and hatchlings).
    - For each pathway, we consider how proposed conservation measures may reduce the severity of the stressor or the probability of an individual fish or bat's exposure (e.g., avoiding direct stream habitat impacts and minimizing indirect water quality impacts that affect blackside dace inhabiting stream reaches or traveling through stream reaches downstream in the watershed by maintaining specified vegetated buffers along all streams on the mine site).
- **Population-level effects** – For each pathway, we use available data on the number of populations and individual blackside dace, Indiana bats, and northern long-eared bats, habitat potentially occupied by blackside dace, roosting Indiana bats and roosting northern long-eared bats that might be affected in the action area by the activities included in this consultation. The species' likely sensitivity to change, resilience, and recovery rate in response to the proposed actions are considered in our estimation of these effects.

4.1. Literature review - effects of pre-mining vegetation clearing

Site preparation for mining would require vegetation removal and subsequent compacting and homogenizing of the rocks and soil during mining alters patterns of water flow through the ground and over the land's surface. Altered ground and surface water flows can lead to altered physical changes to aquatic habitats. When water flows through mined areas with reduced vegetative cover, soil nutrients can be quickly depleted. Reduced vegetative biomass and altered soils on mine sites can lead to accelerated erosion, which can alter stream habitats by depositing sediment in adjacent streams and further downstream in the affected watershed. This sediment deposition can continue until forests become reestablished on mine sites, possibly for several decades.

Removing the vegetation in preparation for mining, and subsequently compacting and homogenizing the rocks and soil during mining would alter patterns of water flow through the ground and over the land's surface. As a result, in comparison with unmined areas, ground and surface water flows, water chemistry, and physical aquatic habitats can be altered, which can have cascading effects to biological assemblages in affected watersheds downstream. When water flows through mined areas, soil nutrients are lost, and higher annual water yields from reduced vegetation biomass increases erosion and sediment deposition in adjacent and downstream stream reaches for decades (until forests become reestablished in the alkaline soils left behind by mining). However, depending on the other land uses in the watershed, the magnitude of these impacts can be diminished with increasing distance from the mine (Petty et al. 2010).

Daniel et al. (2014) analyzed fish assemblage data over three ecoregions (Northern Appalachian, Southern Appalachian, and Temperate Plains ecoregions) in the eastern United States to identify

threshold fish assemblage responses to levels of mine densities. Their data indicate a higher proportion of Southern Appalachian mines affect headwater streams, (27.75%) and creeks (21.72%) in comparison with the other two ecoregions included in their study.
Daniel et al. (2014) identified significant thresholds for more than half their tested relationships between mine density and fish assemblages at mine densities of ≤ 0.05 mines/km2, and they suggested these low density values indicate a single mine can produce threshold responses in fish assemblages. The threshold for effects to fish diversity indices was observed at a density less than 0.002 mines/km2 in all of their study regions.

4.2. Literature review - headwater streams

Most of the streams directly affected by surface coal mining in Tennessee and other areas within the Appalachian coalfields form the headwaters of stream networks, or headwaters of tributary systems to larger drainage networks. In order to understand the geographic, ecological and temporal scope of headwater stream alterations that occur as a result of mining, publications describing the variety of aquatic organisms, their movement throughout stream systems, and the roles these organisms play in processing fuel, providing genetic material, and influencing the components of the animal and plant assemblages found throughout stream systems affected by surface coal mining are summarized here.

Environmental conditions in headwater streams are harsh and temporally variable. Many affected streams are ephemeral, intermittent, or small with perennial flow. Ephemeral streams typically flow following precipitation events. Flow is periodic in intermittent streams, usually during seasons with higher precipitation. These small streams usually have small catchments where smaller water volumes and flows are easily influenced by small scale differences in local conditions. Impacts to these small headwater streams are frequently dismissed because they are dry much of the year, or because they are perceived to be relatively unimportant in comparison with other larger streams in the watershed network (Lowe and Likens 2005, Meyer et al. 2007).

4.2.1. Literature review - chemical characteristics of headwater streams

Streams in the southeastern United States typically exhibit very low levels of dissolved salts. Elevated salinity in streams has only recently been identified as a factor affecting aquatic biological community composition. Water with higher salinity (more dissolved salts) more easily conducts electrical current; and the specific conductivity measured as µS/cm is a measurement of this water quality component. Various activities on the landscape contribute a variety of salts and ions, affecting conductivity and stream water quality, including weathering of concrete infrastructure, runoff of salts from roads and other impervious surfaces, oil and gas exploration, agriculture, silviculture, wastewater and industrial effluents, and coal mining and processing (Griffith 2014).

Griffith (2014) characterized variation in relative, naturally occurring concentrations of ions in streams across Omernick's (1987) ecoregions, and provided the best available estimates of current reference conditions for comparison. He characterized streams in the eastern USA as typically below 200 µS/cm, with the median and 75th percentiles of data he analyzed being less than this level. Although Griffith (2014) indicated that many areas have already been so extensively disturbed that these estimates do not represent natural conditions, he suggested they

provided a reasonable estimate of current reference conditions for comparison with disturbed areas. The conductivity of streams in the Central Appalachians ranged from 25.0 – 103 µS/cm (Griffith 2014). This range is also supported by Wallace et al. (1992), who reported mean conductivity of Southern Blue Ridge and Southern Appalachian streams between 29.2 and 103.3 µS/cm.

4.2.2. Literature review - biological characteristics of headwater streams

Headwater streams are important links between terrestrial ecosystems and other stream reaches downstream as they periodically release large amounts of dissolved organic matter and microbes to fuel aquatic biological communities and support other ecosystem processes downstream in receiving watersheds (Bernhardt and Palmer 2011, Bessemer et al. 2013). In addition, they provide habitats for a rich array of uniquely adapted aquatic and semi-terrestrial species that are food for other species or, if migration corridors are available, are a source of colonists to enhance downstream biological diversity. Seasonally, these headwater streams offer refuges from temperature and flow extremes and shelter from predators and also are important as spawning sites and nursery areas for macroinvertebrates amphibians, and fish (Meyer et al. 2007).

In the southern Appalachians fish diversity of these small headwater streams is considered relatively low, even though biodiversity and endemism in general is among the highest in the temperate zone in the southeastern United States. The fish species that colonize and recolonize headwater streams are uniquely adapted to these environments, exhibiting life history characteristics that allow for rapid population growth (Hitt and Roberts 2012). However, even though they may be adapted to invade and rapidly expand in these headwater habitats, Etnier (1997) considered about 25% of headwater-adapted fish species in the southeastern United States to be disproportionately jeopardized because of anthropogenic impacts to their habitats and water quality.

Headwater-adapted fish species restricted to streams in the Appalachian coalfields of Tennessee are likely to be those species threatened by adverse impacts of surface coal mining. Jeopardized headwater fishes specifically identified by Etnier (1997) included minnows of the genera *Phoxinus*, which includes the threatened blackside dace, *Chrosomus cumberlandensis* (formerly considered a species of *Phoxinus*). Also, the recently listed Cumberland darter, *Etheostoma susanae* was among the headwater stream dwelling darter species considered by Etnier to be disproportionately jeopardized.

4.2.3. Literature review – surface mining impacts to headwater streams

As briefly described above, aquatic species found in the southeastern United States have evolved to survive in streams with harsh environmental characteristics that exhibit water quality parameters very different from those affected by surface mines. However, impacts may be most obvious in intermittent streams and other small (first and second order) perennial streams in upper elevations, whose catchments are small. These streams maintain smaller water volumes and flows, and are easily influenced by small scale differences in local conditions. Interactions between altered abiotic system components and the resultant changes to biological components are complex and only partially understood. However, it is clear that downstream water quality,

biological community structure, and ecosystem functions can be impacted, and multiple mining operations throughout the watershed can have cumulative impacts downstream (Bernhardt and Palmer 2011).

4.2.4. Literature review – surface mining impacts to hydrology

Grossman et al. (1998) analyzed a 10-year physical and biological data set for a Southern Appalachian stream (North Carolina) and concluded that flow variability (both peak and mean flows) had stronger effects on fish assemblages than biological interactions (competition and predation). Species that inhabit the water column were more strongly influenced by stream flow variability than benthic species occupying microhabitats on the stream bottom. Even though periods of low flow (drought) significantly decreased the amount of habitat available, fishes inhabiting the water column (like the threatened blackside dace) invaded upstream areas in the system, and increased in abundance in comparison to higher flow periods. Consequently, hydrologic impacts of surface coal mining that alter the periodicity and magnitude of flows in headwater streams can greatly affect fish assemblages throughout affected stream networks. This is most evident in streams with smaller catchments that are more strongly affected by small scale local changes in land cover.

4.2.5. Literature review – surface mining impacts to water quality

In addition to physical impacts from mining activities, sulfate loads, pH, total dissolved and suspended solids and electrical conductivity of the receiving streams can dramatically increase (Bernhardt and Palmer 2011). Crushing, compacting, and homogenizing the "waste" rock and soil (overburden) left behind when coal is removed, exposes overburden to surface run-off and increases the rate of physical and chemical weathering of the rock. Weathering generates alkaline mine drainage, typically including high concentrations of sulfate, calcium, magnesium, and bicarbonate ions, and in some areas, elevated concentrations of selenium, and trace metals. These ions and trace minerals become suspended or dissolved in the water percolating to the groundwater and running off into surrounding surface waters, increasing salinity, which is measured as conductivity (Hartman et al. 2005, Lindberg et al. 2011, Merriam et al. 2011, U. S. Environmental Protection Agency 2011, Griffith et al. 2012, Ross et al. 2016).

Because sediment ponds on mine sites are specifically intended to collect runoff from the mine site and minimize impacts to receiving streams, they can accumulate trace amounts of metals or other contaminated materials exposed during mining. Insects with aquatic larval stages that use these habitats could become contaminated by metals from mine runoff. As a result, foraging bats using the adult emerging insects as prey could potentially bioaccumulate metals, depending on the proportion of their diet that might consist of insects emerging from these areas. Although this potential is not well studied in North American bats (Salvarina 2016), altered neurochemical responses, loss of homeostatic control (Nam et al. 2012, Pikula et al. 2010, Salvarina 2016) could result.

The natural (background) level of dissolved ions in Central Appalachian streams should be less than 103 µS/cm (Griffith 2014, Wallace et al. 1992). Various land uses result in differing concentrations of salts that can raise stream conductivity, but elevated conductivity related to

coal mining in Appalachian streams with neutral or slightly alkaline pH is largely due to salts of calcium, bicarbonate, sulfate, and magnesium (U. S. Environmental Protection Agency 2011). Conductivity levels of streams affected by surface mines are frequently elevated, and the magnitude of effect can be cumulative, depending upon the proportion of watershed affected by surface mines (Johnson et al. 2010).

Elevated metal concentration in watersheds affected by surface mining can also alter water quality parameters. Several studies of aquatic community primary production (growth of plants, or algae) in streams with altered water quality suggest that altering normal interactions between naturally occurring iron and other elements may have "bottom-up" impacts that produce effects to the entire stream trophic (feeding) interactions (Tate et al. 1995, Hayer et al. 2013, Newcombe and MacDonald 1991). For example, iron is chemically important in aquatic systems because it promotes bioavailability and transport of other elements. Iron is naturally abundant. In acidic water, iron is available for biological uses. However, in basic water (pH above neutral) iron is easily oxidized, is insoluble, and can precipitate and cover the stream bottom.

Elevated levels of iron oxide in non-acidic streams can limit availability of phosphates needed for primary production of algae (including periphyton) by binding or attaching to phosphate (sorption). Phosphates are needed for plant growth, and limiting plant growth can have major repercussions on downstream aquatic communities. However, phosphates bound by iron can be released (and therefore available for plant growth) with exposure to sunlight. Therefore, phosphate availability may not be a limiting factor if affected streams have enough reaches with open canopy to allow for these chemical reactions to occur and release phosphates needed for plant growth.

Tate et al. (1995) field tested laboratory results that described interactions between elevated iron levels, phosphate availability, and exposure to light. They injected radiolabeled phosphates into a Rocky Mountain stream affected by mine drainage to identify whether elevated iron levels in streams actually would affect primary productivity by limiting phosphate availability, as suggested in laboratory experiments. In their field study, about 90% of the injected phosphates were removed from water column within a 175-m stream reach where the stream was exposed to full sunlight. They concluded that phosphate availability in streams is dynamic and is regulated by photoreduction through exposure to sunlight in areas where stream canopies are open (Tate et al. 1995).

Another way precipitating iron can affect primary production in streams is by inhibiting light penetration to the stream bottom, therefore restricting the ability of algae or periphyton in those areas to photosynthesize regardless of phosphate availability. This precipitating ("fouling") iron can also physically interfere with oxygen absorption and food consumption of macroinvertebrates, and fishes, especially aquatic insects that feed by scraping or collecting food from the stream bottom (Merrit and Cummins 2000, Hayer et al. 2013). Dalzell and MacFarlane (1999) also indicated that extremely high levels of iron can accumulate on the gills of brown trout.

In addition to the biological impacts resulting from altered water quality briefly mentioned here, other biological effects have been noted. While the mechanisms for these observations are not

completely understood, some of the possibilities are suggested in *Literature review – impacts of altered ground and surface water chemistry to downstream biological assemblages* section (4.1.8) in this BO.

4.2.6. Literature review – impacts of surface mining to the specific conductance component of water quality

Specific conductance (also referred to as conductivity) is a measurement that indicates how easily water will conduct electricity, and ths ability depends upon the amount of dissolved salts, or ions, in the water. However, the measurement, recorded in µS/cm, doesn't identify which specific ions are present to cause the observed conductivity; the conductivity measurement is directly related to the amount of TDS in water. Only relatively recently has the impact of landscape activities that increase the level of ions that raise salinity in streams been identified as a factor affecting aquatic community composition, particularly in the southeastern United States, where streams typically exhibit very low levels (as measured by specific conductivity). In fact, specific conductance isn't a parameter that has been routinely assessed or reported in results of aquatic community surveys. However, as more data are made broadly available, and more sophisticated tools are used to analyze large datasets, the potential use of this water quality parameter in predicting biological community impacts from various land uses is becoming more obvious. For example, in an assessment of a wide variety of landscape and watershed attributes, land cover types and water quality, Alford et al. (2015) reported that specific conductivity was among five parameters that correlated with fish assemblage structure in five tributary streams and 11 mainstem sites in the Nolichucky River watershed in Tennessee.

Griffith (2014) summarized publications identifying anthropogenic contributions to increased salinity (via contributions from a variety of ions) in aquatic environments, measured as conductivity, including road salts, infrastructure (concrete) weathering, oil and gas exploration, agriculture, silviculture, wastewater and industrial effluents, and coal mining and processing. Griffith's (2014) intent was to characterize variation in relative, naturally occurring concentrations of ions in streams across Omernick's (1987) ecoregions and provide the best available estimates of current reference conditions for comparison. He characterized streams in the eastern USA as typically below 200 µS/cm, with the median and 75th percentiles of data he analyzed being less than this level, and specifically conductivity in the Central Appalachians of 25.0 – 103 µS/cm. Although Griffith indicated that many areas have already been so extensively disturbed that these estimates do not represent natural conditions, he suggested they provided a reasonable estimate of current reference conditions for comparison with disturbed areas.

The effects of elevated conductivity to aquatic invertebrates include altered community composition, reduction in species richness, changes in relative abundance, and total elimination of some taxa (Green et al. 2000, Chambers and Messinger 2001, Yuan and Norton 2003, Pond 2004, 2010, Hartman et al. 2005, Pond et al. 2008, U. S. Environmental Protection Agency 2011). Specifically, Pond et al. (2008) demonstrated that mayflies (order Ephemeroptera) are disproportionately extirpated from areas with elevated conductivity. These sensitive aquatic insects that typically make up large proportions of the macroinvertebrate communities in unaltered Central Appalachian streams are adapted to water quality conditions where ionic

concentrations are orders of magnitude lower than those that can result from surface surface mine runoff.

Bauer et al. (2016) used fish and crayfish occurrences and water quality and landscape variables to produce a model to guide restoration of streams on the Appalachian Plateau in northern Alabama. Conductivity explained 38% of the existing fish biomass, nearly as much as that explained by drainage area. Drainage area was an important explanation for the variation in fish species richness, diversity, community evenness, fish abundance, and fish biomass, but as mentioned above, conductivity explained nearly as much as that explained by drainage area.

Hitt et al. (2016) analyzed water quality and fish data from a variety of surveys 2003 and 2013 where blackside dace were collected (294 samples). Their analysis indicated stronger effects to abundance from conductivity than other natural and anthropogenic factors (watershed size, land use, water temperature). A segmented regression analysis identified change-points in conductivity of 343 µS/cm for blackside dace (95% confidence intervals 124-627 µS/cm. Abundances of this fish were negligible where conductivity measurements were above these levels.

The conductivity change-points for blackside dace and Kentucky arrow darter are close to the EPA's 300 µS/cm (U. S. Environmental Protection Agency 2011) benchmark recommended as protective of 95% of Appalachian headwater stream benthic macroinvertebrate communities, and while consistent with Black et al. (2013a), the Hitt et al. (2016) blackside dace threshold is higher than the 240 µS/cm threshold reported by Black et al. (2013a). Hitt et al. (2016) concluded that differences may be related to the Black et al. (2013a) model that included changes in fish occurrence, not abundance, and the Hitt et al. (2016) model also included abundance.

A variety of studies have been conducted to model (Bryant et al. 2002), predict (Mount et al. 1997), or test toxicity (Tietge et al. 1997, Merricks et al. 2007, Singleton 2000) of the major ions that contribute to elevated salinity (conductivity) in streams draining mined areas levels of ionic strength but levels of conductivity that would consistently affect wide ranges of taxa have not been identified. And, as Freund and Petty (2007) suggested, correlations between chemical parameters and biotic indices may not be appropriate to infer causes of observed impairment, especially when the correlations include potential interactions between multiple water quality stressors, even if the levels of independent constituents may be lower than established water quality criteria. While Freund and Petty's (2007) research focused on stream habitat and water quality linkages related to acid mine drainage, their observation that fish community response to water quality degradation was noted at relatively low stressor concentrations, and that nearly all the minimum threshold stressor concentrations identified as important in affecting fish community composition were at concentrations (primarily metals, including aluminum, iron, and nickel) significantly lower than current water quality criteria.

However, in spite of these identifying a causal relationship for the impact of conductivity levels to aquatic community composition, the EPA (U. S. Environmental Protection Agency 2011) recommended a benchmark of less than 300 µS/cm. This level was identified as that benchmark that would avoid extirpation of 95% of native invertebrate species in parts of the Central Appalachians. This benchmark was derived using field methods that assess various life stages

and ecological interactions between species in aquatic communities, conductivity measurements, resulting from a mixture of alkaline effluents (U. S. Environmental Protection Agency 2011). While this benchmark was specifically derived for West Virginia and Kentucky in Omernick's (1987) Ecoregions 68, 69, and 70, and the applicability of the benchmark elsewhere has not been tested, the EPA (U. S. Environmental Protection Agency 2011) expected it to also be appropriate for these ecoregions in Tennessee. It is important to note that this 300 µS/cm benchmark is well above the 25.0 – 103 µS/cm baseline levels identified for Central Appalachian streams (Griffith 2014), and presumably conditions to which native stream biota have evolved specific adaptations.

More recently, Cormier et al. (2013) used field data sets to identify whether individual component ions or a mixture (measured as specific conductivity) of the ions that affect water quality of streams downstream of coal mining cause the observed extirpation of benthic macroinvertebrates reported by (Pond 2010, Pond et al. 2008). Their study area included the Southwestern and Central Appalachian and Western Alleghany Plateau, ecoregions 68, 69 and 70 (Woods et al. 1996). Their assessment supported a causal relationship with conductivity and the extirpation of 40 and 46 genera in West Virginia and Kentucky, respectively.

Pond et al. (2008) commented that native Appalachian stream invertebrate taxa are likely more sensitive to the pollutants (ions) used in toxicity tests than surrogate test organisms, leaving open the possibility of ion toxicity. While not specific to ion toxicity, Freund and Petty (2007) also supported the conclusion that laboratory toxicity testing with established procedures and test subjects may not provide water quality criteria sufficient to protect aquatic life.

However, several possible mechanisms have been suggested to explain the observed impacts of elevated conductivity to aquatic invertebrate communities identified above. The most likely explanation is the inability of mayflies and other sensitive invertebrates to regulate their bodies' fluid balances that result from increased salinity in their fluid environments, which affects their ability to perform physiological processes required to survive (Pond et al. 2008, Komnick 1977, Gaino and Rebora 2000). Cormier et al. (2013) discussed the likely physiological mechanisms that can be affected when aquatic animals are exposed to different concentrations of dissolved ions, and they concluded that some genera are consistently less and others consistently more tolerant of increased ionic concentrations. They commented that if a species cannot physiologically compensate for increased salinity/conductivity, they must migrate away from the stressors, or die because their cells and organs are unable to exchange the ions and fluid needed to maintain appropriate internal fluid balances required to complete various life processes.

Pond et al. (2008) noted that some benthic invertebrates exposed to increases in salinity/conductivity may avoid exposure to this stressor by increasing drifting behavior that moves them downstream in the water column. However, as reported by Wood and Dykes (2002) and Blasius and Merritt (2002) this observed effect was not consistent across taxa. Pond et al. (2008) also noted that this avoidance behavior should not preclude the drifting taxa from recolonization vacated areas. And finally, Pond et al. (2008) suggested other, indirect effects from elevated metal concentrations as an explanation for observed extirpations or reduction in abundance. However, neither assessments reported by Pond et al. (2008) or Pond (2010) indicate strong correlations between metal concentrations in the water column and declines or

extirpations or other indirect impacts from metals, such as precipitates that alter benthic habitats or dietary impacts from metal incorporation into the food web. Pond's (2010) assessment found the strongest correlation with mayfly abundance and specific conductance.

Few studies have demonstrated toxicity of conductivity, or of the various components (sulfates) that cause the elevated conductivity downstream of mines to aquatic species. However, the typical test subjects are hardy organisms that may not reflect sensitivity that has been demonstrated by other taxa. Also, as Pond et al. (2014) suggested, toxicity testing may identify physiological impacts (e.g., death) to test animals that cannot move away from testing treatments, but behavioral effects (e.g., avoidance of inappropriate water quality, via drifting) that may take place in streams where animals are exposed to these conditions are not documented by these tests.

Likewise, Hitt et al. (2016) mentioned that seven day lab toxicity experiments did not indicate mortality of juvenile blackside dace in reconstituted waters representing conditions found in mining-impacted and reference sites. However, Combs (2016) did observe some sublethal histological effects.

In addition, in 2007 and 2008, a group of experts, led by researchers at the University of Georgia, created a structured decision model describing the most up to date ecological knowledge about the blackside dace (McAbee et al. 2013). Decision analysis is a useful tool to support the recovery process because it provides users with a means to formalize relationships between variables, sources of uncertainty, and management outcomes in quantitative models (Peterson and Evans 2003, Shea et al. 2014), and outcomes can guide future management decisions and scientific research. The blackside dace model constructed by McAbee et al. (2013) focused on human and environmental stressors, ecological system components, and management outcomes of interest, and was then evaluated via sensitivity analysis, determining the relative influence of various inputs, actions, and variables on forecasted outcomes (McAbee et al. 2013). Sensitivity analysis and scenario building demonstrated that mining practices were predicted to be the most influential input, while other inputs seemed to have less substantial impacts (McAbee et al. 2013). McAbee et al. (2013) indicated that the smaller influence of other factors may indicate that blackside dace are a robust species to certain stressors, even in combinations, and that while the influence of stream conductivity on blackside dace presence has empirical support from habitat modeling (Black et al. 2013b), the underlying ecological cause is largely unknown. The combination of high influence on outcomes and little empirical data suggested that effects of conductivity warranted future investigation.

As mentioned above TDS is a water quality parameter that measures all of the dissolved substances in water that is related to conductivity, and neither parameter identifies which specific substances are present in the water that results in specific measurements. Pond et al. (2008) summarized toxicity testing McCulloch et al. (1993) noted that high levels of TDS was not considered a human health issue, but recognized that freshwater organisms exposed to high TDS levels must be able to compensate for the osmotic stress to their fluid balances for successful growth, survival, and reproduction. Therefore, McCulloch et al. (1993) described tests to identify levels of TDS toxic to freshwater organisms. However, as typically used in setting standards, the test organisms used are relatively tolerant species that are readily available for

laboratory use, including water fleas, or cladocerans (*Ceriodaphnia dubia*, and *Daphnia pulex*) and the fathead minnow (*Pomephales promelas*). McCulloch et al. (1993) did identify TDS levels that were acutely toxic to the test organisms, but there were differences in effects to the tested species; fathead minnows were more tolerant than water fleas, and *Ceriodaphnia* were the most sensitive. While direct toxicity may not have been clearly demonstrated, and the mechanism of effect remains unclear, various statistical and empirical studies suggest a conductivity measurement of 300 µS/cm as a threshold to protect 95% of invertebrate taxa (U. S. Environmental Protection Agency 2011).

Recent studies have further recommended conductivity thresholds for some species of fish. It is evident from the scant available evidence to date that there are species-specific differences in tolerance. As identified above, Black et al. (2013a) produced a model that indicated 240 µS/cm specific conductivity as a threshold predictor of blackside dace occurrence, where blackside dace were less likely to occur and persist above this level. Jansch's (2015) model confirmed that specific conductivity at 80% of streams where more than 25 blackside dace individuals had been collected in her dataset was below the threshold identified by Black et al. (2013a). Hitt et al. (2016) analyzed blackside dace and Kentucky arrow darter (*Etheostoma spilotum*) data and reported conductivity change-points for blackside dace and Kentucky Arrow darter abundances as 343 and 261 µS/cm, respectively, close to the 300 µS/cm USEPA recommended benchmark (U. S. Environmental Protection Agency 2011). Abundances of either fish species were negligible above these respective levels. Hitt et al. (2016) suggested several reasons to explain the differences in their 343 µS/cm blackside dace threshold in comparison with the Black et al. (2013a) 240 µS/cm blackside dace level. These include Hitt et al.'s (2016) inclusion of changes in blackside dace abundance, more recent data and model design that assessed continuous variation in response to conductivity gradient were additional variables not covered in the Black et al. (2013a) investigation.

Hitt et al. (2016) included water quality and fish survey data from a variety of surveys 2003 and 2013 where blackside dace were collected (294 samples). Their analysis indicated stronger effects to abundance from conductivity than other natural and anthropogenic factors (watershed size, land use, water temperature). A segmented regression analysis identified change-points in conductivity of 343 µS/cm for blackside dace (95% confidence intervals 124-627 µS/cm. Abundances of this fish was negligible where conductivity measurements were above these levels. As mentioned earlier in the *Habitat characteristics and use* (section 2.3.3), conductivity was included as one of the four most important variables explaining blackside dace occurrence probability (Hitt et al. 2016), where probability decreased as conductivity values increased from the 13 µS/cm minimum observation to approximately 400 µS/cm.

The conductivity change-points for blackside dace and Kentucky arrow darter are close to the 300 µS/cm EPA benchmark recommended as protective of 95% of Appalachian headwater stream benthic macroinvertebrate communities (U. S. Environmental Protection Agency 2011), and while consistent with Black et al. (2013a), the Hitt et al. (2016) blackside dace threshold is higher than the 240 µS/cm threshold reported by Black et al. (2013). Hitt et al. (2016) concluded that differences may be related to the Black et al. (2013a) model that included changes in fish occurrence, not abundance, and the Hitt et al. (2016) model also included abundance.

Hitt et al. (2016) suggested that the mechanism of effect and also the different conductivity change points for the fish species (blackside dace and Kentucky arrow darter) they analyzed may be related to impacts on aquatic community components, and consequently the prey base, for these fishes. The total biomass available as food for fishes likely does not change in relation to water quality changes likely because species more tolerant of conductivity increases and replace less tolerant species. However, mayflies are particularly sensitive to conductivity increases (Pond et al. 2008, Pond 2010, Merriam et al. 2011), likely because of their inability to adapt physiological mechanisms for osmoregulation and exchanging ions across gill membranes (McCulloch et al. 1993).

Hitt et al. (2016) concluded that stream conductivity is an important predictor of imperiled fish occurrence and abundance in Appalachian streams. Similar to the theory that iron concentrations may limit phosphorus availability for primary production, resulting in cascading effects throughout the food web in streams with elevated iron, disproportionate effects of elevated conductivity to different fish species may result from the effects of elevated conductivity to various components of the aquatic community and preferred prey items of different fishes (see further discussion in *Literature review - headwater streams* (section 4.1.1) above. Hitt and Chambers' (2014) report of disproportionate declines of obligate invertivotrous fishes declining in comparison with fishes with other feeding strategies across mining-influenced conductivity gradients supports this hypothesis.

One theory of blackside dace persistence in streams affected by mine drainage may relate to indirect impacts of water quality on food availability. As discussed above in *Literature review – surface mining impacts to water quality* (section 4.1.6), "bottom up" impacts of iron in regulating primary production, theoretically, phosphorus availability and primary productivity (periphyton) in streams or stream reaches with elevated iron concentrations, and also shaded by intact riparian zones and stream canopy, may result in cascading effects to grazing insect larvae and other macroinvertebrates. Consequently, appropriate types and amounts of blackside dace food (periphyton and aquatic insects) may also be limiting.

While photoreduction of streams with high levels of iron oxide may provide adequate levels of phosphorus for primary production in some stream reaches (Hayer et al. 2013), blackside dace are typically found in stream reaches with closed canopy (Black et al. 2013a) where photoreduction seems less likely to occur. Consequently, activities that alter the baseline water quality conditions under which blackside dace have evolved (such as that discussed in *Literature review – chemical characteristics of headwater streams* (section 4.1.2) and documented by Hayer et al. 2013) to the extent that primary productivity is decreased may affect the ability of juveniles to establish fat stores and adequate body condition that ensures survival through harsh winters. Altering the species composition and abundance of the blackside dace's preferred summer (periphyton) food may also result in diminished nutritional value to the dace. Likewise, adults in altered streams may not overwinter with adequate body condition that allows sufficient energy for successful reproduction.

Hitt et al. (2016) suggested the differential effect of water quality (conductivity) to blackside dace persistence, in comparison with Kentucky arrow darter (*Etheostoma spilotum*), relates to blackside dace feeding behavior which includes switching from macroinvertebrate prey during

winter to detritus and periphyton during summer, while Kentucky arrow darter's strictly insectivorous diet consists primarily of macroinvertebrates, including mayflies (U. S. Fish and Wildlife Service 2015f).

4.2.7. Literature review – impacts of altered ground and surface water chemistry to downstream biological assemblages

In contrast with un-mined headwater streams in forested areas which periodically and seasonally receive organic input (i.e., leaves, woody debris) from surrounded vegetated areas, streams draining reclaimed mine sites likely contribute organic matter in the form of algal species occupying grassy reclaimed environments to downstream reaches in the affected watersheds. However, while biological impacts of mining have been reported, studies to identify specific factors responsible for observed biological impacts are ongoing.

In an early study comparing fish and diatom species in streams affected by surface coal mining and similar, undisturbed streams Vaughan (1979) surveyed streams in the New River drainage of the Cumberland River system in Tennessee. Vaughan's data indicated decreasing population sizes and species diversities for both fish and diatom assemblages with increasing proportions of the study's watersheds disturbed by mining. He reported significantly reduced diatom and fish species diversity in streams in watersheds affected by surface mining and noted complete absence of some (darter) fish species groups, especially benthic insectivorous feeders. Streams in unaffected watersheds contained more than twice the diatom species and, based on the relative amount of effort needed to collect fishes, fish populations in unaffected streams were six to 20 times greater. Specifically, Vaughan commented that the more sensitive species of diatoms and fishes (darters) had either been completely eliminated, or the numbers of individuals of sensitive species (and therefore equitability between assemblage components) were reduced in the mine-affected watersheds. Vaughan concluded that, because his study streams were alkaline, and not affected by acid runoff, the likely stressor producing the observed results was suspended solids and sediment deposition.

Verb and Vis (2000) compared the benthic diatom assemblages in ten second-order Ohio streams, including streams within 1.5 km downstream of areas unaffected by mining, areas reclaimed prior to SMCRA enactment, areas reclaimed according to SMCRA regulations, and areas affected by acid mine drainage from abandoned coal mines. Specific conductance, temperature, pH, oxidation/reduction potential, sulfates, and turbidity were among the suite of water quality parameters recorded when the monthly biological collections were made. Verb and Vis (2000) noted that while the SMCRA sites surveyed had significantly higher species richness and more closely resembled unaffected streams than the other study streams, diatom species diversity and density were not significantly different.

Several other studies (Tate et al. 1995, Hayer et al. 2013, Newcombe and MacDonald 1991) of aquatic community primary production (algae) in streams with altered water quality suggest that altering normal interactions between naturally occurring iron and other elements may have "bottom-up" impacts that produce effects to the entire stream trophic (feeding) interactions. Iron is chemically important in aquatic systems because it promotes bioavailability and transport of other elements, including those needed for primary production (phosphorus). Iron is abundant

and easily oxidized (in water with pH above neutral) or reduced (in acidic waters). High levels of insoluble ferric iron (oxidized iron) may have direct and indirect adverse effects to aquatic communities by binding to phosphorus (needed for plant growth), or by altering bottom habitats used by aquatic animals by covering their bottom habitats with thick layers of precipitates thus inhibiting light penetration to these areas. These "fouling" precipitates can also interfere with oxygen and food consumption of macroinvertebrates (especially those species categorized into functional feeding groups described by Merrit and Cummins (2000) as scrapers or collector-gatherers) and fishes (Hayer et al. 2013). Dazell and MacFarlane (1999) also indicated that extremely high levels of iron can accumulate on the gills of brown trout.

Phosphate (PO_4) concentrations can impact nutrient pathways in aquatic communities, so factors limiting phosphate availability can have major repercussions on downstream aquatic communities. Elevated levels of iron oxide, often seen in streams draining mined areas, can limit availability of phosphates needed for primary production of algaes, periphyton, etc., by sorption (attaching to). However, bound phosphates can be released if iron oxides are exposed to sunlight (photo-reduced iron oxide releases sorbed solutes). So phosphates may not be limiting, if affected streams have enough reaches with open canopy to allow for these chemical reactions.

Tate et al. (1995) tested the results of laboratory experiments that described interactions between elevated iron levels, phosphate availability, and exposure to light to see if these laboratory results would also be seen in the field. They injected radiolabeled phosphates into a Rocky Mountain stream affected by acid mine drainage to identify whether elevated iron levels in streams actually would affect primary productivity by limiting phosphate availability, as suggested in laboratory experiments. In their field study, about 90% of the injected phosphates were removed from water column within a 175-m stream reach where the stream was exposed to full sunlight. So in exposed areas, phosphate availability is dynamic, regulated by photoreduction.

In summary, while the studies briefly described above suggest some possible mechanisms to explain changes in aquatic biological community assemblages in streams draining surface mined areas that have been observed by previous researchers, few studies have investigated the complex interactions between tropic levels in affected systems. In one example, Hayer et al. (2013) examined the relationships in aquatic food webs in four second-order circumneutral (pH) South Dakota streams with varying levels of iron, by comparing algal biomass, aquatic macroinvertebrate abundance, and fish relative abundance. This study confirmed Hayer et al.'s hypothesis of "bottom-up" effects to the affected aquatic communities likely resulting from lower primary production (algal biomass) in streams with higher iron concentrations.

Relatively recent studies indicate that elevated conductivity alters aquatic invertebrate community composition, reduces species richness, changes the relative abundances of species in assemblages, and totally eliminates some taxa (Green et al. 2000, Chambers and Messinger 2001, Yuan and Norton 2003, Pond 2004, 2010, Hartman et al. 2005, Pond et al. 2008, U. S. Environmental Protection Agency 2011). Conductivity is a measurement of the ability of water to conduct electricity, which depends upon the concentrations of various types of salts, and their component ions, in solution. Consequently, it is logical that effects of conductivity to stream organisms would vary depending upon the specific compositign of the ions causing elevated conductivity.

As summarized above, coal mining impacts are most noticeable in intermittent streams and other small (first and second order) perennial streams in upper elevations, whose catchments are small and thereforve easily influenced by small scale differences in local conditions. However, alterations in stream water chemistry and aquatic habitats can have cascading impacts to aquatic assemblages downstream in affected watersheds. Interactions between altered abiotic components and the resulting effects to biological components are complex and only partially understood. However, it is clear that water quality, biological community structure, and ecosystem functions downstream can be impacted, and multiple mining operations throughout the watershed can have cumulative impacts downstream (Bernhardt and Palmer 2011).

Because of their size and volume, relative to larger streams, the changes to hydrology and water quality described above can disproportionately affect small, headwater streams, and the animals found there, including blackside dace. Etnier (1997) considered about 25% of headwater-adapted fish species in the southeastern United States to be disproportionately jeopardized because of anthropogenic impacts to their habitats and water quality. Taxa specifically identified by Etnier include minnows of the genera *Phoxinus*, *Rhinichthys*, *Hemitremia* and some darters (including Cumberland darter, *Etheostoma susanae*).

4.2.8. Literature review - effects of altered ground and surface water chemistry to fish dispersal and genetic variation of metapopulations

Many fish species that inhabit small, headwater streams are believed to persist with the majority being relatively immobile, and a smaller group dispersing to other areas (Freeman 1995, Smithson and Johnston 1999, Rodriguez 2002). In fact, the ability of the species to maintain long-term persistence in a watershed may depend upon the ability of the species to move from areas where habitat is temporarily or seasonally inadequate, such as intermittent stream reaches, to other areas with adequate conditions. Individuals might move in order to find a higher quality or quantity of resources (dissolved oxygen content, stream flow, or food availability, for example). In order to find these appropriate habitat reaches, those dispersing individuals may need to traverse through areas typically not inhabited in order to invade, or reinvade, other areas with appropriate habitat.

Albanese et al. (2004) studied spatial and temporal movement over a three-month period of a variety of small stream fishes within a four km reach of watershed in Virginia that included headwater tributaries and main stem stream reaches in the watershed. The goal of the mark-recapture research was to develop predictive models to identify how fish species respond to rapidly changing environmental variables in relation to their varied life history strategies.

Mountain redbelly dace (*Phoxinus oreas*), a relative of blackside dace that inhabits small tributary streams was included in the Albanese et al. (2004) study, because this species exhibited broad variation in movement rates within the study area. They noted mountain redbelly dace movement positively correlated with flow events, water temperature, and day length, and suggested the observed upstream movement may be a mechanism to explore newly available areas for spawning, as these upstream, flow-related movements took place during spawning season. Hitt and Roberts (2012) studied fish community changes in a watershed over a much

longer time period (60 plus years). They also noted that the direction of movement for colonizing (or recolonizing) new areas within their study watershed was not limited to downstream.

Albanese et al. (2004) commented that patterns of increased movement might reflect resource (flow, dissolved oxygen, etc.) availability and/or quality that changes seasonally. The authors cautioned that fishes whose movements are triggered by changes in flow or temperature may be especially vulnerable in watersheds that are hydrologically or physically altered by human activities.

A study reported by Albanese et al. (2009) indicates that populations of rare species recover more slowly, even with relatively small disturbances or in areas where disturbances are frequent or chronic. Further, permanent elimination of some species from a watershed can result, without careful attention to migration barriers in stream corridors within the watershed.

Walker et al. (2012) studied southern redbelly dace (*Chrosomus erythrogaster*) populations (another blackside dace relative), in Ozark headwater streams. They conducted a mark and recapture study in a small watershed that contained stream reaches with perennial flow supported by springs, and other, intermittent stream reaches where habitats periodically consisted of isolated pools with no connectging flow. They concluded that, while the majority of the tagged dace were considered to be "residents" who did not move from their original capture location in the system, the tendency to move varied. Some individuals who remained sedentary for several sample periods moved considerable distances (> 4,000 m or ~ 2.5 mi) during later sampling periods. The proportion of the population that moved varied from 16% to 43%, depending on the season. Most of the recaptured dace moved short distances, to adjacent pools, but others moved further. The maximum movement distance observed in their study was 5,791 m (approximately 3.5 miles), throughout the entire study reach; the authors noted that maximum distance moved could have been farther. However, this possibility wasn't confirmed by recapturing tagged individuals because they did not sample beyond the downstream limit of the watershed study area.

Walker et al. (2012) and Gebhard et al. (2015), who reported that banded sculpins (*Cottus carolinea*) temporally switched dispersal patterns between stationary and mobile movers commented that upstream movement of adults that was noted in winter and early spring (prior to spawning) might be an attempt to take advantage of opportunities when higher flows provide more connections throughout watersheds to allow individuals to explore for more suitable areas to colonize. However, this behavior might also result in those individuals being exposed to less favorable conditions (i.e., isolated pools in intermittent stream segments) during other seasons.

Katz et al. (2014) completed a fine-scale population genetic study of a small, non-migratory species that inhabits small, headwater tributaries. They analyzed population genetics to identify movements and dispersal of the yellowfin shiner (*Notropis lutipinnis*) through a single watershed where the maximum geographic distance for movement through the watershed was approximately 65 km (>40 stream mi). Like blackside dace, yellowfin shiners inhabit first and second-order streams, are relatively short-lived (maximum lifespan of four years), and are nest associates (also like blackside dace) by laying their eggs in nests constructed by bluehead chubs (*Nocomis leptocephalus*). Also similar to blackside dace, yellowfin shiners had not been

previously documented in larger mainstem stream reaches in the studied watershed, and was believed to exhibit restricted movement patterns.

Katz et al.'s (2014) study indicated that habitats in the mainstem that were thought to be inappropriate for yellowfin shiners did not act as a barrier to the species' migration among tributaries, and that the species' persistence throughout the watershed is maintained by a "stepping-stone" model of dispersal. This model involves individuals moving from a tributary through the mainstem to establish new populations or bolster existing populations in other tributaries throughout the system. Their persistence in the watershed is likely sustained because of connectivity within the watershed (tributaries to the mainstems to other tributaries), and factors that could restrict or obstruct movement (physical or other barriers) leave isolated populations extremely vulnerable to extirpations related to environmental stochastic and demographic processes because the species may lack the genetic ability and resulting flexibility to adapt to changing conditions.

Katz et al. (2014) emphasized that maintaining connectivity between small, headwater tributaries and mainstem reaches within a watershed may be essential to allow fish populations in these areas to survive environmental and demographic alterations. Small populations isolated by physical or other barriers (water quality) may not have the genetic flexibility that would allow them to adapt to changing environmental conditions. These authors recommended that identifying populations vulnerable or resilient to local extirpation events as well as tributaries or habitats within a watershed that may act as sink or source populations by disproportionately contributing genetic material that is ultimately lost. Also, source populations that are useful in augmenting robust neighboring populations should be identified as well.

Labbe and Fausch (2000) studied Arkansas darter (*Etheostoma cragini*) occurrence in various spatial scales (pools, reaches, segments, and watershed, ranging from meters to 50 km) to recommend management strategies for the species' persistence. This study suggested that traditional small-scale population management may be ineffective in the long-term. They suggested that effective conservation efforts should recognize the ephemeral nature of local populations, and instead key ecosystem processes and adequate dispersal pathways should be identified and protected that allow populations to establish (or re-establish) and persist with little human intervention. However, in the short-term, Labbe and Fausch (2000) suggested that protecting local populations and some management actions (reintroductions to establish new populations) may be necessary.

Labbe and Fausch (2000) also highlighted the importance of adjacent ephemeral habitats to the species' reproduction and growth, and suggested that the species' persistence within the watershed may be dependent upon source populations that produce emigrants to other areas, some of which may function as "sinks". However, while sinks may appear detrimental to the species' persistence the varying conditions in these newly colonized areas, they may be important in the watershed or temporal scale. For example, areas invaded may not provide conditions adequate to sustain large populations, but conditions there may result in faster growth rates for fewer individuals that are better equipped to survive harsh winter conditions because of higher fitness. These individuals, which may appear to be waifs, or even detrimental to overall populations because they "land" in areas where they may appear to be "genetic sinks", might be

extremely important to the species' persistence at a larger temporal and geographical scale, if the species occurrence follows a "stepping-stone" pattern (Katz et al. 2014).

Albanese et al. (2009) removed fishes (including a close relative of blackside dace, *Chrosomus oreas*) from headwater tributary and a mainstem stream and monitored the species that colonized these areas over a two-year period. They concluded that, while recovery of an overall community assemblage may occur following an event that eliminates fishes from a portion of a watershed, populations of rare fishes takes a longer time to be restored, or restoration may not occur at all. The results of this study indicate that rare and less mobile species are more vulnerable to extinctions in watersheds, especially areas where water or habitat quality or hydrologic impacts that could result in periodic extirpations are frequent.

These studies indicate that, while there may not be large numbers of blackside dace individuals that move very frequently or far within the watershed, factors which affect the species' ability to use stream corridors for dispersal for colonizing new areas or recolonizing historically or periodically occupied areas may affect the species' ability to persist in a watershed and ultimately recover from its present "threatened" status. Rakes (2015) reported that the current understanding of population genetics has led to recommendations to ensure that only one or two effective (successfully reproducing) individuals from another population are added to populations being reintroduced every year or two. This recommendation is used in attempts to reintroduce populations of rare, short-lived rare fishes in the southeastern U. S. Conservation or management plans (including Protection and Enhancement Plans and Species Specific Protection Measures) appropriate for rare species like blackside dace must include an understanding of the species' ecology (including movement and metapopulation dynamics) and consider those factors that are important to ensure persistence of a population of rare fish in a watershed and also to support the recovery of those species.

In addition, species persistence in watersheds may be more complex and may be dependent on the composition of local fish assemblages. For example, Hitt and Roberts (2012) suggested that nest-building fishes like chubs of the genus *Nocomis* should be considered keystone species because they likely facilitate colonization by nest-associate species which may significantly influence stream fish communities over time. The presence of these nest-building chubs in these stream communities can mitigate effects to clean benthic spawning habitat that is scarce because of sediment deposition from variouis landscape impacts. To some extent, these nest builders may provide resiliency from disturbances by constructing and maintaining spawning mounds that are also used by blackside dace (and other nest associates). In addition, thes nest associations also may allow for colonization (or recolonization) of areas with otherwise unsuitable local environmental conditions (Johnston 1999). However, reliance on presence of these keystone species render the fish nesting species associates more vulnerable to factors that affect the nest associate.

In fact, Hitt and Roberts concluded that over the 60-plus years between the fish community surveys, persistence of a species at a survey locality was most strongly associated with whether or not the species used spawning mounds created by *Nocomis* chubs, with 27% average increase over time by species known to be chub nest associates in proportion of sites occupied, compared with only 8% increase of species not known to be associated with nest-building chubs.

Specifically, mountain redbelly dace (*Chrosomus oreas*) exhibited the greatest colonization, increasing occupancy in streams by 57% over the study period reported by Hitt and Roberts (2012).

In summary, the studies discussed above indicate that fish movement, including blackside dace, throughout a system could vary seasonally, in direction of movement (upstream or downstream), and by age class (adults and/or juveniles). Additionally, the studies suggest that identifying and protecting robust, stable blackside dace populations is necessary to ensure the species' persistence in a stream system. These robust populations and refugia likely serve as sources of individuals that migrate and colonize or recolonize to establish populations in other areas of a stream system where habitat is appropriate.

Migrating individuals might travel through habitats and stream types not typically temporarily or permanently inhabited by the species. Migration corridors should not contain physical or other (water quality) barriers that would preclude migrating fish from colonizing appropriate habitats within the watershed.

And finally, the discussion above may suggest that larger-scale watershed survey efforts may be more appropriate, rather than the typical monitoring requirements to monitor only specific streams or stream reaches affected by mining. In watersheds containing blackside dace populations considered critical to the species' recovery, potential adverse water quality impacts of proposed mining (conductivity), even in distant, upstream reaches of these watersheds must be carefully considered and stringent measures included in permit conditions would be appropriate to avoid these impacts.

4.3. Literature review – active mining, effects of transporting and using heavy equipment, materials, and personnel to, from and on the mine site

Studies suggest that bats avoid noisy areas. There is evidence to suggest that increased levels of noise and light may have a negative effect on foraging bats (Murphy et al. 2009). Bennett and Braun (2004) indicated that noise associated with tree cutting can cause an Indiana bat to flush, which could result in harm or harassment of the bat by altering its normal behavior pattern and possibly result in mortality. Heavy construction equipment typically generates an estimated noise level of approximately 85 decibels adjusted (dBA) at 50 feet from the source (U. S. Department of Transportation, Federal Highway Administration 2013). Assuming that the average noise level produced during construction in the action area would be approximately 85 dBA at 50 feet from the source and noise decreases by approximately five dBA per doubling of distance from source over soft ground with heavy vegetative ground cover (U. S. Fish and Wildlife Service 2013), construction noise of 85 dBA could occur from anywhere within the 1,080.8-acre project area and travel up to 3,200 ft (including beyond the mine permit boundary) before the distance traveled by the noise reduces it to 50 dBA (i.e., reaches attenuation).

Garner and Gardner (1992) reported that loud noise can disturb Indiana bat maternity colonies and found that female bats in Illinois used roosts no closer than 1,640 feet from paved roadways. If pregnant females are required to search for new roosting habitat due to disturbances, it is assumed such effort would place additional stress on them at a time when fat reserves are low or

depleted, and they are already stressed from the energy demands of migration (U. S. Fish and Wildlife Service 2007).

OSMRE (U. S. Department of Interior Office of Surface Mining 2014) identified amibient noise levels on the sourthern portion of the Cumberland Plateau in Tennessee at 35-40 decibels (dB). In this study, noise levels from mining activities, including equipment operation and coal transportation, were estimated for five hypothetical mines at ten representative sites. Noise level increases were expressed as average A-weighted sound level during a specified period of time (in this study ten hours) at the ten sites and varied from 0 dB to as much as 16 dB. When combined with the estimated noise levels for this rural area (35-40 dB), maximum noise levels were in the range of 51 to 56 dB. The American National Standards Institute indicates that yearly average noise levels of 55 dB are compatible for neighborhood parks and 60 dB for wildlife and recreation areas (U. S. Department of Interior, Office of Surface Mining Reclamation and Enforcement 1986).

In 2011, OSMRE funded a noise study related to coal mining in the North Cumberland Wildlife Management Area. Using 55 dBA (the threshold for human impacts), the study found that the area of impact for a large surface mine was 268 acres (Ambrose et al. 2012). Thus within an area of 268 acres surrounding the noise source on a large surface mine, humans could be affected by exceeding a noise level generally accepted to represent an “annoyance” threshold. The study noted that the size of the impact area depended on the topography of the mined area which influenced attenuation rates of mining sounds, and that noise impacts due to surface contour coal mining were normally temporary in any given area (generally less than one year).

Because surveys do not indicate Indiana bats use habitats in the permit boundary, and northern long-eared bat habitat would have been removed from within the permit boundary during the non-roosting season (per the conservative application of Indiana bat and northern long-eared bat PEP to the project’s mine plan), any noise impacts to Indiana bats would be indirect, if the species were using habitat in areas adjacent to the permit boundary for roosting. Therefore, indirect effects to these bats that could occur as a result project-related noise disturbances could affect Indiana bats and/or northern long-eared bat during their summer occupancy include: (1) disturbance of maternity roosts (unknown roosts), resulting in females reabsorbing embryos or spontaneous abortion (Tigner and Stukel 2003), and (2) disturbance of maternity roosts (unknown roosts) resulting in abandonment of non-volant pups (Tigner and Stukel 2003, Bat Conservation Trust 2016, U. S. Department of Agriculture, Forest Service 2016).

4.4. Literature review - reclamation, longevity of effects from altered ground and surface water quality

The Forestry Reclamation Approach (FRA) is a recommended reclamation practice that resulted from decades of research (Burger and Zipper 2002, Burger et al. 2005), intended to improve the likelihood that reclamation of mined sites would undergo natural succession and eventually result in high value hardwood forests. Soil compaction from traditional reclamation practices resulted in higher runoff rates in mined watersheds during heavy rain events, in comparison with unmined watersheds. The FRA included specific recommendations to improve soil compaction. Comparable faunal communities are expected to follow establishment of native plant

communities, which is supported by some small mammal, reptile, and amphibian studies (Chamblin et al. 2004, Larkin et al. 2008, and Carrozzino et al. 2011).

Bernhardt and Palmer (2011) discussed currently accepted restoration practices, especially in the context of mine reclamation. They commented on the lack of data demonstrating that ecological processes (decomposition of organic matter, and microbial, primary, and secondary production) are restored when implementing currently accepted stream restoration practices (i.e., Rosgen-type methods). Bernhardt and Palmer (2011) emphasized that these accepted practices fail to consider water quality parameters (sulfate and other ions that contribute to elevated stream conductivity) resulting from coal mining. They concluded that present reclamation practices have not been shown to appropriately offset or reverse chemical and hydrologic stream impacts, or compensate for ecosystem services lost as a result of surface mining impacts. Bernhardt and Palmer (2011) summarized scientific evidence that mining can alter the ability of affected areas to provide ecological functions, at least for the long-term and possibly permanently. Impacts of surface mining far upstream within upper watersheds can affect stream water quality, community structure, and ecosystem functions downstream, and the impacts of multiple mining operations in watersheds can have cumulative impacts on downstream rivers. In summary, Bernhardt and Palmer (2011) suggested that, because high levels of conductivity and sulfate concentrations can persist long after mining ceases, there is no evidence that recovery of streams affected by surface mining is possible.

Pond et al. (2008) commented that little biological recovery was noted at sites they surveyed six to seven years following reclamation, and also referenced reports of continued high conductivity and impaired ecosystem function (including occurrence of mayflies, insects of the order Ephemeroptera) 15 years following mining (Simmons et al. 2008, Merricks et al. 2007). Pond et al. (2008) concluded that water chemistry likely limited biological recovery of aquatic communities. And Pond (2010), presented a broad analysis of mayfly assemblages in perennial headwater streams (3.1 km^2 average catchment area) in eastern Kentucky affected by coal mining and residential land uses. In headwater reference streams, these insects typically comprise 25-50% of the total macroinvertebrate abundance and 20% of taxa collected in riffle samples of headwater reference streams (Pond 2010) and play important ecological roles in stream ecosystems. The study found mean mayfly richness and abundance to be higher in reference streams than in mined streams, and relative mayfly abundance was strongly (negatively) correlated with coal mining and residential land use stressors in the region. Pond (2010) concluded that recovery of extirpated mayfly communities is uncertain, as some affected streams drained areas contour mined 20-years prior.

Zipper et al. (2011) discussed success of restoring ecosystem services on the nearly 60,000 ha (nearly 150,000 acres) affected by coal mining since SMCRA was enacted. They concluded, that traditional mine reclamation practices encouraged by regulatory policies created conditions poorly suited for forest restoration, in spite of the FRA established in 2004 and routinely included in reclamation plans since its establishment. Zipper et al. (2011) identified 438 ha reclaimed (1,082 ac) using the FRA approach in TN between 2006 and 2009 and an additional 1,473 permitted ha (3,640 ac) that incorporated FRA in reclamation plans.
Zipper et al. (2011) concluded that many older, reclaimed mine sites maintain a legacy of non-native invasive plant species, and while some native species easily re-invade mined sites, other

native species with poor dispersal abilities apparently do not. The placement of loose spoil included in FRA reclamation practices results in increases TDS in mine effluents because water is encouraged to move into and through the spoils, picking up the high levels of TDS. It is yet unclear whether time, gravity, and eventual consolidation of the loosely placed rock and soils used in the FRA approach, coupled with natural vegetative succession, might counteract the adverse water quality impacts that have been unintended side-effect of the loose compaction associated with FRA.

The water quality and cascading biological impacts resulting from soluble salts weathering and leaching from exposed rock was not well understood when the FRA was developed. This weathering essentially dissolves mineral salts from the spoils, contributing proportionately higher levels of dissolved solids from mined landscapes to downstream water quality than would result from unmined soils. As noted above in Literature review – surface mining impacts to water quality (section 4.1.6), the recent linkage of TDS, and their proxy measurement of soluble salts, electrical conductivity, have recently been linked to impaired biological communities (Pond et al. 2008). This linkage has resulted in a recommended conductivity bencmark for water discharges (U. S. Environmental Protection Agency 2011) believed adequate to prevent loss of 95% of the native aquatic species in Central Appalachian streams. However, the EPA (U. S. Environmental Protection Agency 2011) noted that this benchmark was not protective of all genera and species, including sensitive taxa and stream reaches with high quality or exceptional waters designations.

Zipper et al. (2011) suggested that placing lower-TDS spoil materials on reclaimed surfaces would be expected to reduce the amount of high conductivity water leaving reclaimed mine sites. They also suggested analyses of the runoff water quality would be appropriate, and if needed, other practices used to isolate acidic spoil materials during reclamation (Skousen et al. 1987) could be used to limit water movement into and through the higher-TDS spoils below the surface layer. However, Zipper et al. (2011) commented that FRA restoration practices and ensuring protection of water quality may not be compatible.

Fruend and Petty (2007) compared results of multimetric fish and macroinvertebrate indices that measure biotic integrity of stream fish and macroinvertebrate assemblages. They discussed use of these indices in assessing impacts to these aquatic assemblages from mining-related water quality stressors, and especially the utility of these indices as measures of restoration success. They noted the importance of dispersal within drainage networks in influencing fish assemblages, and commented that fish composition within given stream segments may be shaped by regional as well as local conditions. As a result, in spite of appropriate local water and habitat quality, influences far upstream in drainage networks can strongly affect fish assemblage compositions.

Fruend and Petty (2007) suggested that simply meeting existing water quality standards for dissolved aluminum and iron are insufficient to demonstrate recovery of streams affected by mining because aquatic biological assemblages appear to be affected by multiple stressors, frequently when none of the constituents exceed water quality criteria. They strongly suggested that mine remediation programs include the full suite of mining-related chemical constituents affecting streams, including acidity, sulfates, heavy metals and trace metals. Cormier et al.

(2013) implicated conductivity as the cause for extirpation of benthic macroinvertebrates in mined streams. They commented that habitat scores at reclaimed mine sites included in their assessment were similar to unmined reference sites where biological metrics were significantly different, which implies habitat restoration is insufficient to restore biological assemblages if conductivity remains high.

Zipper et al. (2011) concluded that neither productive forests nor other managed and economically viable land uses have been established on most of the 600,000 ha affected by coal mining since SMCRA. Further, they suggest that new research is needed to revise the FRA and integrate practices that protect water quality, restore appropriate hydrology, and ensure adequate soil nutrients to support long-term productivity for more complete native plant community restoration. Presently, in addition to the water quality concerns identified above, the expectations of FRA (establishment of native productive forests with comparable faunal communities) have not been realized. It is worth noting that although research leading to development of the FRA was available by the early 1990's, and additional cost of using FRA reclamation was either non-existent or modest, the FRA approach was not widely applied until 2004. For this reason, Zipper et al. (2011) suggested that any future changes to FRA address the issues discussed above may require regulatory intervention to become widely practiced.

The increases in concentrations of sulfate and other ions in streams draining mined areas that result from continued physical and chemical weathering of exposed mining overburden can impact water quality with elevated sulfate loads, pH (where alkaline overburden is exposed), total dissolved and suspended solids, and consequently, electrical conductivity, of receiving streams continue long after upstream mining activities have ceased also continue to affect biological assemblages. For example, Stair et al. (1984) compared creek chub populations (a relatively tolerant species) in one undisturbed stream and streams draining three mined areas in the New River basin in Tennessee. One of the study streams had been affected by mining for more than 25 years, one for three years, and the third had only been affected for one year; the percent of watershed areas disturbed by mining were approximately 19%, 24%, and 10%, respectively. The stream affected by 25 years of mining (and 19% of watershed area) was a second-order stream, and the others were all first order streams. Because the mines weren't known to produce acid runoff, the intent of the study was to identify whether sediment runoff from the mines affected creek chub growth, population structure, and food habits in the affected streams. All fishes were more numerous in the undisturbed stream, and creek chubs were three times as abundant there as in the streams affected by mining. Growth rates of creek chubs in the undisturbed stream were lower than those in mined streams, although the ones found in the undisturbed stream were in better condition (they had a higher proportion of body fat) than those in mined streams. Even though the aquatic food source was more plentiful, more fish competing for the available food in the undisturbed stream likely resulted in the lower growth rate compared to mined streams (Stair et al. 1984).

Creek chubs are opportunistic and have jaws that allow for easy capture of terrestrial insects following emergence in summer when they are abundantly available as fish forage. However, creek chubs also eat aquatic insect larvae when terrestrial emergent insects aren't available. In the mined study streams Stair et al. (1984) reported that creek chubs depended more on terrestrial insects for food, as there was less aquatic food available per individual. This may have been

related to reduced visibility and consequent reduced ability of the creek chubs to capture their aquatic prey. Alternatively, prey (benthic macroinvertebrates) availibity to fishes fishes in the mine-affected streams could have been lower because of the heavy silt load and increased turbidity. No young-of-year were collected in the mining-disturbed streams (Stair et al. 1984). The authors concluded that increasing stream flow may have pushed smaller individuals downstream into receiving streams, and also increased stream flow and movement of eroded substrate may have destroyed spawning habitat and eggs or fry.

It has been estimated to take at least 25 years (Lambert et al. 2004), and possibly a century (Gerke et al. 1998) for water quality, specifically the elevated conductivity that results from the weathering of overburden, to diminish following mine disturbance. However, these estimates of impact recovery within 25 to 100 years are based on assumptions of no additional mining impacts in affected watersheds during the recovery period (Ross et al. 2016), which is unlikely. Ross et al. (2016) compared the three-dimensional physical landscape alterations that take place during valley fill surface mining, discussed the impacts of these alterations to downstream water quality, and attempted to more accurately estimate the longevity of mining impacts. While the analysis reported by Ross et al. (2016) applies to valley fill mining, a surface mining practice not allowed in Tennessee, applicable conclusions to their assessment include the lack of available data to accurately predict how long mining-related water quality impacts (elevated conductivity) will persist in watersheds.

Pond et al. (2014) corroborated Ross et al.'s conclusion about persistence of conductivity, and provided data to demonstrate equally long-term ecological impacts downstream of valley fills. They compared benthic macroinvertebrate communities, water chemistry, and some physical features in reference headwater (first and second order) streams to similar streams draining surface mines (valley fills) where reclamation had taken place 11 to 33 years earlier in spite of habitat that appeared appropriate. They reported significant differences between reference and reclaimed streams in conductivity levels and macroinvertebrate assemblages, including relative abundance of scrapers and shredders. They also noted that sensitive mayfly and caddisfly taxa present in reference streams were absent in reclaimed streams.

The studies summarized above indicate that the impacts of mine disturbance to aquatic assemblages have previously been underestimated, and that mine impacts cumulatively affect regional biological assemblages within larger stream networks. In addition, these mine-related impacts may last for decades or centuries. The success of accepted reclamation practices in restoring ecosystem services and appropriate biological assemblages has not been demonstrated.

4.5. Analysis of effects

4.5.1. Pathway 1

Activity – mine site vegetation clearing

Vegetation would be removed in areas where active mining would occur to facilitate surface mining and to construct and provide access for monitoring and to perform necessary maintainance to sediment pond structures. Logging would be carried out to remove forest cover and facilitate mining, including removal of approximately 472 forested acres (Office of Surface Mining Reclamation and Enforcement 2013, 2017).

Stressor

- Removing vegetation could remove non-hibernating (roosting) habitat potentially used by Indiana and northern long-eared bats if any individuals would attempt to use forested areas on the mine permit boundary by removing the potential roost habitats. Removing vegetation and altering permeability of soils where roads are present would result in increased erosion and runoff that could affect composition and abundance of dissolved and suspended organic matter in streams receiving this runoff, increase sediment deposition in stream habitats used by benthic organisms, and alter permeability and evapotransporation rates of soils, thus changing the watershed's hydrologic characteristics, with more extremes in flow rates, volumes and periodicity. These changes in watershed habitat quality, quantity, and hydrologic patterns, would alter benthic community assemblages in streams used temporarily or permanently by blackside dace. However, proper design and placement of sediment ponds would minimize these impacts by capturing sediment in runoff from the mine site and also storing water to reduce the high flows from precipitation events and other hydrologic impacts in watersheds receiving drainage from the mine.

Exposure (time)

- Physical alteration of the forested habitats potentially used by Indiana and northern long-eared bats for non-hibernation (roosting) would be affected during the non-hibernating season (spring through fall) and the alteration would continue to affect Indiana and northern long-eared bats potentially using the deforested areas on the mine permit boundary until successful reclamation has resulted in forests with trees sufficiently large enough to provide the preferred habitats for each. The amount of time to achieve appropriate forest habitats to provide these habitats could be 75-100 years. Physical, and hydrologic characteristics of those streams closest to the surface mining would be affected year-round; the altered stream hydrology and effects of increased sediment in runoff to stream habitats would be gradually reduced following successful reclamation and vegetation growth on the mined site, which could affect blackside dace for 5-10 years after mining had been completed (up to 20 total years from the time vegetation clearing for mining would be initiated, assuming 10 years of mining).

- **These impacts could directly affect Indiana bats and northern long-eared bats and indirectly affect blackside dace**.

Exposure (space)

- Forested areas on the mine permit boundary that could be used by Indiana bats for summer roosting, and all aquatic habitats temporarily or permanently occupied by blackside dace (i.e., small streams in the affected watersheds that might be temporarily or permanently occupied by blackside dace and larger streams in the affected watersheds that might be used as travel corridors to more permanent habitats) would be affected. Small streams in closer proximity to the mine site would be more strongly affected by hydrologic impacts; pools or slower moving stream habitats further downstream in the affected watersheds would be more strongly affected by sediment deposition than other habitats (riffles and runs) where sediment runoff from the mine site would be swept away by current.

Resource affected

- Forested areas on the mine permit boundary containing trees and forests with habitat characteristics appropriate for use by non-hibernating Indiana and northern long-eared bats for roosting. All occupied and transitory stream habitats used permanently by blackside dace for feeding, breeding, sheltering, and traveling to other such areas in watersheds draining the mine site could be affected.

Individual response

- Indiana or northern long-eared bats using trees on the forested portion of the permit boundary for roosting could be affected and might flush (move) from their roost trees if they were present when trees being removed to facilitate sediment pond construction or active mining. Blackside dace inhabiting affected streams could be affected by increased sediment deposition, if their eggs are smothered by silt or if nest-building fishes (most likely chubs in the genus *Nocomis*) are unable to find adequate nest rocks in silted stream bottoms to construct nest mounds that would be used by blackside dace for spawning in their nests. Altered hydrology could result in periodic low flows that could also inhibit blackside dace reproduction if eggs spawned by blackside dace were desiccated because of low flows. Adult blackside dace could be stranded in streams whose flow were reduced because of altered hydrology, which impacts stream flow. Blackside dace may be unable to find sufficient amounts of food because sediment deposition could affect community composition of benthic food organisms such that adequate amounts of periphyton and/or aquatic insects appropriate for blackside dace diet would not be available; reduced fitness might result in lowered survivorship as a result of reproduction or surviving through stressful winter conditions. Exposure to altered habitat quality, inadequate food supply, and subsequent reduction in fitness could affect the success of migrating blackside dace in attempts to invade or reinvade other appropriate habitats elsewhere in -affected watersheds.

Interpretation

- Removal of forested areas potentially used by Indiana bat and northern long-eared bat could reduce fitness for bats that were disturbed or flushed from trees on the permit boundary because they were required to find other appropriate habitat in the vicinity. Moving to other suitable roost areas could result in reduced fitness because of the energy required to find adequate roosting habitat elsewhere in the vicinity. This impact would

only be possible for the first two years of the permit authorization, because after March 31, 2019 any tree removal needed to facilitate the mining would be prohibited when bats are not in hibernacula and could be using forested areas for roosts. Altered habitat quality as a result of increased sediment in runoff from cleared land surfaces (i.e., reduced vegetative cover) and changes in hydrology in small tributary streams in turn decreases the likelihood of blackside dace persistence due to reduced reproductive success in occupied streams receiving mine drainage and decreases opportunities for blackside dace expansion to other small streams in affected watersheds. Altered habitat quality in affected tributary streams also reduces opportunities for blackside dace individuals to migrate through the watershed to contribute to gene flow, supporting long-term persistence of the metapopulation throughout the affected watershed.

4.5.2. Pathway 2

Activity – transport of heavy equipment, materials, and personnel to and from the mine site

While all of the roads used to access the mine site are existing, and presumably periodically used by vehicles accessing the area for various reasons; heavy equipment, materials, and personnel delivery to the mine site would increase vehicle traffic to and within the site.

Stressor

- Noise from vehicle and other equipment use could affect Indiana bats occupying adjacent forested areas during the non-hibernating season.

- **These impacts could indirectly affect Indiana bats and northern long-eared bats.**

- Soils on roadbeds would alter permeability of soils where roads were present and because these compacted areas would not absorb runoff from the areas exposed for surface mining during precipitation events, increased erosion and runoff could affect levels of dissolved and suspended organic matter in streams receiving this runoff, increase sediment deposition in stream habitats used by benthic organisms, and could alter permeability and evapotransporation rates of soils, thus changing the watershed's hydrologic characteristics, with more extremes in flow rates, volumes and periodicity. These changes in watershed habitat quality, water quantity, and hydrologic patterns, could alter benthic community assemblages in streams used temporarily or permanently by blackside dace.

- **These impacts could indirectly affect blackside dace.**

Exposure (time)

- Noise from vehicle and other equipment use to, from, and at the mine site would affect Indiana and northern long-eared bats using adjacent forested areas for roosting when these activities took place during non-hibernating season (April 1 through mid-October) for the length of active mining (5-10 years).

- Physical, and hydrologic characteristics of streams nearer to the surface mining would be affected year-round, but the altered stream hydrology and effects of increased sediment in runoff to stream habitats and would be gradually reduced following cessation of mining and successful reclamation and vegetation growth on the mined site. However, although the impacts would gradually be reduced with increasing vegetation cover over the mine site, the adverse impacts could continue to affect blackside dace for five to 10 years after mining had been completed (or up to 20 total years, assuming 10 years of mining).

Exposure (space)

- Noise from vehicle and other equipment use to, from, and on the mine site would affect roosting Indiana and northern long-eared bats using adjacent forested areas within a buffer area ½ mile around the mine-site boundary during the non-hibernating season (April 1 through mid-October, depending on annual weather patterns in the mine area and at hibernacula), but the disturbance would gradually move from one area or "cut" being actively mined within the mine permit boundary to other adjacent areas such that bats in forests adjacent to the entire mine site would not be affected by noise throughout the duration of mining and reclamation.

- All aquatic habitats temporarily or permanently occupied by blackside dace could be affected (i.e., small streams in the affected watersheds and larger streams in the affected watersheds that might be used as travel corridors to more permanent habitats). Small streams in closer proximity to the mine site would be more strongly affected by hydrologic impacts as would habitats (pools and slower moving stream reaches) impacted by sediment deposition.

Resource affected

- Roost trees for Indiana and northern long-eared bats in adjacent forested areas (unknown) within ½ mile distance of the mine site.

- All occupied and transitory habitats used permanently by blackside dace for feeding, breeding, sheltering, and traveling to other such areas.

Individual response

- Indiana and northern long-eared bats roosting in adjacent forested habitat within ½ mile distance of the mine boundary could be roused from daytime sleep by increased noise of vehicles traveling to, from, or on the mine site. This could result in reduced fitness by causing unnecessary energy expenditure by waking bats.

- Increased sediment deposition may affect successful blackside dace reproduction, if eggs were smothered by silt or if nest-building fishes (chubs) were unable to find adequate nest rocks in silted stream bottoms to find adequate rocks for constructing nests that would be used by blackside dace spawning in their nests. Altered hydrology could result in periodic low flows that could affect successful blackside dace reproduction if eggs spawned by blackside dace were exposed to dessication because of low flows, or strand adult individuals. Blackside dace may be unable to find sufficient amounts of food because sediment deposition has affected community composition of benthic food

organisms such that adequate amounts of periphyton and/or aquatic insects appropriate for blackside dace diet were not available. Reduced fitness may result in lowered survivorship from reproduction or surviving through stressful winter conditions. Exposure to altered habitat quality, inadequate food supply, and subsequent reduction in fitness may affect success of traveling blackside dace in attempts to invade or reinvade other appropriate areas.

Interpretation

- Loss of suitable Indiana and northern long-eared bat roosts in adjacent forested areas within ½ mile distance of the mine boundary could cause disturbance by rousing roosting bats, which could decrease the bats' fitness, but this effect would be temporary, as equipment causing the disturbance would move within the mine permit boundary when mining was completed within various cut areas.
- Although roads on the mine site would likely receive increased traffic over the amount presently occurring on existing roads, SMCRA regulations include BMPs designed to minimize the impact of increased erosion and also hydrologic impacts (see section 1.4.6.5), and upgrading the existing roads to SMCRA standards would improve runoff from the area over the baseline conditions. Altered habitat quality and/or quantity (i.e., periodic lower flows) in small tributary streams could decrease blackside dace fitness because of sediment deposition and changes in hydrology. Also, opportunities for blackside dace expansion to other small streams in the watershed could be reduced, including ability to find and use areas that might serve as refugia from poor conditions elsewhere in the watershed. Altered habitat quality in affected tributary streams could also reduce opportunities for individuals to travel through the watershed to contribute to gene flow that supports the long-term persistence of the metapopulation throughout the affected watershed.

4.5.3. Pathway 3

Activity – sediment pond construction and maintenance

Sediment ponds are required by SMCRA to control sediment runoff from surface mine projects. In addition, the quality of discharges is regulated by TDEC, when specific parameters of discharges from these structures are included as permit conditions on National Pollutant Discharge Elimination System permits (NPDES) issued for individual mines. The compliance with these permit conditions is regularly monitored by OSMRE and TDEC inspectors. During reclamation, these sediment ponds would be converted to wetlands for wildlife use as watering and feeding areas.

Stressor

- Aquatic insects could be affected by toxicity from metals (e.g., iron, manganese, aluminum) from mine runoff if consumed as prey by Indiana and northern long-eared bats that may feed over sediment ponds. After mining and reclamation has been completed and forest re-growth has been successful in the reclaimed area, the conversion of ponds to wetlands would benefit Indiana and northern long-eared bats that might be roosting in the area by providing water sources and also sources of prey.

- **These impacts could indirectly affect Indiana bats and northern long-eared bats.**

- Hydrology would be altered by water from mined area being stored in sediment ponds thus minimizing the impact of periodically higher stream flows ("flashier") with increased levels of sediment that would have affected streams receiving runoff from the cleared mine site and affecting habitats in streams permanently or temporarily occupied by blackside dace.

- **These impacts could indirectly affect blackside dace.**

Exposure (time)

- Contaminated insect prey base from bats feeding over sediment ponds could affect Indiana and northern long-eared bats during the non-hibernating season when insects emerged from sediment ponds. This effect could gradually diminish with successful vegetation growth during reclamation, but some contamination could remain for an unknown time period.

- Hydrologic effects to streams in the affected watersheds would continue year-round until ponds are converted to wetlands during reclamation (10 to 20 years), and would continue at diminished levels following wetland conversion.

Exposure (space)

- Indiana bats and northern long-eared bats could be exposed to emerging insect prey potentially contaminated by metals or other toxicants at or around the constructed sediment ponds on mine site.
- Blackside dace would be affected in permanently and temporarily occupied streams receiving drainage from the mine site. Impacts would be reduced with increasing distance from the mine site.

Resource affected

- Insect prey for Indiana and northern long-eared bats.

- Hydrologic characteristics of receiving streams and stream benthic habitats where periphyton and aquatic insects used as food sources for blackside dace are found, and habitats used by blackside dace for spawning.

Individual response

- Reduced fitness for bats consuming prey that contain high levels of metals could affect their ability to survive and successfully reproduce. However, the availability of insect prey from other sources minimizes the potential for this potential impact. Also, successful reclamation and improved water quality and prey sources on the re-forested mine site could also result in increased bat fitness after contamination had declined and sediment ponds had been converted to wetlands as a result of reclamation.

- Reduced fitness for blackside dace could result from the need to relocate from areas where hydrology had been altered. However, properly constructed sediment ponds

would minimize adverse hydrologic and sediment impacts of the altered mining landscape to affected streams in the watershed that would be temporarily or permanently occupied by blackside dace.

Interpretation

- While sediment ponds may result in some contamination of insects used as prey by Indiana and northern long-eared bats, the ponds would be constructed to retain sediment and thus minimize the impacts of surface mining to hydrology, physical habitat, and water quality in streams receiving drainage. Aquatic insects contaminated by metals from mine runoff could result in bioaccumulation of metals for foraging bats using this resource, altered neurochemical responses, loss of homeostatic control. The potential for bioaccumulation that could affect bat fitness likely would depend on the foraging distances of these bats and resultant proportion of contaminated versus uncontaminated prey items contained in the diet of bats that may feed over sediment ponds. The potential for bioaccumulation of metals or other contaminants in bats consuming prey from these converted sediment ponds would diminish with increasing time since mining cessation and successful reclamation. The likely availability of other, uncontaminated prey suggests the potential for these adverse effects may be minimal.

4.5.4. Pathway 4

Activity – active mining operations

Active mining on an open surface mine consists of separating "stripping" the surface soil and rock, or overburden, from the top of the coal deposits. This is done by blasting, when needed, and using heavy equipment to remove and separate the coal from the overburden. A series of narrow, flat benches, ledges or steps are the result, when the surface spoil is removed, which allows access and removal of coal as it is encountered at different elevations. The mine site contains an estimated 5.4 miles of pre-SMCRA mining benches that have never been reclaimed (Tennessee Department of Environment and Conservation 2016a); the remining portion of this project would also include using these previously constructed benches to access remaining coal, when possible.

Stressor

- Blasting to separate coal from overburden that takes place during the summer roosting period for bats but could disturb bats roosting within a buffer of ½ mile around the permit boundary.

- **These impacts could indirectly affect Indiana bats and northern long-eared bats.**

- Minerals that are normally sequestered from surrounding aquatic ecosystems would be exposed to weathering when water infliltration of unconsolidated overburden would expose them to weathering, and minerals would become suspended or dissolved in water perculating through the strata to alter the chemical composition of groundwater and/or

surrounding surface waters receiving drainage from the mined area. Altered permeability and evapotransporation rate of exposed soils on the mine site would change the watershed's hydrologic characteristics, with more extremes in flow rates, volumes and periodicity. The increase in dissolved minerals would also change stream salinity (measured by increased conductivity), which could alter the assemblage composition of algaes and benthic macroinvertebrates. Additionally, suspended solids could precipitate along affected stream reaches, fouling spaces used by macroinvertebrates or clogging gills of macroinvertebrates or fishes. Altered hydrology could scour stream habitats during high flows or increase sediment deposition, and strand stream organisms during periods of reduced flows. Sediment deposition in streams could also result in decreased light penetration to the stream bottom, which could reduce the amount of aquatic vegetation (periphyton or phytoplankton), and reduce the amount and diversity of food available for aquatic insects and blackside dace.

- **These impacts could indirectly affect blackside dace.**

Exposure (time)

- Noise from blasting could affect bats potentially roosting within forested habitats in the ½ mile area surround the mine permit boundary during the non-hibernating period (April 1 – October 15), but the blasting would move across the mine site as mining progressed through the various "cuts" identified in the mine operations plan. The impact of this noise would be eliminated after mining had been completed (5-10 years after mining had commenced).
- Altered hydrology affecting blackside dace would gradually improve as vegetation cover is established during reclamation. Chemical composition of ground and surface water could remain altered for several decades. These alterations would be observed year-round, although elevated conductivity would be variable, depending upon precipitation and stream flow (because of dilution or concentration).

Exposure (space)

- Bats potentially roosting within forested habitats in the ½ mile surrounding the mine permit boundary during the non-hibernating period (April 1 – October 15) could be affected by noise related to mine operations.

- Blackside dace permanently or temporarily occupying streams receiving drainage from the mine site could be affected; those areas closest to the mine site would be more strongly affected than those further downstream in receiving watersheds.

Resource affected

- Forested habitats in the ½ mile area surround the mine permit boundary potentially used by roosting bats during the non-hibernating period (April 1 – October 15.

- Hydrologic characteristics, water quality, stream benthic habitats for periphyton and aquatic insects used as food sources for blackside dace, and all occupied and transitory habitats used permanently by blackside dace for feeding, breeding, sheltering, and traveling to other such areas.

Individual response

- Roosting bats disturbed by noise could flush from roosting habitats, requiring them to expend energy finding other appropriate roost trees.

- Increased sediment deposition may affect successful blackside dace reproduction, if eggs were smothered by silt or if nest-building fishes (chubs) were unable to find adequate nest rocks in silted stream bottoms to construct nest mounds that would be used by blackside dace spawning in their nests. Altered hydrology could result in periodic low flows that could affect successful blackside dace reproduction if eggs spawned by blackside dace were exposed to dessication because of low flows, or strand adult individuals. Blackside dace may be unable to find sufficient amounts of food because sediment deposition or water quality (increased salinity, measured as conductivity) had affected community composition of benthic food organisms such that adequate amounts of periphyton and/or aquatic insects appropriate for blackside dace diet were not available. Reduced blackside dace fitness may result in lowered survivorship from reproduction or surviving through stressful winter conditions. Exposure to altered habitat quality, inadequate food supply, and subsequent reduction in fitness may affect success of traveling blackside dace in attempts to invade or reinvade other appropriate areas.

Interpretation

- Bats that flushed to find other roosting habitat because of noise would expend energy that could reduce fitness.
- Altered habitat and water quality and/or quantity (e.g, periodic lower flows) in small tributary streams decreases blackside dace fitness because of sediment deposition, changes in hydrology, changes in water quality (e.g., increased conductivity because of increased salinity from suspended minerals), and changes in diversity and/or abundance of macroinvertebrate prey and periphyton. Altered habitat quality and/or quantity in small tributary streams used by blackside dace for spawning may reduce blackside dace population size because the amount of available spawning habitat and/or number of nest-building fish spawning associates (e.g., creek chubs) may be reduced. Also, opportunities for blackside dace expansion to other small streams in the watershed could be diminished, including reduction in availability of blackside dace refugia from inappropriate conditions elsewhere in the watershed. Altered habitat quality in affected tributary streams could also reduce opportunities for individuals to travel through the watershed to contribute to gene flow that would support the long-term persistence of the metapopulation throughout the affected watershed. Depending upon the number and location of other surface mines in the affected watersheds, these effects would likely be stronger and more pronounced in streams in closer proximity to the mine site, and gradually less pronounced in streams at greater distance because of dilution from other

unaffected tributaries. Also, following successful reclamation and forest regeneration the effects would gradually diminish with increasing time, possibly for several decades.

4.5.5. Pathway 5

Activity – reclamation

Disturbance of 472.5 of the 1,496 acre within the permit boundary would have resulted in removal of vegetation and coal. All of the 470.5 acres to be surface mined have been previously mined pre-SMCRA, and not reclaimed (Office of Surface Mining Reclamation and Enforcement 2013). Therefore, the mining and reclamation would result in removal of as many pre-existing highwalls as feasible (Office of Surface Mining Reclamation and Enforcement 2013) and reclamation of these areas would restore land contours across the permit boundary as close to the approximate original topography as possible. In addition, revegetation of the disturbed permit area would eventually result in restoration of forest to a minimum of 329 acres and conversion of the sediment ponds to wetlands as additional wildlife habitat in the post-mining footprint (Office of Surface Mining Reclamation and Enforcement 2013).

The mine plan's Indiana bat and northern long-eared bat PEP conservation measures identified appropriate tree species to be used for reclamation, and required 400 surviving stems per acres before restoration would be considered successful. In addition, the planting layout was intended to maximize use by roosting Indiana and northern long-eared bats (Office of Surface Mining Reclamation and Enforcement 2016). However, before trees would be planted herbaceous ground cover would be planted in order to control sediment from the mined areas. In areas where there were no soils or rocks that could contribute toxic components potentially affect ground or surface water quality, the FRA restoration method would be used (Office of Surface Mining Reclamation and Enforcement 2013, U. S, Department of Interior 2015). All 470.5 acres that would be replanted in trees would be reclaimed using FRA.

When they would no longer be needed for sediment retention, ponds would be modified to shallow water depressions, or wetlands by filling the pond such that the depression retained a depth of one to two ft, blending the topography with the surrounding area, and constructing a channel through the pond to ensure that large volumes of water were not retained in the shallow depression remaining (Office of Surface Mining Reclamation and Enforcement 2013).

Stressor

- Water infiltration of unconsolidated overburden in the areas reclaimed with FRA would expose minerals that were normally sequestered from surrounding aquatic ecosystems to weathering, and minerals would become suspended or dissolved in water perculating through the strata to alter the chemical composition of groundwater and/or surrounding surface waters receiving drainage from the mined area. Until vegetation had been established, altered permeability and evapotransporation rate of exposed soils on the mine site would continue to change the watershed's hydrologic characteristics, with more extremes in flow rates, volumes and periodicity. The increase in dissolved minerals would also change stream salinity (measured by increased conductivity), which could

result in altered benthic macroinvertebrate assemblage composition. Until vegetation had become well established, altered hydrology could continue to scour stream habitats during high flows or increase sediment deposition, and strand stream organisms during periods of reduced flows. The sediment deposition could result in decreased light penetration to the stream bottom in affected areas and reduce the amount of aquatic vegetation (periphyton or phytoplankton) present, affecting the amount of and diversity of food available for aquatic insects and blackside dace. As a result of successful vegetation establishment, previous adverse habitat impacts from altered hydrology and sediment deposition may improve.

- **The continued adverse impact of altered water and habitat quantity and quality could indirectly affect blackside dace.**

- Indiana and northern long-eared bats foraging in the wetland areas established as a result of sediment pond conversion may be affected by contaminated aquatic insect prey exposed to mine drainage.

- As tree species with characteristics favorable to roosting Indiana and northern long-eared bats are established across the mine site, the amount of available bat roosting habitat would gradually increase.

- **The beneficial effects of increasing availability of non-hibernation (roosting) habitat for Indiana bats and northern long-eared bats would be indirect, as would the beneficial impacts to blackside dace resulting from improved water and habitat quality related to successful reclamation that reduces stream impacts from permit boundary runoff and discharges.**

Exposure (time)

- Vegetation establishment and ecological succession that would eventually result in forests containing species composition and trees of suitable size for use by roosting Indiana bat and northern long-eared bat is conservatively expected to take a minimum of 60 and 40 years, respectively.

- Vegetation establishment and ecological succession described above would result in gradual improvement to the hydrologic changes, sediment deposition, and resultant changes in benthic community assemblages that may have adversely affected blackside dace. Some improvement (i.e., sediment and habitat impacts) may be apparent soon after cessation of mining and establishment of vegetation on the mine site. However, while the chemical composition of water quality would also gradually improve with ecological succession over the areas where vegetation hads become established, elevated conductivity may not be diminished for many decades (or up to a century or more). Therefore, some observed changes to benthic community assemblages may not improve in streams used temporarily or permanently by blackside dace.

Exposure (space)

- All stream reaches in affected watersheds that would be temporarily or permanently occupied by blackside dace would be affected by habitat improvements (sediment deposition and alterations in flow from hydrologic impacts) related to reclamation; habitat improvements would be most noticeable in stream reaches closest to mine permit boundary.

- As forest regeneration were successful, Indiana bat and northern long-eared bats roosting in adjacent areas would benefit across the entire mine site.

Resource affected

- Water quality, stream benthic habitats for periphyton and aquatic insects used as food sources for blackside dace, and all occupied and transitory habitats used permanently by blackside dace for feeding, breeding, sheltering, and traveling to other such areas.

- Forest habitat used for roosting by Indiana bats and northern long-eared bats, and wetland areas (converted ponds) used by bats for water sources and for foraging.

Individual response

- Reduction in sediment deposition may increase proportion of successful blackside dace reproduction and improve community composition of benthic food organisms, including periphyton and/or aquatic insects appropriate for blackside dace diet; increased fitness may result in increased survivorship from reproduction or surviving through stressful winter conditions.

- Indiana bats and northern long-eared bats may be more successful in finding suitable roost sites, when forests have regenerated and trees have matured to provide suitable habitat for these bats. These bats may also benefit from increased water sources and sources of prey (the converted sediment ponds).

Interpretation

- Improved habitat quality, improved food supply, and subsequent improved individual fitness may improve the success of traveling blackside dace attempting to invade or reinvade other appropriate areas in the watershed, including areas that may have previously been adversely affected by mine activities. This could allow for gene flow that would support the long-term persistence of the blackside dace metapopulation throughout the affected watershed.

- Forest regeneration with appropriate tree species would eventually provide habitats suitable for Indiana bat and northern long-eared bat roosts that could increase opportunities for growth and successful reproduction or offset loss of roosting forest habitat from other, unrelated activities in the area. However, as identified above, ecological succession that would provide appropriate habitats for use by roosting Indiana bat and northern long-eared bat was conservatively expected to take a minimum of 60 and 40 years, respectively. These bats may also benefit from increased water sources and sources of prey (the converted sediment ponds).

4.6. Summary of effects

4.6.1. Beneficial effects

Improved habitat and water quality, macroinvertebrate and periphyton food supply that would eventually result from restoring previously disturbed stream reaches and from successful reclamation and ecological succession of previously mined and unreclaimed portions of the permit area could eventually benefit blackside dace attempting to invade or reinvade other appropriate areas in the watershed, including areas that may have previously been adversely affected by mine activities because these traveling blackside dace would likely be more physically fit and able to successfully produce offspring. This could allow for gene flow that would support the long-term persistence of the blackside dace metapopulation throughout the affected watershed.

Successful reclamation that resulted in regenerated forested habitats that include tree species appropriate for roosting bats, would eventually benefit Indiana bat and northern long-eared bats using the region. This successful reclamation and ecological succession would also eventually benefit blackside dace as stream habitats affected by sediment deposition from the pre-SMCRA and proposed mining were improved by reclamation. Eventual water quality improvement may require many decades, but successful reclamation would also likely benefit water quality as well.

4.6.2. Direct adverse effects

4.6.2.1. Indiana bat and northern long-eared bat

Bat surveys in the vicinity of the permit boundary have not indicated that Indiana bats are known to use habitats in the vicinity, including within the mine permit boundary. Summer roosting northern long-eared bats were identified in these surveys, however, and this species could use habitats on the mine site for summer roosting. Therefore, as indicated above, for the first two years of the permit's authorization tree removal to facilitate the mining could occur during any time of the year except June or July, when non-volant bats may be present on the mine site. Afterward (from March 31, 2019), any additional tree removal needed to facilitate the mining would be accomplished according the the Indiana and northern long-eared bat PEP. In addition, the amount of forest removed to facilitate the surface mining and sediment pond construction is a relatively small proportion (427.5 acres) of the total permit boundary (Figure 10).

Direct effects that could result from the two-year authorization to remove trees could cause any Indiana bats or northern long-eared bats that might be using trees on the permit boundary for roosting to move to other suitable areas if trees they were using were removed to facilitate sediment pond construction or for active mining operations. Depending on the distance the bats were required to fly to find suitable roost tree habitats, their energy expenditure could result in reduced fitness, and possibly result in lowered survivorship in offspring for female bats. .

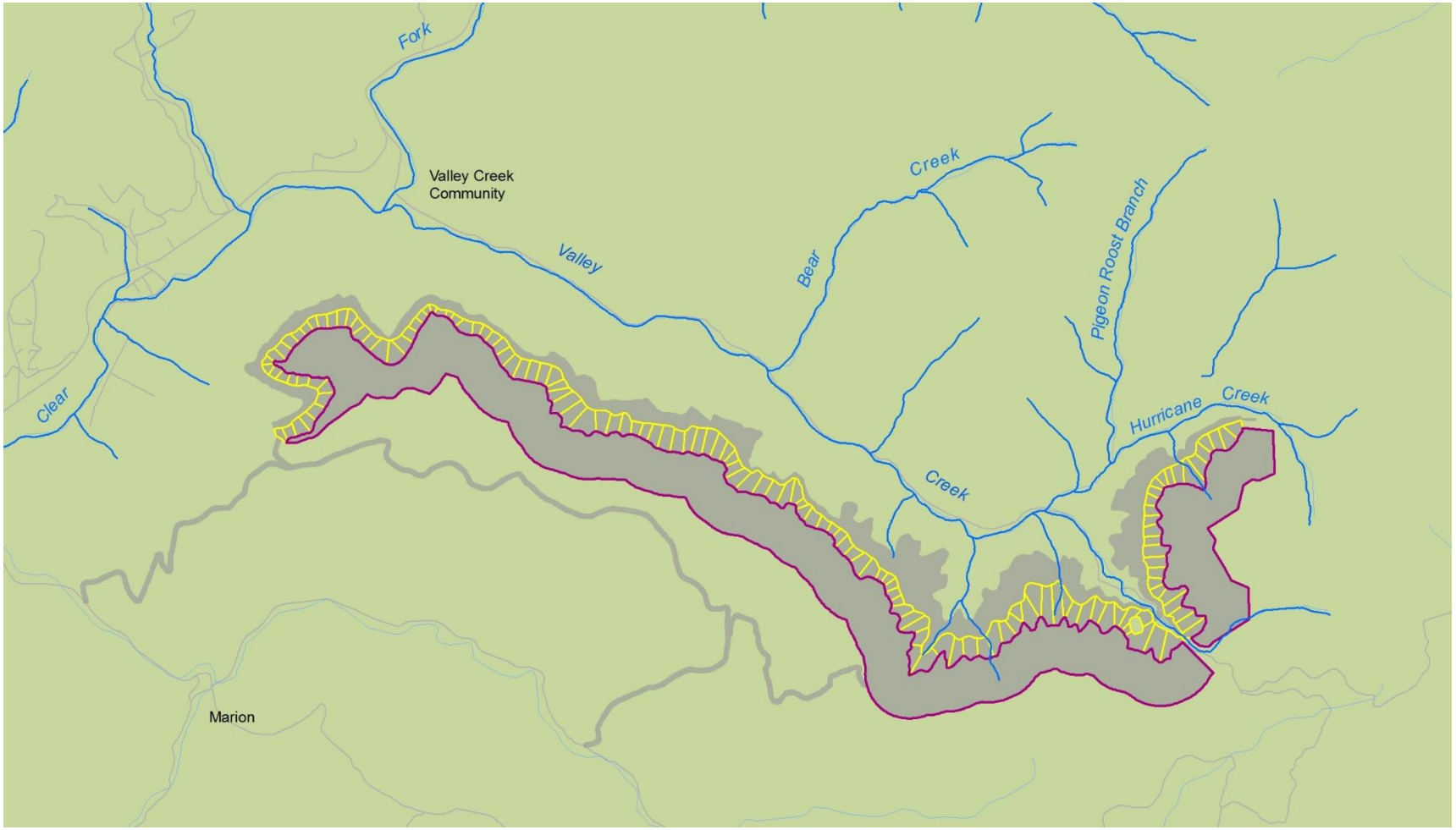

Figure 10. Kopper Glo Mining, LLC Cooper Ridge Surface Mine permit boundary (OSMRE permit 3270), Claiborne County, Tennessee. The entire 1,496-acre permit boundary is shown in gray, and the "cut" sequences (illustrated by yellow polygons) show the 472-acre portion within this boundary that will be surface mined.

4.6.2.2. Blackside dace

There were no identified direct impacts to blackside dace from the proposed surface mining project

4.6.3. Indirect adverse effects

4.6.3.1. Indiana bat and northern long-eared bat

Habitat loss from vegetation clearing for the proposed mining would affect Indiana bat and northern long-eared bat potentially using the mine permit boundary during non-hibernation periods until reclamation had resulted in re-growth of mature forests and appropriate summer roosting habitat, which could be 75-100 years. Indiana and northern long-eared bats that would have searched for appropriate roosting habitat on the mine site would expend energy finding alternative roosting sites elsewhere. In addition, Indiana and northern long-eared bats roosting in forested habitats adjacent to the mine site could be affected by noise from vehicle and other equipment use during vegetation clearing, road construction and maintenance, active mining operations, and transportation of materials, personnel, and coal to and from the mine site. These bats would expend energy finding alternative roosting sites further from the noise disturbance. Sediment impacts that reduced abundance of aquatic insects along the stream reaches affected by increased vehicle traffic for coal transport from the mine site could also affect Indiana bats or northern long-eared bats that might be foraging along this stream corridor. These individuals may expend energy to search for appropriate prey elsewhere. Indiana or northern long-eared bats could be affected by altered neurochemical responses and loss of homeostatic control because of bioaccumulation of metals from foraging for aquatic insects around sediment ponds or converted wetlands.

4.6.3.2. Blackside dace

Blackside dace temporarily or permanently inhabiting streams in the affected watersheds would be indirectly affected by the vegetation clearing, road construction and maintenance (including stream crossings), sediment pond construction, active mining operations (including transportation of materials, personnel, and coal to and from the mine site), and reclamation activities. Altered hydrology that could affect water quantity, rate, and periodicity of stream flows, altered habitat and water quality that could affect blackside dace by affecting the abundance and diversity of their prey, fitness, reproduction, and movement of individuals that could impact the species' ability to expand throughout affected watersheds, and therefore the species' metapopulation persistence.

4.7. Interrelated and interdependent actions

Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration. We have not identified any interrelated or interdependent effects for this consultation.

4.8. Numbers of individuals/populations affected in the action area

4.8.1. Indiana bat and northern long-eared bat

The mine project boundary is not within a five-mile radius of known hibernacula for Indiana bat or northern long-eared bat, nor is there any known winter habitat for these species present within the permit boundary; the permit boundary contains only potential summer roosting bat habitat for northern long-eared bat. Five surveys (2010 – 2013, Table 3) conducted on and in the general area of this proposed mine site failed to identify any Indiana bats use, but two of those surveys did identify the northern long-eared bat as being present in this area (Table 3).

4.8.2. Blackside dace

In spite of numerous stream assessments and surveys, in the action area permanent blackside dace populations are only known to occur in two streams in the action area, Rose Creek and Buffalo Creek (Table 5). These streams are located in the Clear Fork Recovery Unit (U. S. Fish and Wildlife Service 2015d). As noted in Table 5, the Buffalo Creek population is considered "vulnerable", and the Rose Creek population is considered to be "stable" (U. S. Fish and Wildlife Service 2015d).

In addition to information presented in Table 5, recently collected water quality data (Table 6) indicate that existing (baseline) condititions (conductivity levels less than 343 µS/cm as reported by Hitt et al. 2016) within the action area, are not consistently appropriate for blackside dace in many of the streams within the action area. Therefore, impacts of the mining to persistent blackside dace occurrences in stream reaches receiving discharge or runoff from the mine permit boundary are unlikely. However, because elevated conductivity resulting from the mining may persist for decades or up to a century, the recently completed BO for the Middlesboro Mining Company, Inc. Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296, see U. S. Fish and Wildlife Service 2016a) considered that a few individual blackside dace traveling through affected stream reaches that also are within the action area for this current Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270) in attempts to colonize new areas in the watershed may be indirectly affected by hydrologic, and habitat and water quality effects of the mining. Consequently incidental take was authorized for those blackside dace individuals that may be affected by hydrology, habitat quality, or water quality as they travel through or temporarily occupy affected stream reaches for the length of time that OSMRE maintains regulatory authority (e.g., through Phase III Bond Release) for Sterling and Strays Surface Mine Number 1. This Kopper Glo Mining, LLC Cooper Ridge Surface Mine will affect some of the same streams as those same stream reaches where blackside dace incidental take was previously authorized for the Sterling and Strays Surface Mine Number 1. As the hydrologic analyses (Table 6) for water quality (conductivity) changes to these stream reaches from the Cooper Ridge Surface Mine activities do not significantly change predictions for water quality impacts result from the Sterling and Strays Surface Mine Number 1, this BO considers additional blackside dace authorization for Cooper Ridge Surface Mine for these stream reaches is not needed, and that any affected blackside dace individuals in those stream reaches would be included in the prior authorization. If the length of time for which OSMRE retains regulatory authority for the

Middlesboro Mining Company, Inc. Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296) is not adequate to include the time period during which OSMRE would retain regulatory authority for the Kopper Glo Mining, Inc. Cooper Ridge Surface Mine (OSMRE permit 3270) the previously authorized blackside dace incidental take would continue to be authorized for permit 3270 until such time as OSMRE no longer retains regulatory authority (e.g., through Phase II Bond Release) for the Cooper Ridge Surface Mine permit.

5. 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 biological opinion. Factors discussed in section 3.0 (*Environmental baseline*) are expected to continue to affect the listed species considered in this BO into the future. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation under section 7 of the Act.

6. Conclusion

The Service defines "to jeopardize the continued existence of a listed species" as 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 a listed species in the wild by reducing the reproduction, numbers, or distribution of the species.

After reviewing the current status of the blackside dace and Indiana bat, the environmental baseline for these species within the action area, the effects of the proposed surface mining and reclamation and the cumulative effects, it is the Service's biological opinion that the surface mining and reclamation, as proposed, is not likely to jeopardize the continued existence of the blackside dace or Indiana bat. It is further determined that the proposed surface mining and reclamation will not destroy or adversely modify designated critical habitat, because no critical habitat has been designated for blackside dace. Nor will the proposed surface mining and reclamation destroy or adversely modify designated critical habitat for Indiana bat because no such habitat is located within the action area of this biological opinion.

The blackside dace is considered stable throughout its range. The action area of this Biological Opinion is relatively small and contains only a small fraction of the blackside dace overall population. In addition, with the exception of the "stable" Rose Creek population that is not affected by the proposed surface mining and reclamation, all known blackside dace populations in the action area are considered "vulnerable" (U. S. Fish and Wildlife Service 2015d), and are estimated to be small in extent of range and in population size. None of the known extant populations in the action area would be directly affected by the proposed surface mining, although a few individuals traveling through affected stream reaches to attempt to colonize other areas throughout the watershed could be affected by lingering water quality impacts from mine runoff over the long time period that water quality could remain affected by the mining. Therefore, while the individuals that could be affected by the action may be attempting to expand the species' range within the affected watersheds, they are not necessary for survival and recovery of the species. In addition, the baseline water quality in these stream reaches has

already been affected by previous mining and other land uses, including surface coal mining that took place prior to enactment of SMCRA, which also lowers the likelihood of successful blackside dace colonization of other portions of the affected watershed. Some of the areas where pre-SMCRA mining took place have not been reclaimed, and some of the areas mined and permitted according to SMCRA regulations did not include considerations of the relatively recent science regarding blackside dace water quality (i.e., specific conductance) thresholds or metapopulation dynamics.
As a result of the history in the action area, the Service considers that no permanent blackside dace populations are known to persist in the action area that would be affected by the proposed surface mining and reclamation. However, the temporal (specific conductance) impacts of the proposed mining associated with specific conductance are apparently long lasting. During the period in which stream conductivity would likely remain elevated as a result of this surface mining action, a few blackside dace individuals could occasionally, or periodically move through streams within the action area in search of other, more appropriate areas (Table 7). Further, the species could attempt to colonize or recolonize presently unoccupied streams within the action area. These individuals could be affected by the proposed actions.

After considering the status of the Indiana bat and blackside dace within the action area and throughout their ranges, the environmental baselines within the action area, and all of the effects of the proposed action (both adverse and positive), the Service believes that the species' reproduction, numbers and distribution will not be appreciably reduced as a result of the proposed action. Therefore, Indiana bat and blackside dace can be expected to survive and potentially be recovered within the action area and the rest of their ranges.

No DCH for the Indiana bat occurs within the project action area; therefore, none would be destroyed or adversely modified by the agency action.

7. Incidental take statement

Section 9 of the Act and the implementing regulations prohibit the take of endangered and threatened species, respectively, without 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 Service to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Harass is defined by the Service as intentional or negligent actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering.

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 Act, provided that such taking is in compliance with the T&Cs of this Incidental Take Statement.

The reasonable and prudent measures (RPMs) and their implementing terms and conditions (T&Cs), described below are non-discretionary, and must be undertaken by OSMRE, so that they become binding conditions of any grants, permits or contracts, as appropriate, for the exemption in section 7(o)(2) to apply. OSMRE has a continuing duty to regulate the activities covered by this Incidental Take Statement. If OSMRE: (1) fails to assume and implement the T&Cs or (2) fails to adhere to the T&Cs of the Incidental Take Statement through enforceable terms that are added to the grant, permit or contract, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the Service must monitor the action and its impact on the species as specified in this Incidental Take Statement (50 CFR § 402.14 [1][3]), see Section 9.1.5.

7.1. Amount or extent of take anticipated

7.1.1. Indiana bat

No Indiana bat hibernacula are known to occur on the mine project boundary or within a five-mile radius, and no Indiana bats are known to use the project boundary as summer roosting habitat (Table 3). However, as there have been no surveys of the adjacent areas, OSMRE prudently assumed potential presence of summer roosting Indiana bats in the action area. For the first two years' activities authorized by the permit, tree removal to accommodate the mining will be allowed during any month except when young bats could be present in maternity roosts, and would be non-volant (June and July). After these first two years, tree harvest will be restricted to the timelines included in the Indiana bat PEP.

Based on the prudent assumption that there could be habitat appropriate for Indiana bats to use as summer roosts, including maternity roosts, anywhere on the 472-acre portion where surface disturbance for mining would occur on the permit boundary and that trees where maternity roosts might be established during the first two years of this permit's activity, the Service used information regarding the foraging range of female Indiana bats in a maternity colony to include the habitat within a 4-km (2.5 mile) radius of a maternity roost tree as the home range (including roosting and foraging areas) of the colony (U. S. Fish and Wildlife Service 2011). The Service considers the roosting area within the home range to comprise 397 acres, or a circular area 1.4 km (0.87 miles) in diameter (U. S. Fish and Wildlife Service 2015h). Therefore, considering that the roosting areas (0.87 miles) of Indiana bat maternity colonies are not thought to overlap, the 472-acre area where timber harvest would take place during the non-hibernating season (except for June or July) to facilitiate surface mining could potentially contain one Indiana bat maternity colony.

In addition, when considering the potential impacts of noise and other disturbance from within the permit boundary to roosting bats, we included the entire 1,496 acres of the permit boundary and the adjacent ½ mile buffer around the permit boundary, the 5,980-acre area included in the ½ mile buffer adjacent to and around the mine permit boundary (for a total of 7,476-acre terrestrial action area) around the permit boundary could potentially contain approximately 19 different Indiana bat maternity colonies, if occupancy of that area were 100 percent. However, this would likely overestimate the actual number of maternity colonies. For example, the estimated post-WNS occupancy rate for Indiana bats in Kentucky is 1.4 percent (U. S. Fish and Wildlife Service

2015h). While there is no post-WNS occupancy rate calculated for Tennessee, the Service assumes that, based on proximity, it would be similar to Kentucky's occupancy rate for this project (1.4% of 19 < 1 maternity colony). However, based on the WNS occupancy rate, it is reasonable and conservative to assume that one maternity colony occurs where there is potential habitat on the permit boundary and within the ½ mile buffer adjacent to the permit boundary.

Therefore, although there are no known Indiana bat maternity colonies near the permit boundary or the adjacent buffer, and in spite of valid negative Indiana bat surveys confirming summer roosting bat use within the permit boundary (see status of the Indiana bat in the action area, section 3.6.2 in this BO), the buffer area has not been surveyed. As OSMRE has prudently assumed presence of the species in this adjacent buffer area, the Service also conservatively assumes that there could be at least one undocumented maternity colony within this portion of the action area. Based on our knowledge of the biology of the species and an assumed sex ratio of 1:1, each colony comprises 60 adult females, 60 pups, and 60 adult males (U. S. Fish and Wildlife Service 2007). Therefore, approximately 180 Indiana bats could be using the permit area and adjacent buffer around the permit boundary and be affected by the proposed action.

The Service anticipates incidental take of the Indiana bat would be difficult to detect for the following reasons:

1. the individuals are small and when occupying summer habitats are difficult to find (they have not been captured in the action area during past surveys);

2. Indiana bats form small (approximately 10-100 individuals), widely dispersed maternity colonies under loose bark or in the cavities of trees, and males and non-reproductive females may roost individually, which would render finding the species or occupied habitats difficult;

3. finding dead or injured specimens during or following project implementation would be unlikely due to their small size and the vast area contained within the action area;

4. the Indiana bat PEP would minimize adverse effects to the species and minimize the level of incidental take;

5. and incidental take would be non-lethal and undetectable;

However, incidental take of Indiana bats can be expected as a result of the action. Service believes that the Indiana bat could be adversely affected as a result of noise disturbances (generated by mining operation and transportation activities), reduced roosting prospects and less favorable foraging opportunities. The Service has determined that up to 180 Indiana bats potentially occupying suitable summer habitat within the 7,476-acre area that includes the 1,496 acres within the permit boundary and an adjacent (1/2 mi) buffer area of 5,980 acres around the mine permit boundary (Table 8) would be incidentally taken in the form of harm or harass.
In the *Analyses of effects* (section 4.4) in this BO, the Service determined that the action resulted in incidental take of the Indiana bat covered in this opinion in several forms including:

2) harm from: 1) tree removal causing females on the mine permit boundary to abandon (unknonwn) roost trees that would be removed to facilitiate the mining, and relocate to areas with potentially less suitable habitat conditions, resulting in decreased reproductive success; 2) noise disturbances causing females to abandon (unknonwn) roost trees and relocate to areas with potentially less suitable habitat conditions, resulting in decreased reproductive success; 3) noise disturbances causing pregnant females to abandon roosts, and reabsorb embryos or trigger spontaneous abortion due to associated stress; 4) noise disturbances causing individuals to abandon roost sites and migrate outside of the action area to swarming areas earlier than usual, when they would not be as fit as they should be for migrating long distances; 5) reduced roosting prospects and less favorable foraging opportunities due to loss of summer habitat from mine development and operations forcing individuals to relocate to other areas with potentially less suitable habitat; 6) noise disturbances causing females to drop non-volant pups in their haste to escape and carry them to safety; 7) reduced fitness from individuals forced to look for prey items along streams where sediment impacts had reduced availability; and 8) reduced fitness because of bioaccumulation from individuals feeding on insect prey contaminated by metals while foraging around sediment ponds;

(a) harassment from: 1) noise disturbances causing individuals to switch roost sites more frequently (disrupting foraging, roosting and/or rearing of young); and 2) noise disturbances causing individuals to relocate to other areas with potentially less suitable habitat.

Table 7. Estimated Indiana bat incidental take resulting from mining and reclamation activities at the Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270).

Minimum Age of Forest Providing Appropriate Habitat[a]	Terrestrial Buffer Acreage For Temporary Affects[b]	Forest Acreage Affected by Project[c]	Number of Affected Individuals
60 years	5,980 acres	1,496 acres	180

[a]For mined areas where there has been no previous mining or other major landscape disturbance, the minimum age for growth of forests with appropriate characteristics would be less than areas previously disturbed.
[b]Acreage of buffer was calculated by Geographic Information System (GIS) analysis, and includes a half-mile around mine permit boundary.
[c]The amount shown in this cell is the total permit acreage, but surface mining will only affect a subset of the permit area (472 acres), with a total of 752 acres of disturbance for various mine-related activities (roads, deep mine face-up, sediment basins, etc.).

7.1.2. Blackside dace

Mattingly et al. (2005) and Black and Mattingly (2007) identified 12 streams as having the most robust populations of the species. Population estimates for all of these streams exceeded 100 individuals/200-m reach and ranged from a low of 104 (Fall Branch, Tennessee) to a high of 613 (Breedens Creek, Kentucky) individuals. Blackside dace densities reported by Leftwich et al. (1995) for a stream partially located on the DBNF (Middle Fork Beaver Creek) are also similar to those included above. The dace were widely distributed in the stream, and the Leftwich et al. surveys included many habitat units (including pools and riffles), with densities ranging from near 0 to around 1.3 individuals per square meter.

Population estimates for individual habitat units surveyed ranged from 0 to 16 individual blackside dace, and as would be expected for a headwater stream inhabitant, Leftwich et al. reported that densities increased from downstream to upstream reaches. Population densities in the dataset used by Hitt et al. (2016) were 1-153 individuals (mean 11.7 and 26.0 standard deviation).

The Tennessee Wildlife Resources Agency (2011, 2012, 2013, 2014) estimated blackside dace populations in 200 m reaches of Straight Fork (of the Big South Fork drainage), Jake Branch, Hudson Branch, Terry Creek, and Louse Creek between 6 and 142 individuals (in Hudson Branch and Terry Creek, based on 2012 and 2011 electroshocking data, respectively), based on capture of 2-43 individuals, respectively, in their electroshocking samples. Their methodology for calculating population estimates was based on one-pass electroshocking surveys as identified by Black and Mattingly (2007). However, TWRA was advised by Mattingly that the low number of blackside dace collected in some of these Tennessee streams were lower than the model was calibrated, which entered error into the population estimates. However, the error introduced likely overestimated the true population size (Carter personal communication 2016). Conductivities at these survey sites ranged from 80 µS/cm (Hudson Branch 2013) to a high of 394 (Straight Fork of the Big South Fork drainage in 2014).

In spite of the data presented above for blackside dace populations in some stream reaches, most blackside dace populations are considered to be small and remnant in nature (i.e., less than ten individuals observed during surveys), and an adequate understanding of population viability is lacking (Black et al. 2013b). Survey efforts specifically aimed at blackside dace are highly variable throughout the range of the species (U. S. Fish and Wildlife Service 2015d), which also complicates our understanding of population sizes and viability.

Also, depending upon possible blackside dace movement and timing of surveys, individuals may, or may not, be observed. For example, in spite of several previous surveys (Shoup and Peyton 1940, Comiskey and Etnier 1972, Brazinski 1979, O'Bara et al. 1982, Kirsch 1983), the blackside dace was unknown from the Big South Fork National River and Recreation Area (BISO) until Scott (2010) reported results from a 2003-2006 inventory of BISO fishes. Scott completed 68 surveys of 41 tributaries (some tributaries were surveyed at more than one site, and some sites were surveyed more than once). These surveys identified blackside dace at two localities in a single headwater stream in the system, representing a new distributional record for the species. The extent of the size and range of this relatively newly-discovered population in the

BISO remains unknown, but Scott (2010) considered this species rare, comprising only 0.24% of the total individuals of all fish species he collected at the 82 total sites surveyed in 2003-2006.

Based on the information presented above, the Service believes there is no reasonable way to determine how many blackside dace actually exist in streams within the action area during mining, reclamation, how many could occasionally (temporarily) occupy these streams or stream reaches, or how many could attempt to (re)colonize these areas during the time that water quality will continue to be affected by the permitted actions after the project has been completed. As discussed in the *Literature review – reclamation, longevity of effects from altered ground and surface water quality* section (4.3), it can take decades for the water quality impacts (elevated conductivity) resulting from surface mining to completely abate. Evans et al. (2014) reported that water quality impacts from surface coal mining are likely to be long-term, but not permanent. They reported a wide range of time required for salinities (measured as conductivity) to decline below 500 µS/cm in affected streams, with mean of 19.6 (+ 6.6) years. As the best available science at this time indicates that blackside dace persistence is unlikely at conductivities above 343 µS/cm, the Service conservatively assumes the recovery period to be at least double this time or, more than five decades, for water quality impacts to improve in streams receiving drainage from surface coal mines.

7.1.2.1. Action area stream reaches covered by previously authorized blackside dace incidental take

An ESA consultation with OSMRE (U. S. Fish and Wildlife Service 2016) analyzed the potential for surface mining activities by Middlesboro Mining Operations, Inc. at their Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296) to adversely affect blackside dace. In that Sterling and Strays Surface Mine Number 1 BO (U. S. Fish and Wildlife Service 2016) the Service determined the proposed action could result in the incidental take of up to 38 blackside dace individuals per year potentially occupying 52 miles of streams in the watersheds where water and habitat quality would be affected by mining, with this number of individuals diminishing over time with successful reclamation. The incidental take included in this BO is authorized for OSMRE until such time as OSMRE no longer maintains regulatory authority (e.g., through Phase III Bond Release). A total of 18.4 stream miles for stream reaches of Valley Creek, Hurricane Creek, and Clear Fork included in this prior incidental take authorization are also within the action area for this Cooper Ridge BO (Table 9 and Figure 11).

The OSMRE analysis of potential water quality impacts for the Sterling and Strays Surface Mine Number 1 mentioned above included projections of inputs for all existing, proposed (mine projects for which permit applications had already been received by OSMRE), and anticipated (mine projects for which OSMRE had information indicating permit applications would soon be received). This analysis included the Kopper Glo Mining, LLC Cooper Ridge Surface Mine, and also an anticipated “Hignite” mine for which OSMRE had received an application. As a result, the analysis could be considered the “worst case” scenario. Since that BA was finalized, however, the “Hignite” application has been withdrawn. Therefore, the Service is confident that the conditions used to designate stream reaches appropriate for blackside dace incidental take authorization in the Sterling and Strays BO remains applicable to the Cooper Ridge analysis, as included in this BO.

In addition, while the analyses included in the Sterling and Strays BO included decades of data from OSMRE's water quality surveys at "fixed" trend stations, those data are from quarterly water samples. As established to support the Sterling and Strays consultation, continuously recording data loggers have been deployed at three locations within the action area of this BO (upper Valley Creek, Valley Creek near its confluence with Clear Fork, and Clear Fork near the confluence with Straight Creek) from May 2015 to present. These data include many thousands of individual data points (Office of Surface Mining Reclamation and Enforcement unpublished data) that demonstrate considerable variation in conductivity and specific conductance measurements depending on seasonal and daily water temperatures, and influenced by rainfall and corresponding stream flow. Appendix C in this BO was provided to the Service by OSMRE hydrologists, and includes summary statistics and graphic representations of this variability. These data also provide support for the conclusion in this BO that **blackside dace incidental take previously authorized in the Sterling and Strays BO for stream reaches included in the action area for the Cooper Ridge Surface Mine is adequate for blackside dace impacts of the Kopper Glo Mining, LLC Cooper Ridge (OSMRE permit 3270).** However, several stream reaches in the Cooper Ridge action area were not included in this previous incidental take authorization. These are discussed below.

7.1.2.2. Action area stream reaches not covered by previously authorized lackside dace incidental take

Several other Valley Creek and Clear Fork tributaries in the action area of this BO were not included in this Sterling and Strays Surface Mine Number 1 blackside dace incidental take authorization. These streams include Nolan Branch, Spar Branch, and five unnamed Valley Creek headwater tributaries (Table 7 and Figure 11). None of these stream reaches are currently occupied by blackside dace, nor are blackside dace known to have previously occurred in these stream reaches (Figure 9). However, these streams are all small, perennial, first-order streams that could potentially provide appropriate habitat for blackside dace.

As indicated by the OSMRE CHIA for t LLC Cooper Ridge Surface Mine (Office of Surface Mining Reclamation and Enforcement 2016b), water quality trends in the Valley Creek and Clear Fork watersheds are gradually improving, presumably due to mining actions that reclaim previously unreclaimed surface mined areas (remining), as proposed for the Cooper Ridge Surface Mine actions considered in this BO. As water and habitat quality conditions continue to gradually improve as expected, Clear Fork and Valley Creek could become more conducive as "travel corridors" to allow for blackside dace migration throughout the watershed. As discussed in the sections 2.3.2 and 4.1.9 (*Life history*, and *Literature review – effects of altered ground and surface water chemistry to fish dispersal and genetic variation of metapopulations*, respectively), a small propotion of blackside dace individuals might move within a watershed to occupy other (previously uninhabited) streams.

The distances between the nearest Clear Fork system blackside dace population (Buffalo Creek, see Figures 2 and 9) and Spar Branch is within the 2.5 mile maximum distance for individual blackside dace movement documented by Mattingly et al. (2005) and Detar and Mattingly (2013). There are 2.4 stream miles of Clear Fork between the confluences of Buffalo Creek and

Spar Branch. The confluence of Nolan Branch is only a half-mile (2.9 miles total) further from Buffalo Creek, and Spar Branch and Nolan Branch are 3.7 and 4.2 stream miles from the confluence of Rose Creek.

In addition, while the five unnamed Valley Creek tributaries (Figure 2) are located further than this 2.5-mile distance (the shortest and longest distances between these tributaries and Buffalo or Rose Creeks are 7.2 and 9.9 stream miles, respectively) if longer term dispersal of blackside dace through stream systems might be accomplished by the "stepping stone" model of dispersal (Katz et al. 2014, and see *Literature review – effects of altered ground and surface water chemistry to fish dispersal and genetic variation of metapopulations* (section 4.1.9 in this BO) these tributary streams might eventually be colonized.

Therefore, in order to conservatively estimate the potential impact of water and habitat quality to blackside dace, the Service authorizes incidental take for blackside dace individuals originating in Buffalo Creek or Rose Creek (Figures 2 and 9) that might migrate to Nolan Branch, Spar Branch, or the five unnamed Valley Creek headwater tributaries within the time period that OSMRE would retain regulatory authority for this Cooper Ridge Surface Mine. Because of the uncertainty in accurately estimating the number of individual blackside dace occupying the stream reaches affected by the surface mining, as discussed above, the existing baseline conditions in affected streams that are believed to currently be inappropriate for persistent blackside dace populations (Table 6), and the absence of known blackside dace populations occupying stream reaches affected by this proposed mining, the Service believes that the amount of incidental take is best estimated by the total amount of blackside dace habitat where elevated conductivity could affect individual blackside dace traveling throughout the watershed over the time period that water quality impacts of the mining may persist. As part of this take, we expect all blackside dace potentially traveling through affected stream reaches would be harmed, harassed, and/or killed as a result of the proposed action. We have attempted to conservatively estimate a number of individuals that could be taken over the length of time that water quality in the form of elevated conductivity, could persist as a result of this surface mining.

Table 8. Stream reaches in the action area of the Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270) where blackside dace incidental take resulting from surface mining and reclamation activities at Middlesboro Mining Operations, Inc. Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296) has been previously authorized and additional stream reaches where blackside dace incidental take is authorized for the Cooper Ridge Surface Mine.

Stream reaches with previously authorized blackside dace incidental take for surface mining impacts from Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296)		
Stream	**Status[a]**	**Stream Length (miles) in Action Area**
Valley Creek	Vulnerable	5.0
Hurricane Creek	Unoccupied	1.7
Clear Fork	Vulnerable	11.7
Total		**18.4**
Stream reaches included in blackside dace incidental take authorization for surface mining impacts from Cooper Ridge surface mine (OSMRE permit 3270)		
Stream	**Status[a]**	**Stream Length (miles) in Action Area**
Nolan Branch	Unoccupied	0.6
Spar Branch	Unoccupied	0.3
Unnamed Valley Creek tributaries (5)	Unoccupied	3.8
Total		**4.7**

[a]From the blackside dace 5-year review (U. S. Fish and Wildlife Service 2015a). Streams with no blackside dace occurrence records are identified as "unoccupied" but conservatively included because of the potential for temporary, transient, or future permanent occupation. Entire length of stream considered to be potentially temporarily occupied by individuals moving to and from more appropriate areas, or otherwise temporary occupancy by waifs from other areas.

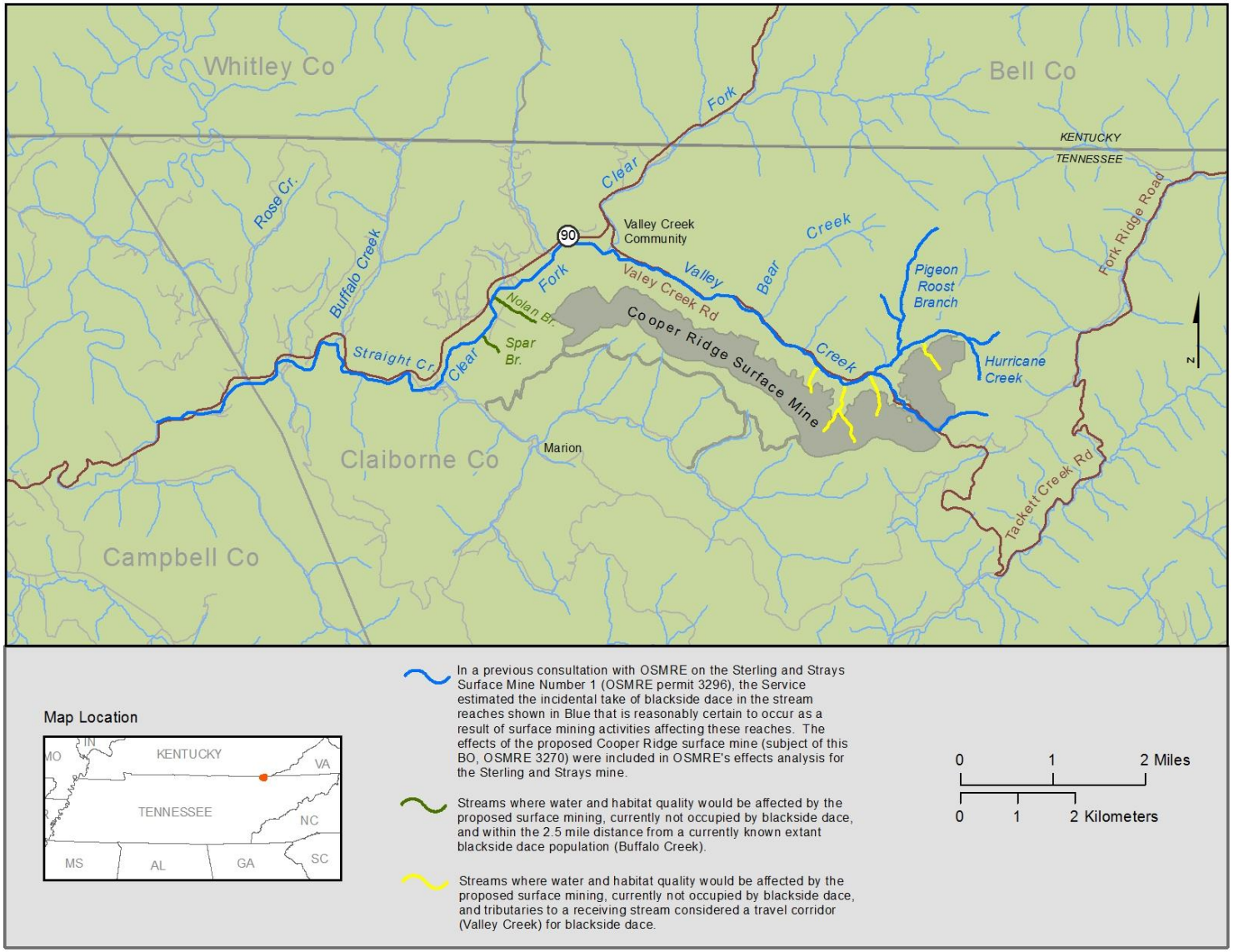

Figure 11. Stream reaches included in the blackside dace incidental take authorization for the Kopper Glo Mining, LLC Cooper Ridge Surface Mine (OSMRE permit 3270), Claiborne County, Tennessee.

We developed two scenarios (outlined below) to estimate the number of blackside dace individuals that could be affected when exposed to altered water quality in the 4.7 miles of streams affected by the action.

Category 1 stream reaches:

Category 1 includes Spar Branch and Nolan Branch, identified by the yellow lines in Figure 11 above (and see Table 10). Neither of these streams are currently occupied, or known to have been historically inhabited by blackside dace, but are located within a reasonable distance for migrating individuals to invade in the future, especially if water and habitat in migratory corridors improves as a result of remining that results in reclamation of formerly abandoned and unreclaimed mines upstream in the watershed.located within (approximately) the 2.5 mile distance from currently known extant population (Buffalo Creek) reported by Mattingly et al. (2005) and Detar and Mattingly (2013) for migrating blackside dace, and Nolan Branch (located 2.9 miles from Buffalo Creek). The water and habitat quality in Spar Branch and Nolan Branch would be affected by runoff and discharges from the minesite.

We consider habitat in these 0.9 total miles of Spar Branch and Nolan Branch to be sub-optimal for invading blackside dace because they drain areas previously mined and unreclaimed. However, with passage of time following cessation of active mining, and as reclamation matures on the site, this should improve. Therefore the following criteria were used to quantify the number of blackside dace that might be affected in these 0.9 miles of Category 1 streams:

- During the time period that OSMRE retains regulatory authority for permit 3270, some individuals in the Buffalo Creek blackside dace population may move the approximately 2.5 mile distance reported by Mattingly et al. (2005) and Detar and Mattingly (2013) for migrating individuals, to enter Spar Branch or Nolan Branch.
- Migrating blackside dace individuals could be affected as they enter Spar Branch or Nolan Branch stream reaches where habitat or water quality conditions are affected by runoff and/or discharges from the mine site.
- The total stream lengths of Spar Branch and Nolan Branch are 0.9 miles (1.4 km) or 7.2 200-m reaches. Mattingly et al. (2005, Black and Mattingly 2007, Black et al. 2013b) used 200-m reaches to estimate population sizes for 12 streams considered to have the robust blackside dace populations.
- We assume that appropriate blackside dace habitat and water quality is present throughout half of the 0.9 miles of stream reaches included in Category 1, or 0.5 miles (0.5 miles = 804 m and 4.1 200 m reaches). This assumption is based on habitat assessments included in TWRA reports (Tennessee Wildlife Resources Agency 2011, 2012, 2013, 2014; Carter 2016) and the TDEC 303d list (Tennessee Department of Environment and Conservation 2016c) that habitat and water quality in many stream reaches have been affected by earlier coal mining.
- Migrating blackside dace could occupy these stream reaches year-round.
- Within the time period that OSMRE retains regulatory authority of permit 3270, a persistent blackside dace population could become established in these stream reaches.

- Blackside dace that might persist in these "sub-optimal" stream reaches would occur at the lowest values estimated by TWRA for 2011, 2012 (or 9 individuals/200 m).

From the assumptions above, a conservative (worst case) estimate of blackside dace individuals potentially affected in the 0.9 stream miles of category 1 streams (Spar Branch and Nolan Branch) is estimated as:

> **4.1 (200 m reaches) x 9 (individuals per 200-m reaches for category 2 stream reaches) = 37 individuals.**

Recall, as discussed in sections 4.1.8 and 4.5.3.2 (*Literature review – impacts of altered ground and surface water chemistry to downstream biological ommunities*, and *Indirect adverse effects to blackside dace*, respectively) we do not expect the incidental take to be lethal; rather, feeding, breeding, and sheltering behavior are assume to potentially be affected by adverse water quality related to the mine. Therefore, while the number of individuals potentially inhabiting these stream reaches could increase with increasing blackside dace abundance, the number of individuals adversely affected (as described above) by mine-related water quality should decrease over time as water and habitat quality improves because of successful reclamation and ecological succession that affects ground and surface water quality and quantity. However, as described above, the length of time that water quality impacts (elevated conductivity) related to the mine runoff and/or discharges could persist are not known; we assume there will be gradual improvement of water quality over five decades. Blackside dace lifespan is short (2-4 years). Therefore, we consider that every year, a new group of individuals could be affected by temporary or permanent conditions in the stream reaches included in this BO. As a result, up to 37 individuals could be affected per year, with diminishing numbers as time passes.

<u>Category 2 stream reaches:</u>

The consultation for Middlesboro Mining Company, Inc. Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296) authorized blackside dace incidental take (U. S. Fish and Wildlife Service 2016a) for the section of Valley Creek which receive drainage from five unnamed tributaries draining the mine site (Figure 11). In this Middlesboro Mining, Inc. Sterling and Strays BO, a persistent blackside dace occurrence in Valley Creek was considered unlikely. However, because elevated conductivity resulting from the mining may persist for decades or up to a century, the BO considered that a few individual blackside dace traveling through affected stream reaches in attempts to colonize new areas in the watershed may be indirectly affected by hydrologic, and habitat and water quality effects of the mining. Consequently incidental take was authorized for those blackside dace individuals that may be affected by hydrology, habitat quality, or water quality as they travel through or temporarily occupy affected stream reaches for the length of time that OSMRE maintains regulatory authority (e.g., through Phase III Bond Release).

Four unnamed streams are tributaries to the reach of Valley Creek included in this incidental take authorization. These tributaries originate on the Cooper Ridge Surface Mine permit (total of 2.2 miles of stream channels, see Figure 11 and Table 9). These streams, all of which have been previously affected by unreclaimed surface mining, will all be affected by the mining as the mine

plan calls for "mining through" them, with subsequent restoration (Tennessee Department of Environment and Conservation 2016a) intended to improve adverse water and habitat quality remaining from the previous perturbations. While these streams are located more than the 2.5 mile dispersal distance reported by Mattingly et al. (2005) and Detar and Mattingly (2013) for migrating blackside dace individuals, assuming the predicted water quality improvement in Clear Fork and Valley Creek is realized, these 2.2 miles of tributary streams could eventually be colonized by a longer term dispersal mechanism (eg., the "stepping stone" model of dispersal described by Katz et al. 2014, and see *Literature review – effects of altered ground and surface water chemistry to fish dispersal and genetic variation of metapopulations*, section 4.1.9 in this BO).

The following assumptions were used to estimate the number of blackside dace individuals that could be affected in the 2.2 miles of streams affected by the action.

- Water quality is expected to gradually improve following stream restoration and reclamation, but conductivity will likely remain above the 343µS/cm threshold for blackside dace reported by Hitt et al. (2016) through the time period when OSMRE maintains regulatory authority (e.g., through Phase III Bond Release).
- Migrating blackside dace individuals could be affected as they enter any of these unnamed tributaries where habitat or water quality conditions are affected by runoff and/or discharges from the mine site;
- The total stream lengths of these four unnamed tributaries are 2.2 miles (3.5 km) or 17.7 200-m reaches. Mattingly et al. (2005, Black and Mattingly 2007, Mattingly et al. 2013b) used 200-m reaches to estimate population sizes for 12 streams considered to have the robust blackside dace populations;
- Because portions of these stream channels will be "mined through" and restored, we assume a lower proportion (25%) of appropriate blackside dace habitat and water quality to be present, or approximately one total mile of stream channels (0.6 mile = 1966 m and 4.8 200 m reaches).
- Migrating blackside dace could occupy these stream reaches year-round.
- Within the time period that OSMRE retains regulatory authority of permit 3270, persistent blackside dace populations could become established in these stream reaches.
- Blackside dace that might persist in thsese "sub-optimal" stream reaches would occur at the lowest values estimated by TWRA for 2011, 2012 (or 9 individuals/200 m).

From the assumptions above, a conservative (worst case) estimate of blackside dace individuals potentially affected in the 2.2 stream miles of category 2 streams (four unnamed Valley Creek tributaries) is:

> **4.8 (200 m reaches) x 9 (individuals per 200-m reaches for category 2 stream reaches) = 43 individuals.**

Recall, as discussed in sections 4.1.8 and 4.5.3.2 (*Literature review – impacts of altered ground and surface water chemistry to downstream biological ommunities*, and *Indirect adverse effects to blackside dace*, respectively) we do not expect the incidental take to be

lethal; rather, feeding, breeding, and sheltering behavior are assume to potentially be affected by adverse water quality related to the mine. Therefore, while the number of individuals potentially inhabiting these stream reaches could increase with increasing blackside dace abundance, the number of individuals adversely affected (as described above) by mine-related water quality should decrease over time as water and habitat quality improves because of successful reclamation and ecological succession that affects ground and surface water quality and quantity. However, as described above, the length of time that water quality impacts (elevated conductivity) related to the mine runoff and/or discharges could persist are not known; we assume there will be gradual improvement of water quality over five decades. Blackside dace lifespan is short (2-4 years). Therefore, we consider that every year, a new group of individuals could be affected by temporary or permanent conditions in the stream reaches included in this BO. As a result, up to 43 individuals could be affected per year, with diminishing numbers as time passes.

Therefore, as described above, this BO estimates that up to 80 blackside dace individuals could be affected per year for the length of time OSMRE retains regulatory authority over this reclamation and until bond release (Tables 9 and 10 and Figure 11). In addition, since the proposed surface mining includes no new areas to be surface mined, and consists totally of remining and reclaiming areas that have previously not been reclaimed, we expect the number of blackside dace individuals to be affected per year to continually decrease over the time period that OSMRE retains regulatory authority of this permit. As indicated in the Literature review sections (4.1) above, the mechanism(s) of effects of elevated conductivity to individual blackside dace impact from elevated conductivity are not fully understood. In fact, laboratory tests using blackside dace eggs and adults in water with elevated conductivity did not indicate direct toxicity to individuals (see reference to Hitt et al. 2016 comments about toxicity testing of blackside dace in section 4.2). Therefore, the Service has concluded that any take resulting from the activities included in this consultation would not be lethal, and would instead be in the form of harm or harassment. Because of the spatial and temporal scope and form of the identified incidental take, it would be difficult to monitor. Observing/collecting and quantifying blackside dace whose feeding, breeding, or sheltering behavior had been affected by the identified stressor throughout the 3.1miles of stream channels during the time period(s) when they might be present is impractical. Therefore, the blackside dace incidental take will be monitored with the use of surrogate water quality measures, especially trends over time in comparison with baseline levels of these water quality measures.

Table 9. Incidental take estimates for blackside dace in various stream reaches within the action area considered in this BO. Methodology and assumptions used to develop the estimates for the various stream categories are described above. See Figure 11 for stream reach designations by color.

Stream reach category (color, Figure 11)	**Characteristics of blackside dace assumed to occupy stream reaches where incidental take is authorized**		**Estimated individuals "taken"**
Category 1 (green)	Immigrant population that could establish and persist by moving approximately 2.5 stream miles into mine impacted stream reaches from known population (Buffalo Creek)	0.9	37
Category 2 (yellow)	Immigrant population that could establish and persist as immigrants moving into mine impacted stream reaches from assumed travel corridor in Valley Creek	2.2	43
TOTAL		**3.1**	**80**

Previous biological opinions, completed for Tennessee populations of Indiana bats, and blackside dace range-wide, which identified incidental take, have been included in a table as Appendix A.

7.2. Reasonable and prudent measures

The measures described below are non-discretionary, and must be undertaken by OSMRE, so they become binding conditions of any permits or contracts, as appropriate, for the exemption in section 7(o)(2) to apply. OSMRE has a continuing duty to regulate the activity covered by this Incidental Take Statement. If OSMRE (and ultimately the project proponent, Kopper Glo Mining, Inc.): (1) fails to assume and implement the T&Cs or (2) fails to adhere to the T&Cs of the Incidental Take Statement through enforceable terms that are added to the grant, permit or contract, the protective coverage of section 7(o)(2) may lapse. In order to monitor the effect of incidental take, OSMRE must report the progress of the action and its effect on the species to the Service as specified in the Incidental Take Statement. (50 CFR § 402.14 [1][3]).

The Service believes the following reasonable and prudent measures are necessary or appropriate to minimize the anticipated taking of northern long-eared bat, Indiana bat, and blackside dace that is incidental to the action and that is not excepted taking under the interim 4(d) rule for this species.

The Service believes the following RPMs are necessary and minimize impacts of incidental take of the Indiana bat, and blackside dace:

1. OSMRE must ensure that the proposed action will occur as designed, planned and documented in the biological assessment, all supporting information provided by the OSMRE and Kopper Glo Mining, LLC, and this biological opinion.
2. OSMRE must ensure that Kopper Glo Mining, LLC implements the conservation measures and avoidance and minimization measures included in *Description of the proposed action,* and including the *Conservation measures* (sections, 1.1 and 1.3.6 in this BO, respectively), to avoid, minimize and offset impacts associated with the proposed action.
3. OSMRE must ensure that its inspectors and Kopper Glo Mining, LLC adequately monitor the proposed action to document potential changes to terrestrial habitat and habitat and to water quality conditions of receiving streams resulting from the action.

7.3. Terms and conditions

The Service believes the following terms and conditions (T&Cs) are necessary and minimize impacts of incidental take of the Indiana bat and blackside dace. In order to be exempt from the prohibitions of section 9 of the Act, OSMRE and Kopper Glo Mining, LLC must comply with the following T&Cs, which carry out the RPMs described above. These T&Cs are non-discretionary.

A. OSMRE will ensure that Kopper Glo Mining, LLC strictly adheres to BMPs, as included in the description of the proposed mining operations and reclamation, and as required by

SMCRA regulations and the conditions of the SMCRA permit. This T&C supports RPMs 1 and 2.

B. OSMRE will promptly notify the Service of any substantial issues noted during routine inspections that could result in impacts to blackside dace. This T&C supports RPMs 1 and 2.
C. OSMRE will collaborate with the Service and other partners to adjust water and habitat quality and biological sampling requirements associated with this permit to support broad range-wide data needed to identify trends in blackside dace distribution and population sizes, as much as practicable. This T&C supports RPM3
D. OSMRE will collaborate with the Service and other partners to analyze data from T&C "C" above to verify the accuracy of predictions included in OSMRE's BA and upon which the incidental take authorized in this BO is based. This T&C supports RPM3.
E. OSMRE will provide the Service with summary annual reports describing mining and reclamation activities taking place on the mine site and summarizing results of inspections, water quality investigations, and biological investigations. This T&C supports RPM 1, 2, and 3.
F. OSMRE will work with the Service to regularly (at least annually) review the results of the water and habitat quality and blackside dace sampling programs identified above. If water quality, habitat, and blackside dace population trends related to the Cooper Ridge Surface Mine become apparent, indicating the authorized incidental take level is being approached, adaptive management actions will be identified and implemented, as appropriate. This T&C supports RPM

The RPMs, with their implementing T&Cs, are designed to minimize the effect of incidental take that might otherwise result from the proposed action. The Service believes that no more than 180 bats in a total of 7,476 acres (in the 1,496-acre permit boundary and an adjacent 5,980 acres of a ½ mile buffer around the mine permit boundary) and up to 80 blackside dace individuals each year as they immigrate and attempt to establish popuolations on 3.1 miles of stream channels (Tables 9 and 10 and Figure 11) in watersheds affected by the mining will be incidentally taken.

As indicated in the literature review and effects analysis (sections 4.1 and 4.4, respectively) the mechanism(s) of blackside dace impact (including toxicity testing) from elevated conductivity have not been identified. Therefore, the Service has concluded that this take would not be lethal, and would instead be in the form of harm or harass. Because of the spatial and temporal scope and form of the identified incidental, it would be difficult to monitor. Observing/collecting and quantifying 80 or fewer blackside dace whose feeding, breeding, or sheltering behavior had been affected by the identified stressor over 3.1 miles of stream channels during the time period(s) when they might be present is impractical. Therefore, surrogate measures to monitor this take will include analyzing water quality and biological conditions in these receiving streams (identified in T&Cs "C" and "D") while mining takes place and during reclamation and comparing those identified parameters (conducitivity) with those projected in OSMRE's BA, EA, and CHIA and in this BO.

The incidental take included in this BO is authorized for OSMRE until such time as OSMRE no longer maintains regulatory authority (e.g., through Phase III Bond Release). The incidental take included in this BO is authorized for OSMRE until such time as OSMRE no longer maintains regulatory authority (e.g., through Phase III Bond Release). Since the prior blackside dace

incidental take authorization for the Sterling and Strays Surface Mine Number 1 (OSMRE permit 3296, and see U. S. Fish and Wildlife Service 2016a) was considered sufficient in scope to cover the effects to blackside dace from the activities included in this Cooper Ridge Surface Mine (OSMRE permit 3270) consultation, that prior authorization extends to OSMRE and Kopper Glo Mining, LLC for the Cooper Ridge Surface Mine (permit 3270). However, if OSMRE retains regulatory authority for the Cooper Ridge Surface Mine longer than Sterling and Strays Surface Mine Number 1, the blackside dace incidental take authorization would then apply to Cooper Ridge Surface Mine until such time as OSMRE no longer maintains regulatory authority (e.g., through Phase III Bond Release) for the Cooper Ridge Surface Mine.

If, during the course of the action, this level of incidental take is exceeded, as identified by evaluating the surrogate water quality and biological data, and identified in collaboration with the Service, such incidental take represents new information requiring reinitiation of consultation and review of the RPMs provided. OSMRE must immediately provide an explanation of the causes of the taking and review with the Service the need for possible modification of the RPMs.

7.4. Monitoring and reporting requirements

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 Service as specified in the incidental take statement (50 CFR §402.14(i)(3)). The incidental take monitoring and reporting requirements applies to the action covered under this statement and shall facilitate documenting projects that are excepted from the taking prohibitions applicable to the Indiana bat and blackside dace, and that subsequent to such documentation, require no further consultation under section 7(a)(2) of the ESA to proceed lawfully. Specifically the required monitoring is identified in T&Cs B and E above, may also include other reports or documents (deliverable and schedule to be defined according to results of collaboration between the Service and OSMRE) to address T&Cs D and F above.

8. Conservation recommendations

Section 7(a)(1) of the Act directs Federal agencies to use their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help carry out recovery plans, or to develop information.

We offer the following conservation recommendations for consideration:

1. OSMRE should convene science workshops to:
    - Discuss the most current state of knowledge regarding known and suspected impacts to listed and at-risk species from mine-related water quality impacts.
    - Identify mechanisms to communicate findings to broader (i.e., mine applicants, consultants, state water quality regulators) audiences and draft communication plan,
    - Identify specific range-wide blackside dace conservation needs & identify actions appropriate for OSMRE and applicants, including recommendations for mitigation.

- Draft document(s) including these needs and actions in order to assist OSMRE and the Service in providing applicants for SMCRA permits with opportunities to actively participate in species conservation and recovery efforts and also to assist in accomplishing SMCRA requirements and to facilitate the regulatory process.

2. OSMRE should provide support for future research efforts that focus on methods to mitigate legacy effects of past mining on stream water quality, and to the affected aquatic biological communities within a landscape that includes historical and current mining.
3. OSMRE should conduct surveys to identify if Indiana bats or northern long-eared bats use habitats in adjacent areas for roosting during the non-hibernating season, and if either or both species are found to occupy these areas, estimate the number of and types of roosts found there.
4. OSMRE should encourage mine applicants, including Kopper Glo Mining, LLC, to maintain documentation for areas where fish barriers may prevent blackside dace movement throughout streams affected by surface mining. OSMRE and the Service, with input from other stakeholders, could prioritize these areas for possible future recovery (enhancement) efforts.
5. OSMRE should convene workshops that include a variety of stakeholders, including mining companies and conservationists, to develop a conservation plan for blackside dace and other aquatic species potentially affected by surface coal mining in the southern Appalachian coal fields (see discussion of decision support tools in *Literature review – impacts of surface mining to the specific conductance component of water quality* (section 4.2.6 in this BO).

9. Reinitiation notice

This concludes formal consultation on Office of Surface Mining Reclamation and Enforcement approval of Surface Mining at Kopper Glo Mining, LLC Cooper Ridge Surface Mine, OSMRE permit number 3270 (the action). As written in 50 CFR Section 402.16, reinitiation of formal consultation is required where discretionary OSMRE involvement or control over the action have been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the OSMRE action that may affect listed species or critical habitat in a manner or to an extent not considered in this biological opinion; (3) the OSMRE action is later modified in a manner that causes an effect to a listed species or critical habitat not considered in this biological opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease until reinitiation.

For this biological opinion, the incidental take would be exceeded when the take exceeds: (1) 180 Indiana bats occupying 7,476 acres of forested habitat that includes 1,496 acres on the Cooper Ridge Surface Mine permit boundary and including a 5,980-acre buffer area adjacent and around the mine permit boundary where mine development and operations would occur and, (2), up to 80 individual blackside dace per year for up to 100 years migrating through approximately 3.1 miles of stream channels affected by the mining, which is what has been exempted from the taking prohibitions under 50 CFR §17.31 and §17.32 by this biological opinion. The Service appreciates the cooperation of OSMRE during this consultation. For further information regarding this consultation, please contact Peggy Shute at 931/525-4982.

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Appendix A

Total Incidental take of Indiana bat in Tennessee and blackside dace in Tennessee and Kentucky.

The following list includes previous biological opinions, issued for adverse impacts and completed for the blackside dace (within Kentucky and Tennessee) and the Indiana bat (within Tennessee), which identified incidental take:

SPECIES	OPINIONS (year)	INCIDENTAL TAKE NUMBERS	HABITAT	
			Critical Habitat	Suitable Habitat
Blackside dace	1995	0	N/A	2 river miles of Ryan's Creek (instream habitat below Chitwood Hollow) in McCreary County, Kentucky
	2007	All individuals in Mill Branch watershed, Kentucky	N/A	Not specified
	2010	161 individuals in the Dog Slaughter Creek watershed, Kentucky	N/A	Not specified
	2016	A percentage of blackside dace in a 12.3 river mile reach of the Rock Creek system, McCreary County, Kentucky	N/A	Not specified
	2016	Up to 9 individuals per year traveling through 12.6 miles of the Davis Creek watershed, Campbell County, Tennessee	N/A	Not specified

SPECIES	**OPINIONS (year)**	**INCIDENTAL TAKE NUMBERS**	**HABITAT**	
			Critical Habitat	**Suitable Habitat**
	2016	Up to 38 individuals per year traveling through 52 stream miles of the Clear Fork system, Claiborne County, Tennessee	N/A	Not specified
	2016	Up to 1,105 individuals per year occupying or traveling through 71 miles of streams in the Clear Fork and Jellico Creek systems, Campbell andn Scott counties, Tennessee	N/A	Not specified
Indiana bat	1997	Not specified		
	2003	One Indiana bat maternity colony in Great Smoky Mountain National Park, Blount and Sevier Counties, Tennessee, and Haywood County, North Carolina		
	2013	Not specified	N/A	1,300 ac/year, Cherokee National Forest, Tennessee

	2015	Not specified	N/A	Not specified
SPECIES	**OPINIONS (year)**	**INCIDENTAL TAKE NUMBERS**	**HABITAT**	
			Critical Habitat	**Suitable Habitat**
	2015	Not specified	N/A	3,679.7 acres in the Middle Citico Creek Watershed, the Cherokee National Forest, Tennessee
	2015	An unknown number of Indiana bats and northern long-eared bats within 15,925 acres of suitable summer habitat on Arnold Airforce Base in Coffee and Franklin counties, Tennessee	N/A	25,700 acres on the Southern Districts of the Cherokee National Forest, Tennessee
	2016	All Indiana bats occurring within 13.2 acres of suitable Indiana bat summer roosting habitat in Van Buren County, Tennessee.	N/A	32.5 acres of suitable swarming or roosting habitat, and impacts to greater than 0.23-acre of wetlands in 717 linear feet of stream length in Van Buren and Bledsoe Counties, Tennessee

Appendix B

Comparison of Sediment Pond Design Standards for States in the Appalachian Coal Mining Region

Tennessee

- Sediment storage is 0.2 acre-feet/disturbed acre (0.2 feet in TN Regs at 942.816d)
    - Minimum value is 0.1 – but can only be used if BMP's are included in sediment control plan
- Pond volume is designed via SedCAD with sediment storage input (0.1 to 0.2)
- Sediment cleanout level = 80% of maximum sediment storage capacity
- Dewatering pipe inlet elevation = 1.0 foot above sediment storage cleanout elevation
- Use peak sediment concentration to arrive at 0.5 ml/L
- Dewater time = 24 hours min

Virginia

- Sediment storage is 0.125 acre-feet/disturbed acre
- Pond volume is 0.125 acre-feet/disturbed area
- Sediment cleanout level = 60% of maximum sediment storage capacity
- Dewatering pipe inlet elevation = 0.5 foot above sediment storage cleanout elevation
- Use peak sediment concentration to arrive at 0.5 ml/L

Kentucky

- Sediment storage is 0.125 acre-feet/disturbed acre
- Pond volume is designed via SedCAD with sediment storage input (0.125 acre-ft)
- Sediment cleanout level = 100% of maximum sediment storage capacity
- Dewatering pipe inlet elevation = 1.5 foot above sediment storage cleanout elevation
- Use arithmetic average of peak sediment concentration to arrive at 0.5 ml/L

Ohio

- Sediment storage is 1000 cubic feet (0.023 acre-feet)/disturbed acre
- Pond volume is 1800 cubic feet (0.041 acre-ft)/watershed area
- Sediment cleanout level = 100% of maximum sediment storage capacity
- Dewatering pipe inlet elevation = at sediment storage cleanout elevation
- Dewater time = 48 hour mininum, 7 days max

Pennsylvania

- Sediment storage is 1000 cubic feet (0.023 acre-feet)/disturbed acre
- Pond volume is 5000 cubic feet (0.115 acre-ft)/watershed area
- Sediment cleanout level = 100% of maximum sediment storage capacity
- Dewatering pipe inlet elevation = 0.5 or 1.0 feet above sediment storage cleanout elevation
- Dewater time = 48 hour minimum, 7 days max

West Virginia

- Sediment storage is 0.125 acre-feet/disturbed acre
- Pond volume is designed via SedCAD with sediment storage input (0.125 acre-ft)
- Sediment cleanout level = 100% of maximum sediment storage capacity
- Dewatering pipe inlet elevation = same elevation as sediment storage cleanout elevation
- Use peak sediment concentration to arrive at 0.5 ml/L

Appendix C

Statistics and graphical representation of data from continuously recording conductivity meters placed at three locations within the action area of this consultation.

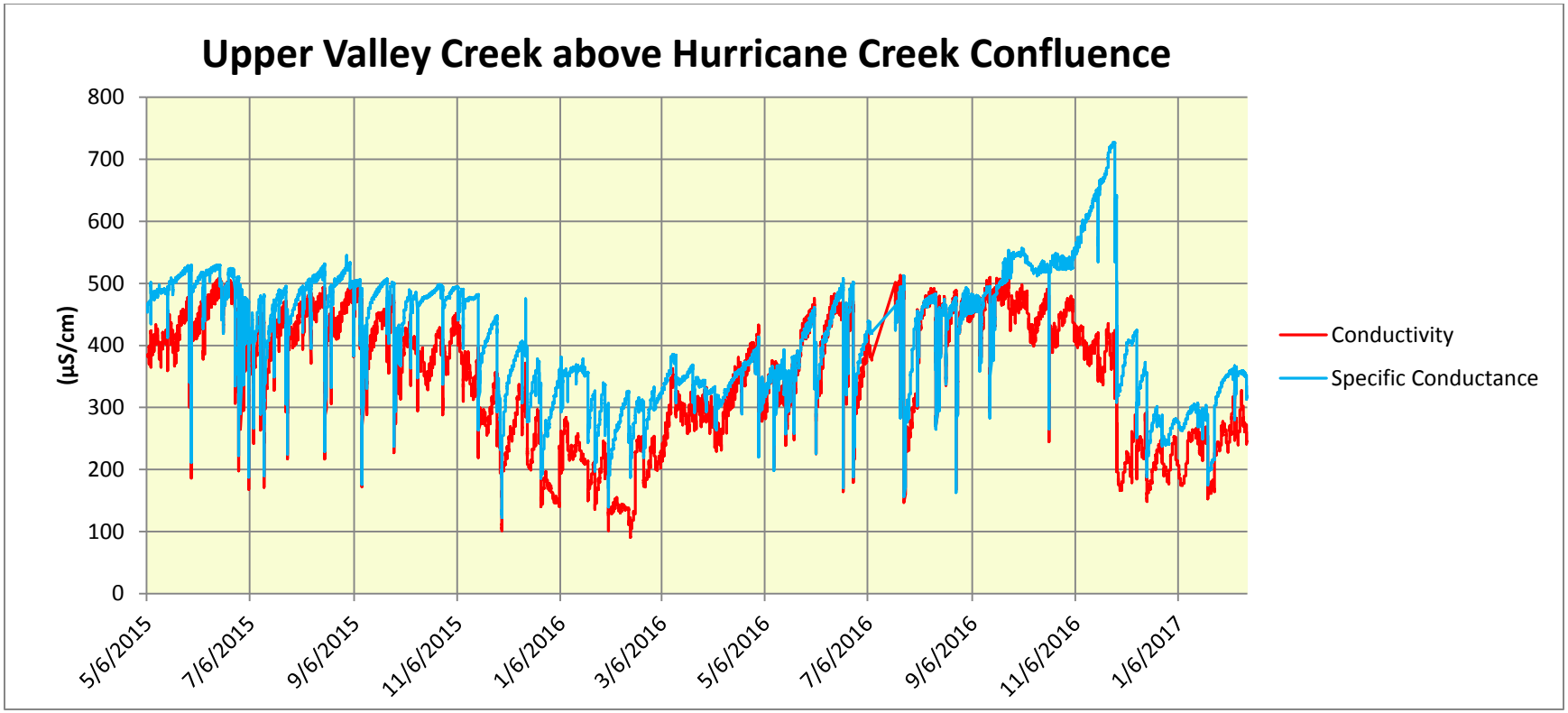

Figure C-1. Conductivity and specific conductance in upper Valley Creek headwaters (upstream of the mouth of Hurricane Creek) May 2015 through January 2017, as measured by continuously recording data loggers (OSMRE unpublished data 2017).

Conductivity, µS/cm		*Specific Conductance, µS/cm*	
Mean	349.8125281	Mean	415.3577448
Standard Error	0.586776769	Standard Error	0.560157508
Median	374.18	Median	425.5
Mode	445.55	Mode	476.7
Standard Deviation	102.5769039	Standard Deviation	97.92347937
Sample Variance	10522.02122	Sample Variance	9589.007811
Kurtosis	-1.009707464	Kurtosis	-0.294218592
Skewness	-0.432773777	Skewness	0.056712383
Range	422.09	Range	605.2
Minimum	90.91	Minimum	122.5
Maximum	513	Maximum	727.7
Sum	10690270.86	Sum	12693332.68
Count	30560	Count	30560
Confidence Level (95.0%)	1.150106888	Confidence Level (95.0%)	1.097932027

Table C-1. Summary statistics for the data from the continuously-recording data logger in upper Valley Creek (OSMRE unpublished data 2017) and illustrated in Figure C-1.

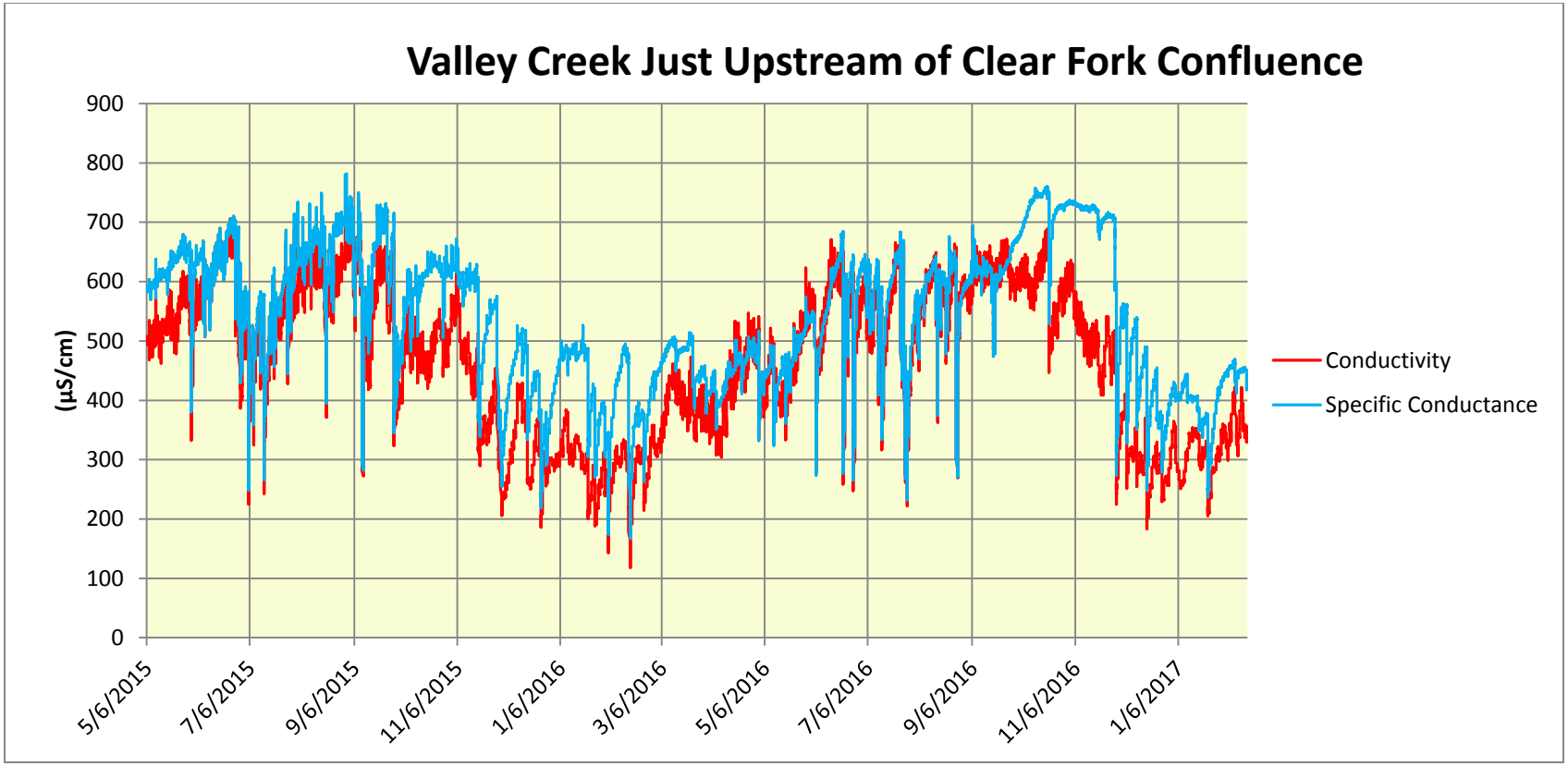

Figure C-2. Conductivity and specific conductance in lower Valley Creek just upstream of its confluence (mouth) at the Clear Fork May 2015 through January 2017, as measured by continuously recording data loggers (OSMRE unpublished data 2017).

Conductivity, µS/cm		*Specific Conductance, µS/cm*	
Mean	465.0198491	Mean	538.1800483
Standard Error	0.715860522	Standard Error	0.657452503
Median	484.035	Median	547.65
Mode	308.39	Mode	483.8
Standard Deviation	126.6081856	Standard Deviation	116.2780541
Sample Variance	16029.63266	Sample Variance	13520.58586
Kurtosis	-1.129248056	Kurtosis	-0.844311721
Skewness	-0.225183408	Skewness	-0.165933347
Range	619.66	Range	613.5
Minimum	118.14	Minimum	168.2
Maximum	737.8	Maximum	781.7
Sum	14545820.88	Sum	16834271.91
Count	31280	Count	31280
Confidence Level (95.0%)	1.403115136	Confidence Level (95.0%)	1.288633092

Table C-2. Summary statistics for the data from the continuously-recording data logger in lower Valley Creek (OSMRE unpublished data 2017) and illustrated in Figure C-2.

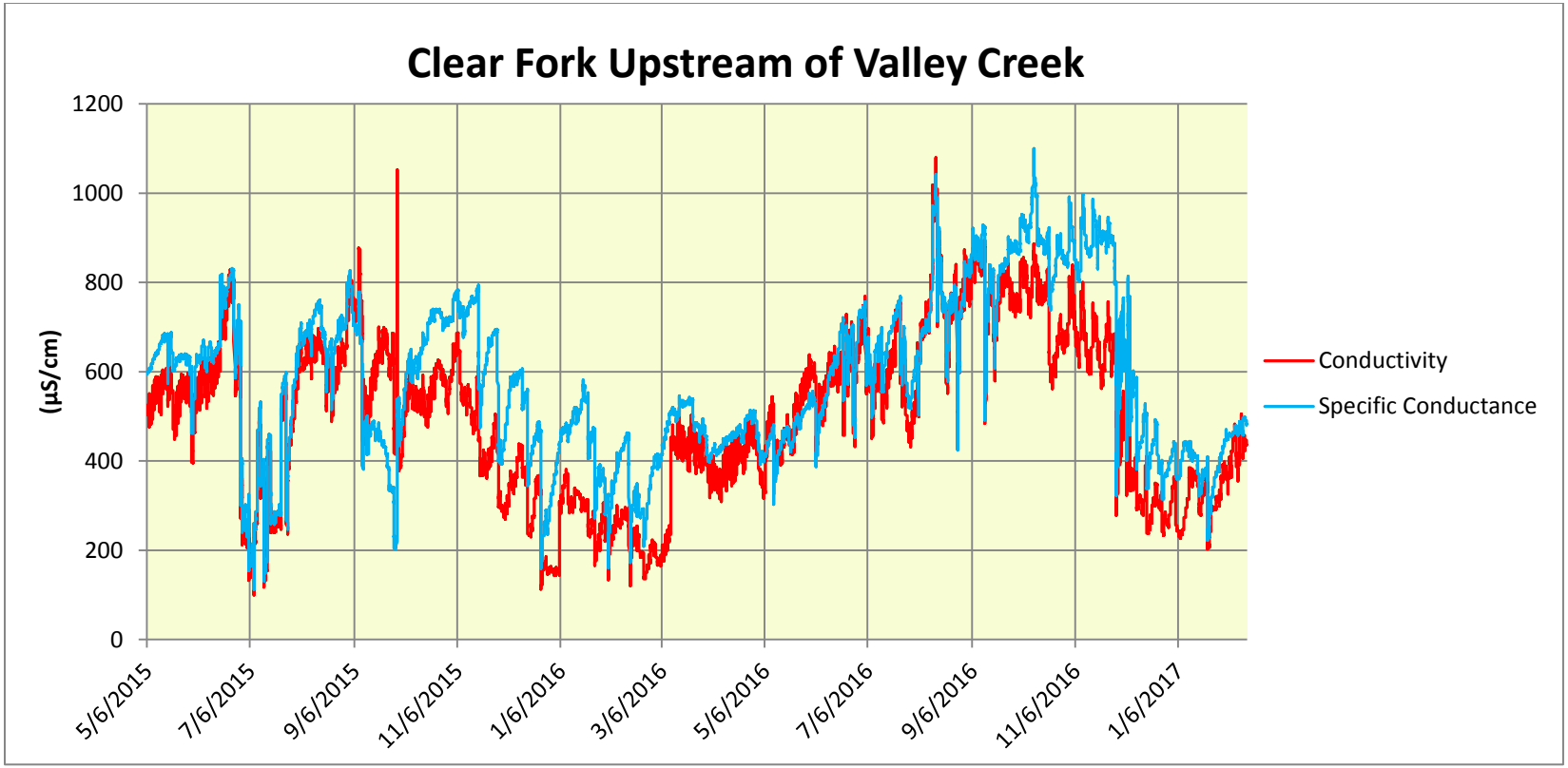

Figure C-3. Conductivity and specific conductance in Clear Fork just upstream of its confluence with Valley Creek at the May 2015 through January 2017, as measured by continuously recording data loggers (OSMRE unpublished data 2017).

Conductivity, µS/cm		*Specific Conductance, µS/cm*	
Mean	499.4032915	Mean	579.2250832
Standard Error	1.052734582	Standard Error	1.035775177
Median	511.145	Median	570.7
Mode	290.48	Mode	458
Standard Deviation	186.0453363	Standard Deviation	183.0481722
Sample Variance	34612.86714	Sample Variance	33506.63336
Kurtosis	-0.777257703	Kurtosis	-0.661744062
Skewness	0.043266896	Skewness	0.191439952
Range	979.66	Range	988.3
Minimum	99.74	Minimum	112
Maximum	1079.4	Maximum	1100.3
Sum	15597363.6	Sum	18090357.8
Count	31232	Count	31232
Confidence Level (95.0%)	2.063401834	Confidence Level (95.0%)	2.030160723

Table C-3. Summary statistics for the data from the continuously-recording data logger in Clear Fork just upstream of Valley Creek confluence (OSMRE unpublished data 2017) and illustrated in Figure C-3.

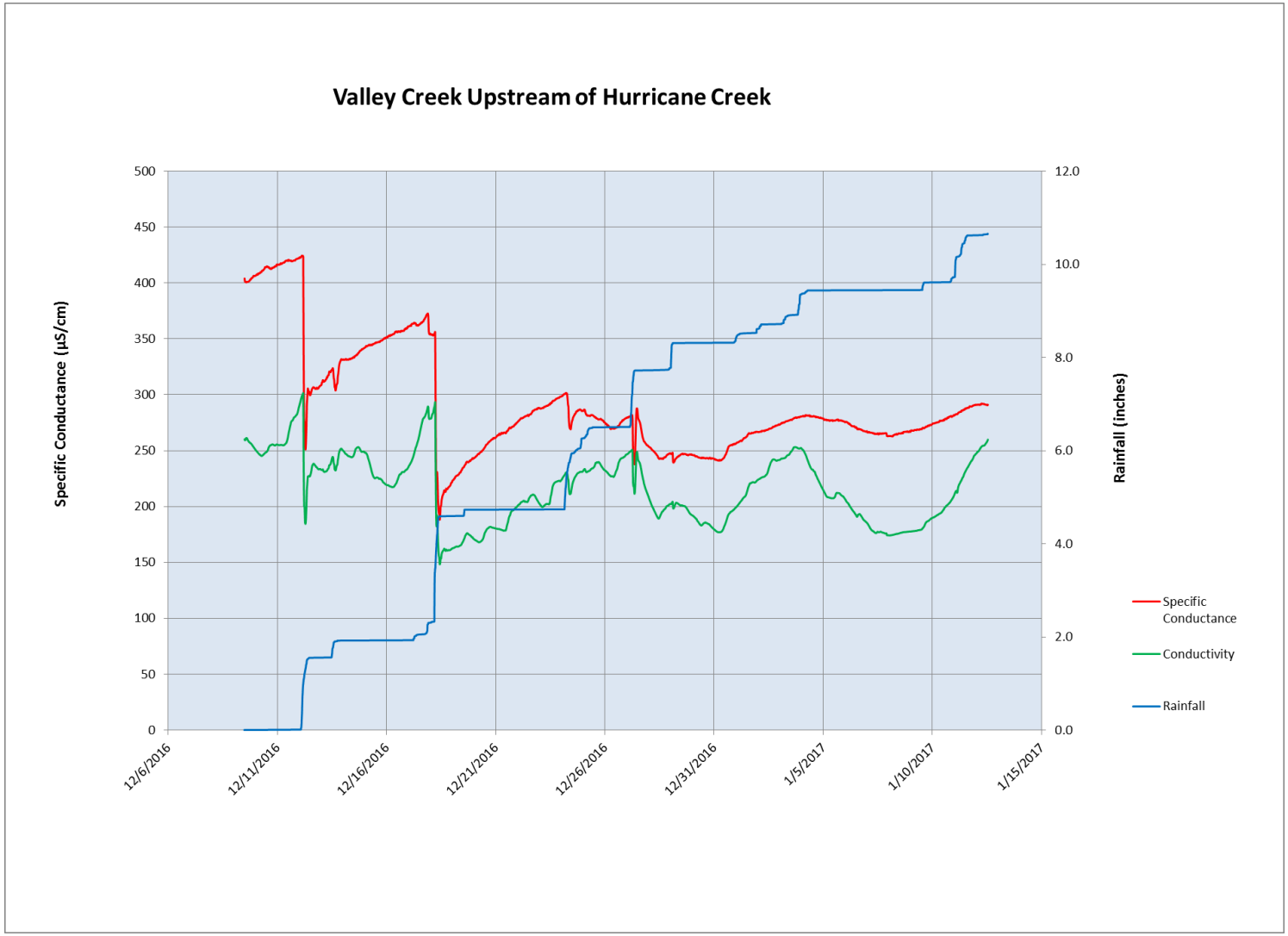

Figure C-4. Conductivity and specific conductance and rainfall in upper Valley Creek May 2015 through January 2017, as measured by continuously recording data loggers (OSMRE unpublished data 2017).

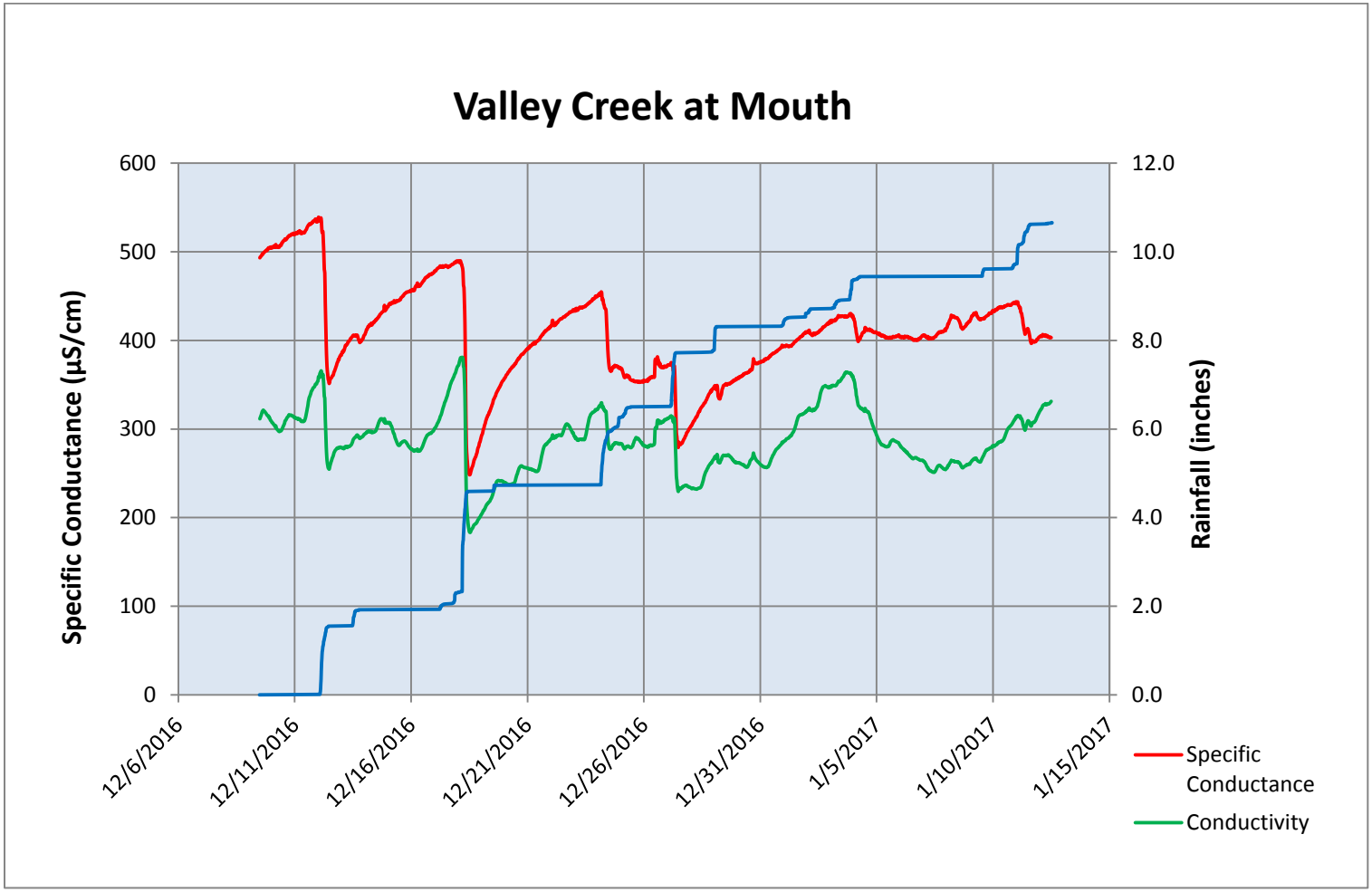

Figure C-5. Conductivity and specific conductance and rainfall in Vallley Creek just upstream of its confluence with Clear Fork May 2015 through January 2017, as measured by continuously recording data loggers (OSMRE unpublished data 2017).

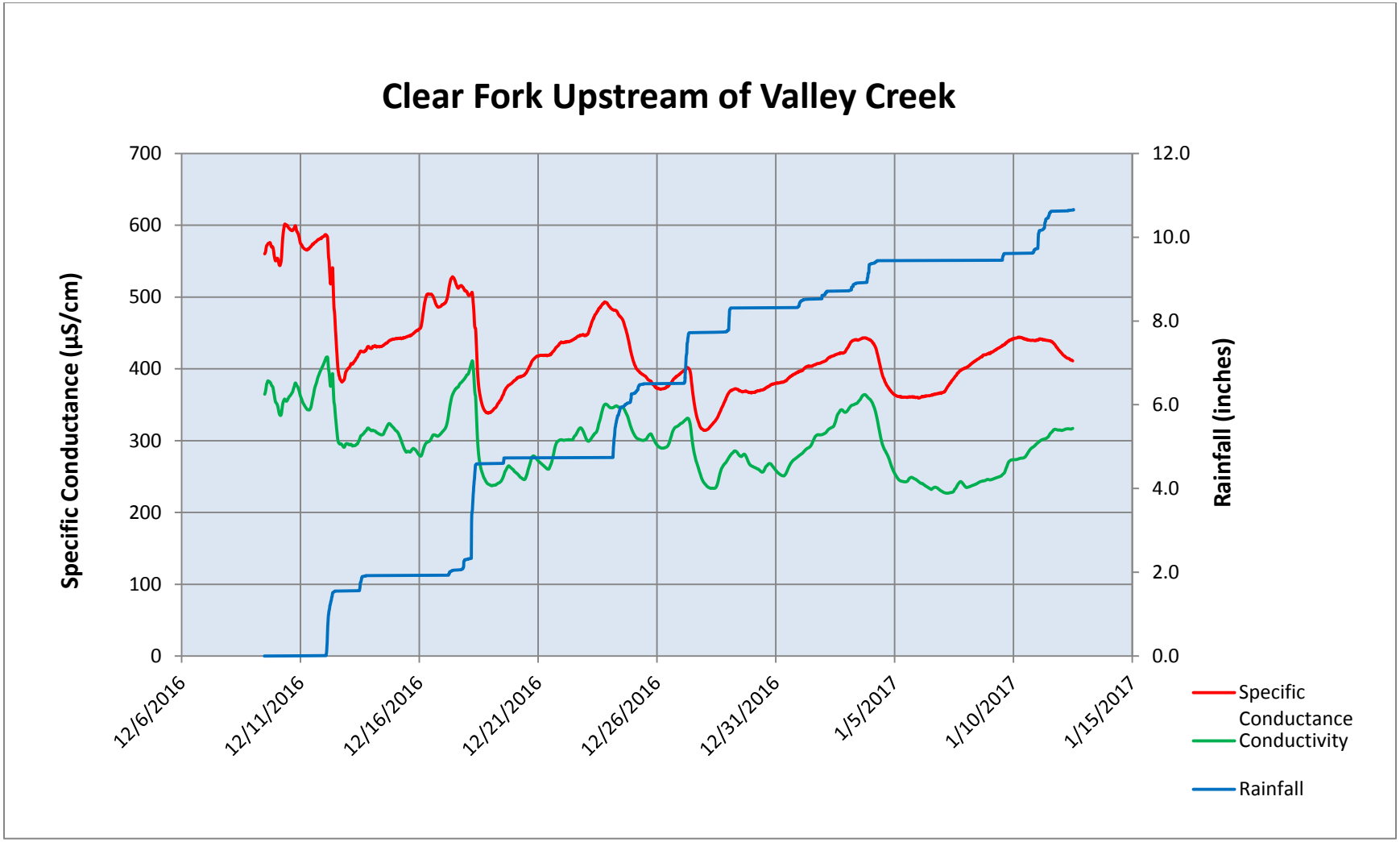

Figure C-6. Conductivity and specific conductance and rainfall in Clear Fork just upstream of its confluence with Valley Creek May 2015 through January 2017, as measured by continuously recording data loggers (OSMRE unpublished data 2017).