United States Department of the Interior
Fish and Wildlife Service
105 West Park Drive, Suite D
Athens, Georgia 30606
706-613-9493 Fax: 706-613-6059

West Georgia Sub Office
P.O. Box 52560
Ft. Benning, Georgia 31995-2560
706-544-6428 Fax: 706-544-6419

Coastal Sub Office
4980 Wildlife Drive
Townsend, Georgia 31331
912-832-8739 Fax: 912-832-8744

June 27, 2014

Mr. John Brent
Chief, Environmental Management Division
Headquarters, United States Army Infantry Center
Ft. Benning, GA 31905-5000

FWS Log No.: 2014-F-0471

Dear Mr. Brent;

This document is the U.S. Fish and Wildlife Service's (Service) biological opinion (BO) of the impacts of operating Fort Benning's Farnsworth Range on red-cockaded woodpeckers (RCW) (*Picoides borealis*). We based our opinion on information you provided in the November 22, 2013, biological assessment (BA). This BO was prepared in accordance with section 7 of the Endangered Species Act (Act) of 1973, as amended (16 U.S.C. 1531 et seq.).

A complete administrative record of the consultation is on file in our office. The proposed action is not likely to adversely affect other listed species under the Service's purview, and no further action for these species is required under Section 7 of the Act. This document does not address requirements of other environmental statutes, such as the National Environmental Policy Act.

CONSULTATION HISTORY

December 2010: Fort Benning (Ft.) informed the Service of bullet strikes in RCW clusters located in Training Area Compartments A-02 and A-03. After a field inspection with the Fort Benning staff, the Service advised Installation biologists to continue monitoring the clusters and to consider entering into formal consultation. Installation biologists were to determine if the affects from ongoing range use was anticipated to lead to adverse effects on RCWs.

April 2011: Installation staff determined that the bullet intrusions in RCW clusters A03-A and A03-C were from soldiers firing from the prone position at the zero firing distance (10-25 meters). The solution or minimization strategy was to move the target frames as close as possible to the protective berm. Later surveys concluded that bullet intrusions into the two clusters were significantly reduced (Ft. Benning 2012a).

September 2011: The Installation initiated minimization efforts, which included: (1) removing the concrete pad that was on top of the Farnsworth Range firing line, with (2) raising the prone firing positions at the 100-550 meter firing position, that significantly helped reduce rounds from breaching the toe berm and entering RCW clusters. Additional minimization efforts in 2011 included correction of a drainage problem, which prior to repairing, discouraged soldiers from setting up target frames next to the berm when conducting zero fire training.

May and December 2011: Installation foresters collected forest inventory data for eight affected clusters. During the 2011 nesting season a potential "bud" occurred in cluster A03-A. Once the foraging habitat was repartitioned and habitat reallocated for the surrounding RCW territories, the newly formed A03-C clusters territory was deficient. The Installation staff was concerned that the loss of one 10" stem in the A03-C territory would then leave the cluster adversely impacted by military range use resulting in an ESA violation (i.e., a non-permitted incidental take).

December 2012 and January 2013: Extensive berm work was completed at Farnsworth Range to raise the toe berm and raise the firing line berms. Many nearby ranges also had erosion control measures applied to their toe berms during this period.

October 2013: To improve surrounding downrange RCW habitat, 400 acres and 600 acres of habitat in Training Compartments A30 and A20, respectively, were treated with herbicide. Roughly 251 treated acres were within clusters affected by range training.

November 2013: Formal consultation was initiated to evaluate the impacts of training on clusters downrange of Farnsworth Range (Training Compartments A02, A03 and A30).

BIOLOGICAL OPINION

DESCRIPTION OF THE PROPOSED ACTION

The Army proposes to reinitiate operations at Farnsworth Range, which includes; zero qualification and familiarization, close quarters marksmanship (CQM), known distance marksmanship, stress shooting, and sniper training. Ranger Anti-Armor Weapon System training can also be used at Farnsworth. Weapons supported by the Range include rifles (M16 series), sniper rifles (M24, SR25), machine guns (M240 B/C/G, M249, MK48), carbines (M4), semi-automatic rifles (M82A1 Barrett), pistols, revolvers, submachine guns (MP5) and grenade launchers (Fort Benning 2013b). The majority of the training conducted is CQM, advanced marksmanship training and zero firing from 0-25 meters (m). Under the current Record of Environmental Consideration, the 75th Ranger Regiment is approved to use 7.62 millimeter (mm), 5.56 mm and 9 mm live and blank ammunition (Ft. Benning 2012a).

Conservation measures implemented to minimize rounds from entering downrange RCW clusters (A02-A, A03-A, -C, and A30-A, -D, -E, -F and -G) included elevating and reshaping the firing line berms, elevating and reshaping of toe berms, moving target frames as close as possible to the

protective berms, and correcting drainage problems which will enables soldiers to set up targets closer to the berms.

For the purpose of consultation under Section 7 of the Endangered Species Act, the Action Area (AA) is defined at 50 CFR 402 to mean "all areas affected directly or indirectly by the Federal action, and not merely the immediate area involved in the action." For projects impacting RCWs, the AA must include the RCW "neighborhood," which is defined by a buffer extending beyond the directly impacted area(s) equal to the average dispersal distance of RCWs within that population or subpopulation (USFWS 2005). Dispersal is defined as the movement of individuals from their natal cluster to their first breeding location, or between consecutive breeding locations (USFWS 2003). For this BO, dispersal distance is defined as the average distance Fort Benning RCWs have traveled from their natal cluster to find an available territory or between consecutive breeding locations. This includes birds that were part of a breeding pair, helpers to an unrelated breeding pair and solitary birds defending a vacant territory. In preparation for the 2003 Maneuver Center of Excellence (MCOE) BO, Fort Benning RCW dispersal data collected over 11 years was analyzed by the Installation and revealed an average dispersal distance of 2.57 miles (mi.) (USACE 2008). In order to determine the Action Area for the proposed action, a 2.57-mi. buffer was applied to a polygon encompassing the ranges being assessed (from Fiske Range east to Pool Range) and the 8 clusters being analyzed (Clusters A02-A, A03-A and -C, and A30-A, A30-D, -E, -F and –G). The Action Area for this BO encompasses 23,731 acres (see Figure 3-1in the BA for the full depiction of the AA).

STATUS OF THE SPECIES

The U. S. Department of the Interior identified the RCW as a rare and endangered species in 1968. In 1970, the RCW was officially listed as endangered (Federal Register 35:16047), and, with passage of the Act in 1973, the RCW received the protection afforded species listed under the Act. No critical habitat has been designated for RCWs.

The RCW is a small woodpecker about 8 inches in length, with a wingspan of about 14 inches. An adult weighs about 1.7 ounces. The bird is black and white, with a ladder back, and is distinguished from other woodpeckers by its black capped head and nape, and surrounding large white cheek patches. Adult males possess a tiny red streak or tuft of feathers, the cockade, in the black cap near each ear and white cheek patch. The small cockade usually is covered by the black crown, except when protruded during excitement, and is not readily visible except upon close examination. Adult males and females are not readily distinguishable in the field. Juvenile males have a red crown patch until the first molt, which can be distinguished from the black crown of juvenile females.

Life History

The RCW is a territorial, non-migratory, cooperative breeding species (Lennartz et al. 1987; Walters et al. 1988). It is the only North American woodpecker that exclusively excavates its cavities for roosting and nesting in living pines. Usually, the trees chosen for cavity excavation are infected with a heartwood decaying fungus (*Phellinus pini*) (Jackson 1977; Conner and Locke 1982). The heartwood associated with this fungus is not generally present in longleaf (*Pinus*

palustris) and loblolly pine (*Pinus taeda*) until a tree is 90-100 and 75-90 years of age, respectively (Clark 1992a; Clark 1992b). Large trees also are required because the cavity is constructed and placed entirely within heartwood where pine resin will not flow. Each group member has its own cavity, although there may be multiple cavities in a cavity tree. RCWs chip bark and maintain resin wells on the bole around the cavity where the fresh flow of sticky resin is a deterrent against predatory snakes (Rudolph et al. 1990) and indicates an active cavity tree. The aggregate of cavity trees, surrounded by a 200-foot, forested buffer, is called a cluster (Walters 1990). Cavities within a cluster may be complete or under construction (starts) and either active, inactive or abandoned. Clusters with one or more active cavity trees are considered active RCW clusters.

Most RCW populations occur where longleaf pine historically dominated the forest community. Populations in other vegetation types occur in the western, northern interior and southernmost regions flanking the longleaf pine ecosystem. RCWs in the West Gulf Coastal Plain occupy loblolly pine forests in parts of southern Arkansas, east Texas, and Louisiana on flatwood terraces and more dissected upper terraces where loblolly pine was dominant or with shortleaf pine as a natural community type (e.g., Moore and Foti 2005; Moore and Foti 2008). Shortleaf pine-dominated communities currently with RCWs are in portions of the coastal plain in east Texas, the Ouachita Mountains of Arkansas and eastern Oklahoma, the Piedmont and Cumberland Plateau of Alabama, and the Georgia Piedmont. In south Florida, RCWs persist in hydric pine flatwoods dominated by South Florida slash pine (*Pinus elliottii* var. *densa*). In northeastern North Carolina and southeastern Virginia, small populations remain associated with pond pine (*Pinus serotina*) communities and pocosins. However, the fundamental ecology of RCWs remains the same --populations occupy fire-maintained, open pine forests, with pine of sufficient age and size for cavities and foraging.

RCW live in social units called groups. This cooperative unit usually consists of a monogamous breeding pair, offspring of the current year, and 0-4 adult helpers (Walters 1990). Helpers typically are male offspring from previous breeding seasons that assist the breeding pair by incubating eggs, feeding young, excavating cavities, and defending the territory (Ligon 1970, Lennartz and Harlow 1979, Lennartz et al. 1987, Walters et al. 1988). Some large populations infrequently have female helpers (Walters 1990; DeLotelle and Epting 1992; Bowman et al. 1998). Some clusters are only occupied by a single adult male (called single bird groups).

The RCW is territorial and each group defends its home range from adjacent groups (Hooper et al. 1982; Ligon 1970). The defended territory includes cavity trees and foraging habitat. RCWs feed mostly on arthropods, particularly ants and wood roaches, foraging predominately on and under the bark of larger and older living pines. Males tend to forage in crowns and branches, while females commonly forage on the trunk. Dead and dying pines are important temporary sources of prey, and hardwoods are used occasionally. RCWs tend to forage with other group members within 0.5 miles of their cluster.

RCWs have large home ranges relative to body size (86-556 acres; Conner et al. 2001; U.S. Fish and Wildlife Service 2003). Home range size is variable within and between populations, but tends to reflect foraging habitat quantity and quality, boundaries of adjacent RCW territories, and possibly cavity tree resource availability (Conner et al. 2001; U.S. Fish and Wildlife Service 2003). The average RCW home range size tends to be greater at xeric and wet communities or sites in Florida than more productive pine sites in Georgia and South Carolina (Nesbitt et al. 1983; DeLotelle et al.

1987; Epting et al. 1995; Hardesty et al. 1997a). Home range size has been related to the area of suitable habitat within 2 km of the cluster, pine basal area, pine density, pine density greater than 25 cm dbh, RCW group density, hardwood midstory, and other factors (Hooper et al. 1982; DeLotelle et al. 1987; Bowman et al. 1998; Hardesty et al. 1997a; Walters et al. 2000, 2002b). Variation in home range size reflects a response to habitat quality, where a larger home range is generally required in low quality habitat.

About 90 percent (%) of the breeding pairs in a population nest each year. The nesting season occurs from April to July. Females usually lay 3 or 4 eggs in the cavity of the adult male. The short incubation period lasts approximately 10-12 days, and eggs hatch asynchronously. Nestlings fledge after 24-29 days, although all nestlings rarely survive to fledglings. Partial brood loss of nestlings is common in RCWs, although the number of hatchlings successfully fledged tends to increase with group size. Older and more experienced breeders have greater fledgling success, which maximizes at about 7 years of age, then declines sharply at 9 years of age or older. About 20% of nests will fail completely. Groups with helpers experience whole brood loss less frequently than breeding groups without helpers. Renesting rates are geographically and annually variable. In good years, up to 30% of breeding groups will renest. Productivity of second nesting is lower.

RCWs in southern and coastal RCW populations tend to have lower productivity and greater survival rates than more northern and inland populations (Lennartz and Heckel 1987; DeLotelle and Epting 1992). These differences may be due to lower winter temperatures and survival with greater reproductive effort in northern populations, and life history evolution in more favorable southern climates where greater survival and lower annual reproduction are responses to increased competition (Conner et al. 2001).

Subadult/juvenile females from the current year breeding season normally disperse before the next breeding season, or are driven from the group's territory by the group (see Walters et al. 1988, for additional sociobiological/cooperative breeding information). Juvenile females remain at their natal territory to assume the breeding vacancy of the female only when the breeding male dies and the breeding female disperses or dies. A breeding female will disperse, creating a breeding vacancy, when her male offspring inherit the male breeding position (incest avoidance). Dispersing juvenile females move to nearby RCW territories in search of a breeding vacancy. These females either become breeders in a territory, or floaters among more than one territory where they are not associated with a particular group.

Juvenile males remain in their natal territory or disperse. Those that remain become helpers or, if the breeding male dies before the next breeding season, breeders. Dispersing juvenile males search for positions as breeders in nearby territories where they either become breeders, helpers, or floaters. Most adult male helpers remain on their natal territory as helpers; about 15% will inherit the territory as a breeding male in any given year. Some adult helpers disperse to other territories becoming breeders, solitary males, helpers, or floaters. However, breeding males are highly territorial and most will remain even without a breeding female. In contrast, about 10% of breeding females will break the pair-bond between breeding seasons and disperse to another territory as a breeder with a different male (Walters et al. 1988; Daniels and Walters 2000b).

Habitat Quality

High quality RCW foraging habitat consists of an open fire-maintained pine forest, with no or a sparse midstory of hardwood or pine, low densities of small pine (less than 10 inches dbh), moderate densities of medium-sized (10 – 14 inches diameter at breast height [dbh]) and large (greater than14 inches dbh) pine, at least low densities of old growth pine, and a well-developed herbaceous plant ground cover (James et al. 2001; Walters et al. 2002b). RCWs selectively forage on larger and older pines more frequently than their availability relative to younger and smaller trees (Zwicker and Walters 1999; Walters et al. 2002b). Overall, RCWs preferentially use pine 12-20 inches dbh, prefer trees greater than 20 inches dbh, use trees less than 20 inches dbh depending on the availability of larger trees, and avoid trees less than 12 inches dbh when larger trees are available (Walters et al. 2000). RCW group size, productivity (fledglings produced), or both are negatively related to an increasing density of small and intermediate-size pine, as well as the density and height of the hardwood midstory (Conner and Rudolph 1991; Rudolph and Conner 1994; Hardesty et al. 1997a; Engstrom and Sanders 1997; James et al. 1997, 2001; Walters et al. 2002b).

Reasons for Listing

The precipitous decline of RCWs was caused by an almost complete loss of habitat. Before European settlement, the number of RCW groups inhabiting longleaf pine forests and all southern pine forests has been estimated at 920,000 (Costa 2001) and 1.5 million (Conner et al., 2001), respectively. Fire-maintained old growth pine savannahs and woodlands that once dominated the Southeast (92 million acres pre-European settlement; Frost 1993) no longer exist except in a few small patches (less than 3.0 million acres today; Frost 1993). Longleaf pine ecosystems are now among the most endangered systems on earth (Simberloff 1993; Ware et al. 1993).

Loss of the original pine ecosystems was primarily due to intense logging for lumber and agriculture. Logging was especially intense at the turn of the century (Frost 1993). Two additional factors in loss of pine systems in the 1800's and earlier were exploitation for pine resins and grazing of free-ranging hogs (Wahlenburg 1946, Frost 1993). Later in the 1900's, fire suppression and detrimental silvicultural practices had major impacts on the status of RCWs (Frost 1993, Ware et al. 1993, Ligon et al. 1986, 1991, Landers et al. 1995). Additionally, longleaf pine suffered a widespread failure to reproduce following initial cutting, at first because of hogs and later because of fire suppression (Wahlenburg 1946, Ware et al. 1993).

Current Threats and Population Trends

Most RCW populations on public lands, as well as those private land populations in partnerships with the Service, have been stabilized, and many are now increasing. This steady increase can be attributed to various factors, including aggressive prescribed burning programs, artificial cavity (Copeyon 1990; Allen 1991) and translocation (Costa and DeLotelle 2006). In 1993/1994, the range-wide population was estimated at 4,694 active clusters (Costa and Walker 1995). In 2001and 2003, the range-wide population estimates were 5,627 (U.S. Fish and Wildlife Service 2003; Costa 2004) and 5,800 active clusters, respectively (Costa and Jordan 2003); and in 2006 it was 6,105 active clusters (U.S. Fish and Wildlife Service unpublished data). Thirty seven of the 39 recovery

populations (95%) were either stable or increasing, based on number of active clusters, during the most recent 5-year growth period (2008-2012) for which data is available. Only two populations had a declining trend: Hal Scott and Withlacoochee Citrus Tract Essential Support.

Most of the 39 recovery populations are composed of one of more adjacent properties. The current recovery plan identifies 63 properties involved in recovery -- 16 (25%) had a net 5-year declining trend. Large recovery populations remain rare. Of the 63 recovery properties, only 6 (9.5%) exceed 250 active clusters. Most property populations are small, much more vulnerable populations of 50 or fewer active clusters.

Primary threats to species viability all are related to lack of suitable habitat in a fire-maintained ecosystem. On public and private lands, the quantity and quality of RCW habitat are impacted by past and current fire suppression and detrimental silvicultural practices (Ligon et al. 1986, 1991, Baker 1995, Cely and Ferral 1995, Masters et al. 1995, Conner et al. 2001). Serious threats stemming from this lack of suitable habitat include: (1) insufficient numbers of cavities and continuing net loss of cavity trees (Costa and Escano 1989, James 1995, Hardesty et al. 1995), (2) habitat fragmentation and its effects on genetic variation, dispersal and demography (Conner and Rudolph 1991), (3) lack of good quality foraging habitat (Walters et al. 2000, James et al. 2001), and (4) fundamental risks of extinction inherent to critically small populations from random demographic, environmental, genetic, and catastrophic events (Shaffer 1987).

The natural growing season fire regime that maintained quality foraging habitat has been lost due to fire suppression and landscape alterations that have altered the availability of lightning-flammable fine plant litter fuels. In the absence of prescribed fire, fire intolerant hardwoods survive and grow to midstory or higher levels in the forest canopy. RCWs, being sensitive to midstory hardwood encroachment, will abandon their cavities and clusters due to hardwood encroachment (Conner and O'Halloran 1987; Costa and Escano 1989).

Genetics

Genetically, most variation is partitioned (greater than 86%) among individuals within populations, rather than among populations (14%), according to allozyme (Stangel et al. 1992; Stangel and Dixon 1995) and random amplified polymorphic data (Haig and Rhymer 1994, Haig et al. 1996). The genetic structure of populations is significantly, although weakly, spatially heterogeneous (Stangel et al. 1992; Haig and Rhymer 1994), but somewhat more structured than in most non-endangered birds (Haig and Rhymer 1994). Genetic distance (dissimilarity) tends to increase as the geographic distance between populations increases (Stangel et al. 1992; Haig et al. 1996). Mean heterozygosity among populations is relatively high and comparable to other species, although allelic diversity in some small populations is reduced (Stangel et al. 1992). Unique alleles are not known to distinguish populations. These genetic characteristics are generally expected in a historically widely-distributed species that only relatively recently has become reduced in fragmented populations (U.S. Fish and Wildlife Service 2003). However, inbreeding depression has been detected within a relatively large population, adversely reducing rates of hatching and fledgling survival (Daniels and Walters 2000a).

Population Dynamics

Like other species, RCW population size during a given year is affected by birth, death, immigration, and emigration rates. However, RCW population dynamics also are significantly affected by the species' cooperative breeding system and behavior of territorial RCW groups with helpers. The spatial distribution and aggregation of groups affects the likelihood that breeders in a group will be replaced upon their death or dispersal by other RCWs. All of these factors regulate population size, stability, and viability as mediated by the effects of habitat, genetics, demographic and environmental stochasticity, and environmental catastrophes. RCW population size is commonly measured as the number of potential breeding groups (PBG), which is defined as an adult male and female, with or without helpers, occupying the same cluster. A total count of individuals, including non-breeding helper, single male groups, and floater adults, would not account for group and territory dynamics or the buffering effect of helpers as a replacement pool for breeders. In the absence of data for the number of PBGs, the number of active clusters is an index estimate of population size (number of groups). An active cluster is one where fresh resin from RCW activity at a cavity occurs on one or more trees. An active cluster may be occupied by a PBG or a single-male group. The number of single-male groups also is important because a large proportion of single-bird groups indicate a declining population.

Demographically, a RCW population is strongly affected by the dispersal distances of males and females from their natal group or birds that search for and compete for breeding vacancies at other groups. At North Carolina study sites (North Carolina Sandhills and Camp Lejeune), dispersing juvenile and helper males rarely assumed breeding vacancies at clusters located more than 2 miles from their natal or group site (Daniels 1997; Walters et al. 1988). Juvenile females from the same study areas were capable of longer forays, becoming breeders at clusters up to 3.7 miles away (Walters et al. 2008). In western Florida (Eglin Air Force Base), adults dispersed an average 1.1 miles, juvenile females 2.0 miles, and juvenile males 5.0 miles (Hardesty et al. 1997b).
The number of RCW group territories increases by two primary processes (1) pioneering, the colonization of a previously unoccupied territory, and budding, the creation of a new group by subdividing an existing group territory and its cavity trees, usually by a group helper or an immigrant male (Conner et al. 2001). . Pioneering and budding rarely occur under current conditions, with rates, respectively, of only 0.06-1.5% per year (U.S. Fish and Wildlife Service 2003) and 0.6-2.1%.

Population Stability

RCW population viability depends on a sufficient number of stable groups to avoid adverse effects of inbreeding, genetic drift, and impacts from stochastic genetic demographic, environmental, and catastrophic events. Small populations are particularly sensitive to exacerbating effects of stochastic factors (Soule 1987, Clark and Seebeck 1990), which can drive local extirpation or extinction (Gilpin and Soule 1986).

Spatially Explicit Population Models(SEPM) simulate the movement and fate of each individual in a population depending on its status and currently are the most accurate models simulating RCW population dynamics and viability (e.g., Letcher et al. 1998; Daniels et al. 2000; Walters et al. 2002a). RCW SEPMs have revealed significant effects of spatial structure and distribution of

groups on viability, based on the relatively short dispersal distances of male juveniles and helpers (2 miles) and females (3.7 miles) to inherit breeding vacancies in nearby territories (Walters et al. 1988, Daniels 1997, Walters et al. 2008) -- groups located at greater distances and at lower densities are much less likely to sustain breeding pairs, becoming demographically isolated and more vulnerable to local extirpation.

Letcher et al. (1998) determined, using SEPM analysis with the added effects of demographic stochasticity, that small populations with 49 highly aggregated groups are stable over 100 years, and smaller populations of 25 highly aggregated groups are highly persistent for about 60 years. Highly aggregated groups share common territorial boundaries. Even smaller, highly aggregated populations of 20 and 10 groups have good persistence for 20 years, although population growth rates are less than 1.0 and slowly declining (Crowder et al. 1998). Highly aggregated populations of 49 groups are more stable than minimally aggregated populations of 169 or 250 groups, and non-highly aggregated populations with less than 100 groups decline and are not viable. The strong persistence of highly aggregated RCW populations reflects the demographic effect of the helper birds. Variation in breeder mortality is dampened by helpers that replace breeders. Fluctuating periods of greater breeder mortality tends to reduce the size of the helper class instead of reducing the number of breeding groups (Walters et al. 2002a). Regardless of the aggregation or clumping of the modeled populations in their study (Letcher et al. 1998), populations of 500 groups were viable, as were moderately aggregated groups of 250.

Walters et al. (2002a) concluded that RCW population persistence and viability in response to demographic and environmental stochasticity was similar to that of comparable populations affected only by demographic stochasticity. The added effects of environmental stochasticity were relatively small compared to viability analysis of other species. Once again, the non-breeding class of helpers in the RCW cooperative breeding system had a buffering effect on breeder mortality and loss of breeding groups.

Inbreeding

Daniels et al. (2000) used a RCW SEPM to assess potential inbreeding effects with demographic and environmental stochasticity to viability in small populations of 25, 49 and 100 groups with a moderate level of group aggregation. In earlier studies, Daniels and Walters (2000a) documented actual effects of inbreeding depression in RCWs, including reduced hatching success and fledgling survival. Daniels et al. (2000) found that inbreeding depression is a serious viability threat to small, isolated, and declining RCW populations. RCW populations of 25 and 49 groups declined, as in other RCW SEPMs. The stable population of 100 groups was only marginally persistent over their 50-year simulation period, and may not have been stable if simulated for a 100-year period. The mean percentage of closely related breeding pair increased for all populations. Closely related breeding pairs were most prevalent in populations of 25 and 49 groups, which were at risk of extremely high inbreeding. However, two or more immigrants to these populations per year could stabilize a declining trend and reduce significantly the number of closely related breeding pairs.

Catastrophes

Hurricanes, tornadoes, and southern pine beetles are the primary catastrophic events affecting RCW population stability. These events damage or destroy habitat, reducing the number of breeding groups by the loss of cavity trees and foraging habitat. Hurricanes are the greatest catastrophic threat, as indicated by their frequency, widespread distribution, intensity, and effects (Hooper and McAdie 1995). Hurricane Hugo, a category IV storm, destroyed about 87% of RCW cavity trees in the Francis Marion National Forest, reducing the estimated pre-storm population of 477 active clusters to 277 clusters that retained at least one remaining cavity tree (Hooper et al. 1990; Watson et al. 1995). Coastal populations, particularly small populations, are highly vulnerable while the most inland populations are at least risk. RCW populations in the Croatan National Forest (SC), Francis Marion National Forest (SC), Apalachicola National Forest (FL), DeSoto National Forest (MS), Eglin Air Force Base (FL), and Conecuh National Forest (AL) and nearby regions are the most vulnerable based on hurricane return periods and intensity (Hooper and McAdie 1995).

Southern pine beetle epidemics adversely affect loblolly pine much more than longleaf, which have greater resin production and resistance to attack. The loss of planted loblolly pine, which was planted in much of the historic longleaf pine range, as well as loblolly in its natural habitat, can be locally significant. More than 50 RCW groups lost all loblolly cavity trees in the Sam Houston National Forest in the 1980s, where more than 300 cavity trees were killed by beetles between 1982 and 1984 (Conner et al. 2001). Loss of cavity trees and foraging habitat in small populations can be locally severe, leading to a reduction in breeding groups and potentially threatening local extirpation in small populations (Mills et al. 2004).

Recovery Criteria

Recovery criteria in the Recovery Plan were formulated on the basis of 11 recovery units delineated according to ecoregions. Populations required for recovery are distributed among recovery units to ensure representation of broad geographic, ecologic, and genetic variation in the species. The wide geographic distribution reduces the threat of catastrophic habitat destruction and population loss by hurricanes. The distribution of populations and recovery units also will facilitate periodic RCW immigration and emigration among populations, which will be required to offset or reduce the loss of potential adaptive genetic variation within populations by genetic drift.

Downlisting to threatened status will be considered when each of the following criteria is met.

1. **Criterion 1**. There is one stable or increasing population of 350 PBGs (400 to 500 active clusters) in the Central Florida Panhandle (i.e., “The Central Florida Panhandle primary core population”). This criterion has been met. The Apalachicola Ranger District, one of the five properties comprising the Central Florida Panhandle Primary Core population, harbors more than 350 PBGs.

2. **Criterion 2**. There is at least one stable or increasing population containing at least 250 PBGs (275 to 350 active clusters) in each of the following recovery units Sandhills, Mid-Atlantic Coastal Plain, South Atlantic Coastal Plain, East-Gulf Coastal Plain, West Gulf Coastal Plain, Upper West Gulf Coastal Plain, and Upper East Gulf Coastal Plain.

Three (Sandhills, Mid-Atlantic Coastal Plain, and South Atlantic Coastal Plain) of the six recovery units are required to have a population with 250 PBGs are present.

3. **Criterion 3**. Excluding the populations that meet previous criteria, there is at least one stable or increasing population containing at least 100 PBGs (110 to 140 active clusters) in each of the following recovery units Mid-Atlantic Coastal Plain, Sandhills, South Atlantic Coastal Plain, and East Gulf Coastal Plain. This criterion has been met. Each of the listed recovery units contains at least one population (different from the populations listed under Criterion 2 above) that harbors at least 100 PBGs.

4. **Criterion 4**. There is at least one stable or increasing population containing at least 70 PBGs (75 to 100 active clusters) in each of four recovery units: Cumberlands/Ridge and Valley, Ouachita Mountains, Piedmont, and Sandhills. In addition, the Northeast North Carolina/Southeast Virginia Essential Support Population is stable or increasing and contains at least 70 PBGs (75 to 100 active clusters). Only the Sandhills recovery unit contains a population harboring at least 70 PBGs (that would not be needed to satisfy either Criterion 2 or 3, which also require Sandhills populations of certain sizes).

5. **Criterion 5**. There are at least four populations each containing at least 40 PBGs (45 to 60 active clusters) on state and/or federal lands in the South/Central Florida Recovery Unit. This criteria has been met. Five populations exceed 45 active clusters: Big Cypress, Goethe, Ocala, Three Lakes and Withlacoochee Citrus Tract.

6. **Criterion 6**. There are habitat management plans in place for each of the above populations that identify management actions to increase the populations to recovery levels, with special emphasis on frequent prescribed burning during the growing season. Although Criterion 6 is referring to the need for populations to have such plans when they achieve their size goals, the majority of the populations required for delisting already have management plans that address habitat management (e.g., prescribed burning) and population monitoring. These plans are generally updated at 5-year intervals. The plans take the form of Integrated Natural Resource Management Plans (military installations), Land and Resource Management Plans (national forests), Comprehensive Conservation Plans (national wildlife refuges), and property-specific state wildlife management area and state forest plans.

Delisting will be considered when each of the following criteria is met.

1. **Criterion 1.** There are 10 populations of RCWs that each contain at least 350 PBGs (400 to 500 active clusters), and 1 population that contains at least 1000 PBGs (1100 to 1400 active clusters), from among 13 designated primary core populations, and each of these 11 populations is not dependent on continuing installation of artificial cavities to remain at or above this population size. One population (North Carolina Sandhills) of the 10 primary core populations required has achieved 350 PBGs.

2. **Criterion 2.** There are nine populations of red-cockaded woodpeckers that each contain at least 250 PBGs (275 to 350 active clusters), from among 10 designated secondary core populations and each of these nine populations is not dependent on continuing installation of

artificial cavities to remain at or above this population size. None of the 10 secondary core populations harbors 250 PBGs.

3. **Criterion 3.** There are at least 250 PBGs (275 to 350 active clusters) distributed among designated essential support populations in the South/Central Florida Recovery Unit, and six of these populations (including at least two of the following: Avon Park, Big Cypress, and Ocala exhibit a minimum population size of 40 PBGs that is independent of continuing artificial cavity installation (i.e., "population trends"). This criterion has not been achieved.

4. **Criterion 4**. There is one stable or increasing population containing at least 100 PBGs (110 to 140 active clusters) in northeastern North Carolina and southeastern Virginia, the Cumberlands/Ridge and Valley recovery unit (Talladega/Shoal Creek), and the Sandhills recovery unit (North Carolina Sandhills West), and these populations are not dependent on continuing artificial cavity installation to remain at or above this population size. One (North Carolina Sandhills West) of the three populations required to exceed 100 PBGs is present.

5. **Criterion 5**. For each of the populations meeting the above size criteria, responsible management agencies shall provide (1) a habitat management plan that is adequate to sustain the population and emphasizes frequent prescribed burning and (2) a plan for continued population monitoring. Although criterion 5 is referring to the need for populations to have such plans when they achieve their size goals, the majority of the populations required for delisting already have management plans that address habitat management (e.g., prescribed burning) and population monitoring. These plans are generally updated at 5-year intervals. The plans take the form of Integrated Natural Resource Management Plans (military installations), Land and Resource Management Plans (national forests), Comprehensive Conservation Plans (national wildlife refuges), and property-specific state wildlife management area and state forest plans.

ENVIRONMENTAL BASELINE

The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the AA, the anticipated impacts of all proposed Federal projects in the AA that have already undergone formal Section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50CFR § 402.02). The impacts associated with the past and present Federal, State or private actions that define the environmental baseline within the proposed AA includes: the development and use of 13 small arms ranges (Fiske, Farnsworth , Roosevelt, English, Simpson, Shelton, McAndrews, Easley, Hibbs, Phillips, Parks, Hook and Pool) including the development and use of interrelated infrastructure (roads, buildings, etc), infrastructure development used to support the operations of tracked/ wheeled vehicles, infrastructure development and use that supports a portion of the Installation's cantonment area (barracks, hospital, offices, roads, etc), and infrastructure development and use that supports Lawson Army Air Field (See Figure 3-1 from the BA which depicts the AA and the baseline impacts).

The portion of the Action Area outside of the Installation boundary, but within the RCW neighborhood and Action Area, lies within Muscogee County, Georgia and Russell County, Alabama. Land uses in this area include pine plantations, agricultural fields, the Chattahoochee River and associated floodplain, the National Infantry Museum and a portion of the Chattahoochee Riverwalk. No Federally-listed species are known to occur within the off-Post portion of the Action Area.

Status of Red-cockaded Woodpecker within the Action Area

Based on the November 32, 2013, BA, there are 64 RCW clusters currently being managed within the AA; 59 are active and 5 are inactive.

EFFECTS OF THE ACTION

This section includes an analysis of the direct and indirect effects of the proposed action, including any interrelated or interdependent activities, on the listed species exposed to those effects. The analysis assumes that all conservation measures described as part of the proposed action will be implemented as described in the Dixie Road Small Arms Ranges BA.

Factors Considered for Red-cockaded Woodpeckers

Loss of RCW cavity trees

One of the concerns with bullet strikes within a RCW cluster is damage to, or death of, cavity trees. In addition to the effect of major damage such as shearing off limbs or leaders, the sap flow from bullet wounds can make trees more susceptible to wildfire damage (Delaney, Guertin et al. 2011). Repeated minor injury to cavity trees could also make them more susceptible to disease and insect infestation.

In a study conducted by Delaney, Guertin et al. (2011) in the Malone Small Arms Complex on Fort Benning, bullet damage found at plots at 3,543 feet (ft.) and 3,116 ft. downrange of Malone 5 (from the range toe) was primarily low velocity bullet strikes that did not cut into the trees' cambium layer. Broken limbs were observed in 6% and 29% of the trees sampled at 3,543 and 3,116 ft., respectively, and 65% and 75% of the trees had bullet wounds of some kind. At 2,690 feet, a mixture of low-velocity strikes and higher-velocity strikes, that had scarred trees were observed. Thirty-two percent of trees sampled had broken limbs, and 80% had been wounded.

Loss of RCW foraging habitat

The potential impacts to cavity trees described above also apply to foraging habitat within RCW partitions (pine stands over 30 years old). When ranges are constructed, trees downrange of firing positions and outside of range footprints are typically left unaltered in order to act as a buffer for the surrounding area. Over time, however, these trees incur some degree of mortality from fired munitions hitting or shearing them, either directly or from ricochet. As trees die, there is less of a buffer, potentially allowing ordnance to travel further and thereby expanding the areas of impact.

Of the areas assessed in this BO, the only area where considerable loss of foraging habitat was a concern was the area between the toe of Farnsworth Range and Cluster A03-A, where it is clear that bullets were regularly breaching the berm and damaging vegetation downrange. Downrange strikes to trees resulting from sustained, unobstructed, direct munitions impacts and are believed to have been alleviated by the extensive berm work completed at Farnsworth in February 2013; therefore, no additional measurable loss of foraging habitat is expected. Since the longleaf pine plantation directly downrange of Farnsworth Range should no longer be subject to the shearing effects of weapon fire, foraging habitat is expected to increase over time. Although pines could still occasionally be lost due to bullet impacts, this loss of foraging substrate is not expected to be severe or even quantifiable; therefore, no loss of foraging habitat was analyzed in this BO.

While trees being impacted by bullets may not provide stable RCW nesting habitat due to possible harassment impacts and the risk of tree death from injuries, fire or pine beetles, these trees can continue to be valuable foraging habitat. For this reason, stands containing bullets but meeting foraging standards were not discounted in the foraging habitat analysis. Also, the October 2013 herbicide treatment within Compartments A20 (approximately 600 total acres) and A30 (approximately 400 total acres) should have a beneficial effect on the RCW by reducing the density of the hardwood midstory within analyzed RCW foraging partitions and others within the Action Area.

Noise and harassment

Bullet strikes within the cluster or bullets traveling through the cluster could also have a detrimental "harass" effect on local RCW groups, although it is probable that the birds have adapted to this disturbance. Over the past 30 years, several research projects have assessed the potential effects of military noise, primarily from large-caliber ranges and artillery simulators, on certain elements of RCW fitness (Jackson and Parris 1995, Doresky et al. 2001, Pater et al. 1999, Delaney et al. 2002, Hayden et al. 2002, Walters 1992). Research suggests that military training (e.g., heavy maneuver training or light infantry) and/ or civilian activity in the vicinity of RCW clusters may affect RCW behavior by causing more frequent flushing during incubation and/or less frequent feeding of nestlings, which can cause a reduction in nest success or the number of young fledged. In the populations studied, however, such disturbances did not conclusively have a detrimental effect on overall population health or demography (Hayden et al. 2002, Delaney et al. 2002, Delaney and Pater et al. 2011, Perkins 2006).

In one study, only a very small proportion of the clusters studied (3 of 51) was found to have a high risk of exposure to military training. This sample, however small, revealed lower nesting and fledgling success than clusters studied with less frequent activity (Hayden et al. 2002). The majority of these studies examined RCW groups that were located on, or adjacent to, established ranges. These groups had likely become acclimated to disturbance, which would also be the case for clusters in Compartments A02, A03 and A30.

Delaney and Pater et al. (2011) found that in clusters normally exposed to moderate to high levels of military training, RCWs did not flush as a result of M16 machine gun live fire from 22,960 ft. up to 656 ft. from the nest tree. In the same study, they also introduced artillery simulator and blank 0.50 cal. machine gun fire to RCW groups that were only habituated to low or moderate levels of military activity. RCWs did not flush from their nests or alter their nestling feeding schedules when

either weapon was fired >500 ft. away. At 400 Ft., RCWs flushed in response to 16.7% of the 0.50 cal. blank fire events and this frequency increased as the distance from the nest tree decreased. On average, RCWs returned to nests within 6.3 minutes after 0.50 cal. fire tests, with a maximum of 26.8 minutes. Even with the disturbances, reproductive success of experimental groups was not statistically different in number of eggs, fledglings, failed nests or other metrics. Of the 8 clusters analyzed in this BO, only Cluster A02-A has any cavity trees within 500 ft. of a range firing point.

Injury or death of RCWs

Although not highly likely, injury or death could result from bullets traveling through clusters or foraging areas. While the Installation has used practical minimization measures to avoid impacts of this type, this risk cannot be completely alleviated. The proposed action would not create high risk areas for direct impacts of this type and would not cause a significant risk of harm to RCWs.

Cluster Level Analysis

Cluster-level analyses, including cluster status, cavity tree impacts, pre-project foraging habitat totals, project impacts and post-project foraging habitat totals are presented below.

Cluster A02-A:

Cluster A02-A was inhabited by a PBG from 2008 to 2012 and consisted of 8 cavity trees (7 active) in various stages of completion and suitability.

Bullet strikes were found within the 200 ft. radius cluster buffer during inspections in 2010 (Ft. Benning 2012a). It was evident that intrusions had been occurring for many years and several of the bullets and wounds appeared to have been less than a few months old (Ft. Benning 2012a). The closest active cavity tree associated with Cluster A02-A is approximately 220 ft. from Farnsworth Range and 745 ft. from the toe at English Range, but this cluster is not in the direct line of fire from either range. This cluster is also 1,345 ft. downrange of Roosevelt Range, but the toe berm at this range is considered to be tall enough to prevent most rounds from breaching. Although it is likely that occasional bullet intrusions will continue to occur, the damage observed was neither widespread nor severe; therefore, Incidental Take due to mortality of cavity trees or foraging habitat is not thought to be necessary for A02-A at this time.

Weapons fired within 500 ft. of nest trees have been found to cause RCWs to flush off of nests while incubating and disrupt feeding of nestlings (Delaney, Pater et al. 2011). Two of the cavity trees associated with Cluster A02-A (both active in 2012) are within 500 ft. of the 5 m zero firing line at Farnsworth Range; however, the remaining 6 cavity trees (including the 2012 nest tree) are >500 ft. from a firing line and 4 of these contain suitable cavities (Ft. Benning, unpub. data). No “take” is considered to be necessary due to harassment impacts.

Using the Managed Stability Standard (MSS), current foraging habitat totals are 2,149.13 ft^2 of pine basal area on 40.81 acres of suitable habitat and 4,589.09 ft^2 of pine basal area on 134.93 acres of future potential habitat. Of the future potential habitat, 2,087.62 ft^2 of pine basal area on 36.70 acres are suitable habitat, but are separated from other foraging stands by pine stands <30 years old, and therefore are temporarily noncontiguous. This cluster currently does not meet MSS standards.

Using the Recovery Standard (RS), current foraging habitat totals are 6,738.21 ft^2 of pine basal area on 175.74 acres of future potential habitat. There was no suitable or potentially suitable habitat. Cluster A02-A contains sufficient total pine habitat and therefore could meet the RS in the future. There is a possibility that the portion of the "future potential" pine plantation closest to Farnsworth Range may never mature into a "suitable" stand due to ordnance impacts; however, these impacts should be greatly lessened by the berm work recently conducted and the other changes described in this BO. This area should, at a minimum, reach a sufficient basal area to serve as a connection between the cluster and the currently noncontiguous habitat on the south side of the partition. Cluster A02-A has 8 active clusters within 1.25 mi. Approximately 8 acres within the A02-A cluster area were treated with herbicide in 2012 to control hardwood midstory.

Cluster A03-A:

Cluster A03-A was inhabited by a PBG from 2008 to 2012. In 2012, there were 13 cavity trees (9 active) considered to be associated with A03-A, but as noted previously, this group recently split and the cavity trees used by the 2 groups are very close together (the nest trees at Clusters A03-A and A03-C in 2012 were 742 ft. apart). The split of the resident group caused most of the foraging habitat previously available to A03-A to be reallocated to Cluster A03-C.

Fort Benning staff observed historical and recent bullet strikes on trees throughout the A03-A cluster area, including cavity trees (Ft. Benning 2012a). The majority of these appeared to have originated from Farnsworth Range and therefore this is where the Army has concentrated its minimization efforts. These measures are expected to greatly reduce the number of rounds entering the cluster, but with the frequency at which Farnsworth Range is used, occasional intrusions are still expected. The closest active cavity tree in Cluster A03-A is approximately 3,450 ft. from the toe of Farnsworth Range; the majority of bullet strikes found at this distance in the study conducted at the Malone Complex were low-velocity. At this distance and at the occasional frequency expected after the berm improvements at Farnsworth Range, the threat of injury or death to cavity trees is not considered to rise to the criteria necessary to meet the criteria of Incidental Take.

Using the MSS, current foraging habitat totals are 55.84 ft^2 of pine basal area on 1.09 acres of suitable habitat, 606.44 ft^2 of pine basal area on 11.23 acres of potentially suitable habitat, and 1,299.77 ft^2 of pine basal area on 33.90 acres of future potential habitat. An additional 4.70 acres of future potential habitat is permanently noncontiguous. This cluster's partition only contains a total of 46.22 acres of contiguous habitat and therefore cannot meet MSS standards now or in the future.

Using the RS, current foraging habitat totals are 55.84 ft^2 of pine basal area on 1.09 acres of potentially suitable habitat and 1,906.21 ft^2 of pine basal area on 45.13 acres of future potential habitat, with an additional 4.70 acres of permanently noncontiguous future potential habitat. There was no suitable habitat. This partition does not contain enough habitat to meet the RS in the future.

Cluster A03-A currently has 14 active clusters within 1.25 mi., and approximately 20.0 acres within the A03-A cluster area and/or foraging partition, were treated with herbicide in 2013 to control hardwood midstory.

Cluster A03-C:

Cluster A03-C was first occupied by a pair of RCWs in 2010 and was inhabited by a PBG from 2010 to 2012. In 2012, this cluster consisted of 11 cavity trees (7 active) in various stages of activity and suitability.

Fort Benning personnel found historical and recent bullet strikes within the southwest edge of this cluster, most of which were thought to have originated at Farnsworth Range, but the majority of the cluster did not appear to be impacted. The closest active cavity tree in Cluster A03-C is 2,855 ft. downrange of Simpson Range and 3,110 ft. from Farnsworth Range.

Using the MSS, current foraging habitat totals are 3,928.60 ft^2 of pine basal area on 65.75 acres of suitable habitat, 1.03 ft^2 of pine basal area on 0.02 acre of potentially suitable habitat, and 1,106.14 ft^2 of pine basal area on 28.75 acres of future potential habitat. An additional 5.88 acres of future potential habitat are permanently noncontiguous. This cluster does not currently meet MSS standards, but contains enough habitat to meet it in the future.

Using the RS, current foraging habitat totals are 694.85 ft^2 of pine basal area on 9.2 acres of potentially suitable habitat and 4,340.92 ft^2 of pine basal area on 85.31 acres of future potential habitat. An additional 5.88 acres of future potential habitat are permanently noncontiguous. There was no suitable habitat. This partition does not contain enough habitat to meet the RS in the future.

There are 13 active clusters within 1.25 mi. of Cluster A03-C. Approximately 14.4 acres within the A03-C cluster area and/or foraging partition were treated with herbicide in 2013 to control hardwood midstory.

Cluster A30-A:

Cluster A30-A was inhabited by a PBG from 2008 to 2012. In 2012, this cluster consisted of 6 cavity trees (2 active) in various stages of activity and suitability. Fort Benning personnel observed some historical and recent bullet strikes within the A30-A cluster area which, based on the azimuth of the strikes and the general location of the cluster, were more likely to be from Simpson or English Ranges than from Farnsworth Range. The frequency of bullet strikes in this cluster is not thought to be high enough to warrant Incidental Take due to mortality of cavity trees or foraging habitat.

Using the MSS, current foraging habitat totals are 705.09 ft^2 of pine basal area on 14.64 acres of suitable habitat and 802.31 ft^2 of pine basal area on 24.71 acres of future potential habitat. This cluster's foraging partition only contains a total of 39.36 acres of habitat and therefore cannot meet the MSS now or in the future.

Using the RS, current foraging habitat totals are 463.08 ft^2 of pine basal area on 9.08 acres of potentially suitable habitat and 1,044.32 ft^2 of pine basal area on 30.28 acres of future potential habitat. This partition does not contain enough habitat to meet the RS in the future. There are currently 15 active clusters within 1.25 mi. of Cluster A30-A. Approximately 22.4 acres within the

A30-A cluster area and/or foraging partition were treated with herbicide in 2013 to control hardwood midstory.

Cluster A30-D:

Cluster A30-D was inhabited by a PBG from 2008 to 2012. In 2012, a second PBG nested on the south side of the cluster; however, only one RCW group was present in Cluster A30-D in 2013. Therefore, as generally recommended by the Service, Fort Benning biologists did not designate the second group in 2012 as a cluster "split". Including trees used by both groups in 2012, Cluster A30-D consisted of 16 cavity trees (11 active) in various stages of activity and suitability.

Fort Benning personnel observed historical and recent bullet strikes on trees within the A30-D partition (Ft. Benning 2012a). Bearings were not measured for every bullet strike; however, based on the location of the cluster alone, bullet intrusions were more likely to have come from Simpson or English Ranges than from Farnsworth Range.

Using the MSS, current foraging habitat totals are 2,957.60 ft^2 of pine basal area on 57.66 acres of suitable habitat, no potentially suitable habitat, and 153.52 ft^2 of pine basal area on 8.51 acres of future potential habitat. This cluster does not currently meet MSS standards and will not be able to in the future (there are only 66.17 acres of habitat in the partition).

Using the RS, current foraging habitat totals are 2,957.60 ft^2 of pine basal area on 57.66 acres of potentially suitable habitat and 153.52 ft^2 of pine basal area on 8.51 acres of future potential habitat. There was no suitable habitat. This partition does not contain enough habitat to meet the RS in the future. There are currently 21 RCW groups within 1.25 mi. of Cluster A30-D. Approximately 57.6 acres within the A30-D cluster area and/or foraging partition were treated with herbicide in 2013 to control hardwood midstory.

Cluster A30-E:

Cluster A30-E was inhabited by a PBG from 2008 to 2012. In 2012, this cluster consisted of 12 cavity trees (6 active) in various stages of activity and suitability. The foraging partition for Cluster A30-E also contains abandoned Cluster A03-B, which was deleted from management with Service approval in 2012.

Fort Benning personnel observed historical and recent bullet strikes on trees within the A30-E partition (Ft. Benning 2012a). Bearings were not measured for every bullet strike, however, based on the location of the cluster alone, bullet intrusions were more likely to have come from Simpson or English Ranges than from Farnsworth Range.

Using the MSS, current foraging habitat totals are 1,819.18 ft^2 of pine basal area on 38.73 acres of suitable habitat and 827.67 ft^2 of pine basal area on 24.03 acres of future potential habitat. An additional 1,645.19 ft^2 of pine basal area on 39.09 acres of suitable habitat and 22.0 ft^2 of pine basal area on 0.40 acre of potentially suitable habitat are permanently separated from the cluster by a hardwood drainage. Not including the noncontiguous habitat, this cluster only has 62.76 acres of habitat in its partition and will not meet the MSS, now or in the future.

Using the RS, current foraging habitat totals are 2,646.85 ft^2 of pine basal area on 62.76 acres of future potential habitat, with an additional 2,161.58 ft^2 of pine basal area on 54.98 acres of noncontiguous future potential habitat. There was no suitable or potentially suitable habitat. Even including noncontiguous habitat, this partition would only have 117.94 acres of habitat in its partition and therefore would not be able to meet RS in the future.

Cluster A30-E currently has 13 RCW groups within 1.25 mi. Approximately 36.5 acres within the A30-E cluster area and/or foraging partition were treated with herbicide in 2013 to control hardwood midstory.

Cluster A30-F:

Cluster A30-F was inhabited by a PBG from 2008 to 2012. In 2012, this cluster consisted of 12 cavity trees (6 active) in various stages of activity and suitability. Cluster A30-F is 4,250 ft. from the nearest range at its closest point, therefore bullet impacts to this cluster are not expected to be severe or to require "take." Using the MSS, current foraging habitat totals are 3,310.36 ft^2 of pine basal area on 64.91 acres of suitable habitat, 878.12 ft^2 of pine basal area on 16.26 acres of potentially suitable habitat, and 45.29 ft^2 of pine basal area on 1.16 acres of future potential habitat. This cluster currently meets the MSS.

Using the RS, current foraging habitat totals are 3,310.36 ft^2 of pine basal area on 64.91 acres of potentially suitable habitat and 923.40 ft^2 of pine basal area on 17.42 acres of future potential habitat. This partition does not contain enough habitat to meet the RS now or in the future. Cluster A30-F currently has 15 active clusters within 1.25 mi. Approximately 56.5 acres within the A30-F cluster area and/or foraging partition were treated with herbicide in 2013 to control hardwood midstory.

Cluster A30-G:

Cluster A30-G was inhabited by a PBG from 2008 to 2012. In 2012, a second PBG nested on the south side of the cluster; however, only one RCW group was present in this cluster in 2013. Therefore, as generally recommended by the Service, Fort Benning biologists did not designate the second group in 2012 as a cluster "split". Including trees used by both groups in 2012, this cluster consisted of 12 cavity trees (9 active) in various stages of activity and suitability. Fort Benning staff noticed a fresh bullet strike on the nest tree in 2009, along with 2 older, healed bullet wounds on other trees in the cluster area (Ft. Benning 2009c). This cluster has been monitored annually for additional bullet intrusions, and new bullet strikes have been infrequent. Therefore, Incidental Take is not considered to be necessary due to damage to nesting or foraging habitat or harassment impacts.

Using the MSS, current foraging habitat totals are 5,180.07 ft^2 of pine basal area on 92.86 acres of suitable habitat, 208.52 ft^2 of pine basal area on 3.86 acres of potentially suitable habitat, and 0.00 ft^2 of pine basal area on 2.14 acres of future potential habitat. This cluster currently meets MSS standards.

Using the RS, current foraging habitat totals are 5,180.07 ft^2 of pine basal area on 92.86 acres of potentially suitable habitat and 208.52 ft^2 of pine basal area on 6.00 acres of future potential habitat. There was no suitable habitat. This partition does not contain enough habitat to meet the RS now or in the future. Cluster A30-G currently has 21 RCW groups within 1.25 mi. Approximately 43.5 acres within the A30-G cluster area and/or foraging partition were treated with herbicide in 2013 to control hardwood midstory.

Group Level Analysis

The group level analysis evaluates effects of group density reduction to clusters directly impacted by ordnance damage from the western Dixie Road ranges, but not "taken" at the cluster level. Group density was calculated for the 8 clusters affected by the proposed action, all of which were determined to have "dense" concentrations. Since no loss or "take" of RCW groups is expected, there will be no reduction of group density.

Neighborhood Level Analysis

The neighborhood level analysis evaluates indirect group density impacts to clusters not directly impacted by the proposed action, but within the 2.57 mile radius "neighborhood." Since no loss of RCW groups is expected, this level of analysis is not necessary and was not conducted.

Population Level Analysis

According to Service guidance (2005), the population level analysis assesses whether the expected loss of RCW groups ("take") based on the cluster, group and neighborhood level analyses "will potentially result in the population's inability to meet its recovery goal." Fort Benning's ability to reach its recovery goal is continually tracked by maintaining spatial data on foraging habitat that will be available after all projects and proposed actions that are currently in progress have been completed. The last recovery map produced by Fort Benning showed sufficient habitat to support 421 clusters (each having ≥150 acres of habitat or <150 acres, but with approval from the Service). Based on averages from the last 5 nesting seasons, 387 total managed clusters will be needed in order to yield 351 PBGs at recovery (Table 1). Note: These totals incorporate 2013 nesting data; totals elsewhere in this document reflect 2012 nesting season data.

(Intentionally blank)

Table 1. Ft. Benning RCW Population Statistics 2009-2013

Year	# Managed Clusters	# Active Clusters	# Active Clusters with PBGs	% of Managed Clusters with PBGs
2009	316	302	287	90.82%
2010	367	349	332	90.46%
2011	369	354	333	90.24%
2012	368	354	337	91.58%
2013	367	357	332	90.46%
---	---	---	Average	**90.71%**

Total # of Managed Clusters Needed for Recovery		% of Managed Clusters with PBGs (5 year average)		Recovery Goal (# PBGs)
387	x	90.71%	=	351

Source: Dixie Road Small Arms Biological Assessment, November 2013

As described above, the proposed action is not expected to result in the loss of RCW groups and will not cause clusters to be unmanageable or otherwise cause them to not count toward the Installation's recovery goal. The latest recovery map showed that Fort Benning could realistically support 421 clusters, which is 34 more clusters than the estimated 387 managed clusters necessary to yield 351 PBGs.

Recovery Unit Analysis

All projections for future recovery are made by the Service using the best available information. To this extent, the most comprehensive data used is a combination of data derived from the Annual RCW Reports which is a dataset of all managed RCW populations, and various other datasets available to the Service's central reporting official, the RCW Recovery Coordinator. Based on the latest data set, this proposed action will not jeopardize the recovery of the species.

CUMMULATIVE EFFECTS

Cumulative effects are defined in the USFWS Consultation Handbook to "include the effects of future State, tribal, local or private actions that are reasonably certain to occur in the action area." Since most future Federal actions will at some point be subject to the Section 7 consultation process, their effects on a particular species will be considered at that time and are not included in the cumulative effects analysis (*Federal Register*, 50 CFR Part 402.02).There are no known off-Post developments meeting the above-listed criteria that are expected to occur within the Action Area.

CONCLUSION

After reviewing the current status of the RCW, the environmental baseline for the Action Area, the effects of the proposed action, and the cumulative effects, it is the Service's biological opinion that the project, as proposed, is not likely to jeopardize the continued existence of this species.

INCIDENTAL TAKE STATEMENT

Section 9 of the Act and Federal regulations pursuant to section 4(d) of the Act prohibit the take of endangered and threatened species, respectively, without special exemption. Incidental take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or 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, carrying out 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 a prohibited taking under the Act, provided that such taking is in compliance with the terms and conditions of this incidental take statement.

The measures described below are non-discretionary, and must be undertaken by Fort Benning so that they become binding conditions of any grant or permit issued to Fort Benning, as appropriate for the exemption in section 7(o)(2) to apply. Fort Benning has a continuing duty to regulate the activity covered by this incidental take statement.

If Fort Benning (1) fails to assume and implement the terms and conditions or (2) fails to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permits or grant documents, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, Ft. Benning 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(1) (3)]

AMOUNT OR EXTENT OF INCIDENTAL TAKE

The Service anticipates no incidental take associated with this action.

EFFECT OF THE TAKE

In the accompanying BO, the Service determined that anticipated take is not likely to occur and therefore is not likely to result in jeopardy to the species or destruction or adverse modification of habitat.

REASONABLE AND PRUDENT MEASURES

We have no additional recommendations.

TERMS AND CONDITIONS

We have no additional recommendations.

CONSERVATION RECOMMENDATIONS

We have no additional recommendations.

REINITIATION NOTICE

This concludes formal consultation on the action outlined in the November 22, 2013, request. As provided in 50 CFR §402.16, reinitiation of formal consultation is required where discretionary Army involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species not considered in this 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 operation causing such take must cease pending reinitiation of consultation.

Sincerely,

for Don Imm
Field Supervisor

CC: W. McDearman, FWS RCW Coordinator
J. Ambrose, GDNR
FWS R4 S7 Coordinator

REFERENCES CITED

Allen, D.H. 1991. Constructing artificial red-cockaded woodpecker cavities. USDA Forest Service General Technical Report SE-73.

Baker, W.W. 1995. The distribution and status of the red-cockaded woodpecker (Picoides borealis) in Georgia, 1992. Pages 465-469 *in* D.L. Kulhavy, R.G. Hooper, and R. Costa, editors. Red-cockaded woodpecker: recovery, ecology and management. Center for applied Studies in Forestry, College of Forestry, Stephen F. Austin State University, Nacogdoches, Texas, USA.

Bowman, R., D.L. Leonard Jr., L.K. Backus, P.M. Barber, A.R. Mains, L.M. Richman and D. Swan. 1998. Demography and habitat characteristics of the red-cockaded woodpecker *(Picoides borealis)* at the Avon Park Air Force Range. Final Report 1994-1998. Archbold Biological Station, Lake Placid, Florida, USA.

Carlile, E.W. Spadgenske, B.A. Macallister and R.H. Melton. 2002. Assessment of training noise impacts on the red-cockaded woodpecker: 1998-2000. US Army Corps of Engineers, ERDC/ ERL Technical Report 02/32, Champaign, IL.

Cely, J.E. and D.P. Ferral. 1995. Status and distribution of the red-cockaded woodpecker in South Carolina. Pages 470-476 *in* D.L. Kulhavy, R.G. Hooper, and R. Costa, editors. Red-cockaded woodpecker: recovery, ecology and management. Center for applied Studies in Forestry, College of Forestry, Stephen F. Austin State University, Nacogdoches, Texas, USA.

Clark, A., III. 1992a. Heartwood formation and loblolly and longleaf pines for red-cockaded woodpecker nesting cavities. Proc. Ann. Conf. Southeastern Association of Fish and Wildlife Agencies. 46: 79-87.

Clark, A., III. 1992b. Influence of the tree factors and site formation heartwood in loblolly and longleaf pine for the red-cockaded woodpecker colonization in the southeast. Final Report U.S. Forest Service, Southeastern Forest Experiment Station, Athens, Georgia, USA.

Conner, R.N. and B.A. Locke. 1982. Fungi and red-cockaded woodpecker cavity trees. Wilson Bulletin 94:64-70.

Conner, R. N. and K. A. O'Halloran. 1987. Cavity tree selection by red-cockaded woodpeckers as related to growth dynamics of southern pines. Wilson Bulletin 99:398-412.

Conner, R.N. and D.C. Rudolph. 1991. Forest habitat loss, fragmentation, and red-cockaded woodpecker populations. *Wilson Bulletin*. 103(3): 446-457.

Conner, R.N., D.C. Rudolph and J.R. Walters. 2001. The red-cockaded woodpecker: surviving in a fire-maintained ecosystem. University of Texas Press, Austin, Texas, USA.

Copeyon, C.K. 1990. A technique for constructing cavities for the red-cockaded woodpecker. Wildlife Society Bulletin 18:303-311.

Costa, R. 2001. Red-cockaded woodpecker. Pages 309-321 *in* J.G. Dickson, editor. Wildlife of southern forests: habitat and management. Hancock House, Blaine, Washington, USA.

Costa, R. 2004. State of the red-cockaded woodpecker world: highlights of the previous decade (1992-2002). Pages 39-46 *in* R. Costa, and S. J. Daniels, editors. Red-cockaded woodpecker: road to recovery. Hancock House, Blaine, Washington, USA.

Costa, R. and N. Jordan. 2003. Biological opinion on all section 10(a)(1)(A) management, monitoring and research permits issued to all private, state and federal agencies and individuals involved with management, conservation and recovery of the red-cockaded woodpecker throughout the range of the species. U.S. Fish and Wildlife Service, Clemson, South Carolina, USA.

Costa, R. and J. Walker. 1995. Red-cockaded woodpecker. Pp. 86-89 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, and others, eds. Our living resources: a repo1i to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S. National Biological Survey, Washington, D. C.

Costa, R. and R. Escano. 1989. Red-cockaded woodpecker: status and management in the southern region in 1986. US Forest Service Technical Publication R8-TP12.

Costa, R. and R.S. Delotelle. 2006. Reintroduction of fauna to longleaf pine ecosystems: opportunities and challenges. Pages 335-376 *in* S. Jose. E.J. Jokela and D.L. Miller, editors. The longleaf pine ecosystem: ecology, silviculture, and restoration. Springer Science + Business Media, Inc., New York, USA.

Crowder, L. B., J. A. Priddy, and J. R. Walters. 1998. Demographic isolation of red-cockaded woodpecker groups: a model analysis, Project Final Report. Duke University, Nicholas School of the Environment, Beaufort, NC.

Daniels, S.J. 1997. Female dispersal and inbreeding in the red-cockaded woodpecker. M.Sc. thesis, Virginia Polytechnic Institutes and State University, Blacksburg, VA.

Daniels, S. J., J. A. Priddy and J. R. Walters. 2000. Inbreeding in small populations of red-cockaded woodpeckers: insights from a spatially explicit individual-based model. Pages 129-147 *in* A. G. Young, and G. M. Clarke, editors. Genetics, demography and viability of fragmented populations. Cambridge University Press, London, UK.

Daniels, S. J. and J. R. Walters. 2000a. Between-year breeding dispersal in red-cockaded woodpeckers: multiple causes and estimated cost. Ecology 81:2473-2484.

Daniels, S. J. and J. R. Walters. 2000b. Inbreeding depression and its effects on natal dispersal in red-cockaded woodpeckers. Condor 102:482-491.

Delaney, D.K., L.L. Pater, R.J. Dooling, B. Lohr, B. Brittan-Powell, L. Swindell, T.A. Beaty, L.D.

Delaney, D.K., P. Guertin, S. Tweddale and M. White. 2011. Evaluation of military range berm effectiveness in protecting red-cockaded woodpecker foraging and nesting habitat. U.S. Corps of Engineers, Engineer Research and Development Center, Construction Engineering Research Laboratory. April 2011. 58 pp.

Delaney, D.K., L.L. Pater, L.D. Carlile, E.W. Spadgenske, T.A. Beaty and R.H. Melton. 2011. Response of red-cockaded woodpeckers to military training operations. *Wildlife Monographs* 177:1-38, 2011.

DeLotelle, R.S. and R.J. Epting. 1992. Reproduction of the red-cockaded woodpecker in central Florida. Wilson Bulletin 104:285-294.

DeLotelle, R.S., R.J. Epting and J.R. Newman. 1987. Habitat use and territory characteristics of red-cockaded woodpeckers in central Florida. Wilson Bulletin 99:202-217.

Doresky, J., K. Morgan, L. Ragsdale, H. Townsend, M. Barron, and M. West. 2001. Effects of military activity on reproductive success of red-cockaded woodpeckers, *Journal of Field Ornithology* 72(2): 305-311.

Engstrom, R. T. and F.J. Sanders. 1997. Red-cockaded woodpecker foraging ecology in an old growth longleaf pine forest. Wilson Bulletin 109: 203-217.

Epting, R. T., R. S. DeLotelle and T. Beaty. 1995. Red-cockaded woodpecker territory and habitat use in Georgia and Florida. Pages 270-276 *in* D. L. Kulhavy, R. G. R.G. Hooper, and R. Costa, eds. Red-cockaded woodpecker: recovery, ecology and management. Center for Applied Studies in Forestry, Stephen F. Austin State University, Nacogdoches, TX.

Fort Benning. 2012a. Memorandum for Record Re: Farnsworth Range - Update, Fort Benning, Georgia. 23 February 2012. 8 pp.

Frost, C.C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. Pp. 17-44 *in* S.M. Hermann, ed. The longleaf pine ecosystem: ecology, restoration, and management. Tall Timbers Fire ecology Conference Proceedings No. 18. Tall Timbers Research Station, Tallahassee, FL.

Haig, S.M., and J.M. Rhymer. 1994. Translocation recommendations for red-cockaded woodpeckers resulting from random amplified polymorphic DNA analysis of populations. South Carolina Coop. Fish and Wildl. Res. Unit Publ. 94-1. 14pp.

Haig, S. M., R. Bowman and T. D. Mullins. 1996. Population structure of red-cockaded woodpeckers in south Florida: RAPDs revisited. Molecular Ecology 5:725-734

Hardesty, J. L., K. E. Gault, and H. F. Percival. 1997a. Ecological correlates of red-cockaded woodpecker (*Picoides boralis*) foraging preference, habitat use, and home range size in northwest Florida (Eglin Air Force Base). Final Report, Research Work Order 99, Florida Cooperative Fish and Wildlife Research Unit, University of Florida, Gainesville, USA.

Hardesty, J.L., K.E. Gault and F.P. Percival. 1997b. Trends, status and aspects of demography of the red-cockaded woodpecker in the sandhills of Florida's panhandle. Final Report Research Work Order 146, Florida Cooperative Fish and Wildlife Research Unit, University of Florida, Gainesville, FL.

Hayden, T.J., R.H. Melton, B. Willis, L.B. Martin, III and T. Beaty. 2002. Assessment of effects of maneuver training activities on the red-cockaded woodpecker populations on Fort Stewart, Georgia. US Army Corps of Engineers, CERL Technical report 02/17. 74 pp.

Hooper, R. G., and C. J. McAdie. 1995. Hurricanes and the long-term management of the red-cockaded woodpecker. Pages 148-166 *in* D. L. Kulhavy, R. G. Hooper, and R. Costa, editors. Red-cockaded woodpecker: recovery, ecology and management. Center for Applied Studies in Forestry, College of Forestry, Stephen F. Austin State University, Nacogdoches, Texas, USA.

Hooper, R. G., L. J. Niles, R. F. Harlow and G. W. Wood. 1982. Home ranges of red-cockaded woodpeckers in coastal South Carolina. Auk 99:675-682.

Hooper, R. G., J. C. Watson and R. E. F. Escano. 1990. Hurricane Hugo's initial effects on red-cockaded woodpeckers in the Francis Marion National Forest. Trans. NA Wildlife and Natural Resources Conference 55:220-224.

Jackson, J.A. 1977. Red-cockaded woodpeckers and pine red heart disease. Auk 94:106-163.

Jackson, J.A. and S.D. Parris. 1995. The ecology of red-cockaded woodpeckers associated with construction and use of a multi-purpose range complex at Fort Polk, Louisiana. Pp 277-282. *in* D.L. Kulhavy, R.G. Hooper and R. Costa (eds). Red-cockaded woodpecker: species recovery, ecology and management. Stephen F. Austin State University, Nacogdoches, TX.

James, F.C., C.A. Hess and B.C. Kicklighter. 2001. Ecosystem management and the niche gestalt of the red-cockaded woodpecker in longleaf pine forests. Ecological Applications 7:854-870.

James, F.C., C.A. Hess and D. Kufrin. 1997. Species-centered environmental analysis: indirect effects of fire history on red-cockaded woodpeckers. Ecological Applications 7:118-129.

Landers, J.L., D.H. Van Lear and W.D. Boyer. 1995. The longleaf pine forests of the southeast: requiem or renaissance? Journal of Forestry 93: 39-44.

Lennartz, M. R., and R. F. Harlow. 1979. The role of parent and helper red-cockaded woodpeckers at the nest. Wilson Bulletin 91:331-335.

Lennartz, M.R. and D.G. Heckel. 1987. Population dynamics of red-cockaded woodpecker population in Georgia Piedmont loblolly pine habitat. Pp. 48-55 *in* R.R. Odom, K.A. Riddleberger, and J.C. Ozier, eds. Proceedings of the third southeast nongame and endangered wildlife symposium. Georgia Department of Natural Resources, Game and Fish Division, Atlanta, GA.

Lennartz, M.R., R.G. Hooper, and R.F. Harlow. 1987. Sociality and cooperative breeding of red-cockaded woodpeckers (*Picoides borealis*). Behavioral Ecology and Sociobiology. 20:77-88.

Letcher, B.H., J.A. Priddy, J.R. Walters and L.B. Crowder. 1998. An individual-based, spatially explicit simulation model of the population dynamics of the endangered red-cockaded woodpecker, *Picoides borealis. Biological Conservation* 86:1-14.

Ligon, J. D. 1970. Behavior and breeding biology of the red-cockaded woodpecker. Auk 87:255-278.

Ligon, J.D., P.B., Stacey, R.N. Conner, C.E. Bock, and C.S. Adkisson. 1986. Report of the AOU Committee for the conservation of the red-cockaded woodpecker. Auk 103:848-855.

Ligon, J.D., W.W. Baker, R.N. Conner, J.A. Jackson, F.C. James, D.C. Rudolph, P.B., Stacey and J.R. Walters. 1991. The conservation crisis- the red-cockaded woodpecker: on the road to oblivion? Auk 108:200-213.

Mills, L. M., K. J. Feltner and T. O. Reed. 2004. The rise and fall of the red-cockaded woodpecker population in Kentucky: a chronology of events preceding extirpation. Pages 392-402 *in* R. Costa, and S. J. Daniels, editors. Red-cockaded woodpecker: road to recovery. Hancock House, Blaine, Washington, USA.

Moore, D. and T. Foti. 2005. Gulf Coastal Plain Flatwoods. Potential Natural Vegetation Group. Rapid Assessment Reference Condition Model. LANDFIRE National Vegetation Dynamics Models. USDA Forest Service and U.S. Department of Interior.

Moore, D. and T. Foti. 2008. West Gulf Coastal Plain Pine-Uplands Flatwoods. Potential Natural Vegetation Group. Rapid Assessment Reference Condition Model. LANDFIRE National Vegetation Dynamics Models. USDA, Forest Service and U.S. Department of Interior.

Nesbitt, S. A., B. A. Harris, A. E. Jerauld, P. J. Skoog, C. B. Brownsmith and G. L. Evink. 1983. Response of a male red-cockaded woodpecker to drastic habitat alteration. Pages 101-104 *in* D. A. Wood, editor. Red-cockaded woodpecker symposium II proceedings. Florida Game and Fresh Water Fish Commission, Tallahassee, Florida, USA.

Pater, L. D.K. Delaney, T.J. Hayden, B. Lohr and R. Dooling. 1999. Assessment of training noise impacts on the red-cockaded woodpecker: preliminary results. US Army Corps of Engineers. CERL Technical Report 99/51. 100 pp.

Perkins, J. L. 2006. Effects of military training activity on red-cockaded woodpecker demography and behavior *and* new territory formation in the cooperatively breeding red-cockaded woodpecker. Masters Thesis, Virginia Polytechnic University, Blacksburg, VA. 68 pp.

Rudolph, D.C. and R.N. Conner. 1994. Forest fragmentation and red-cockaded woodpecker population: an analysis at intermediate scale. *Journal of Field Ornithology*. 65(3):365-375.

Rudolph, D. C., H. Kyle and R. N. Conner. 1990. Red-cockaded woodpeckers versus rat snakes: the effectiveness of the resin barrier. Wilson Bulletin 102:14-22.

Shaffer, M.L. 1987. Minimum viable populations: coping with uncertainty. Pages 69-86 in M.E. Soule, editor. Viable populations for conservation. Cambridge University Press, Cambridge, United Kingdom.

Simberloff, D. 1993. Species-area and fragmentation effects on old growth forests: prospects for longleaf pine communities. Pages 1-14 *in* S.M. Herman, editor. The longleaf pine ecosystem: ecology, restoration, and management. Tall Timbers Fire ecology Conference Proceedings, No. 18. Tall Timbers Research Station, Tallahassee, Florida, USA.

Stangel, P. W. and P. M. Dixon. 1995. Associations between fluctuating asymmetry and heterozygosity in the red-cockaded woodpecker. Pages 225-226 *in* D. L. Kulhavy, R. G. Hooper, and R. Costa, editors. Red-cockaded woodpecker: recovery, ecology and management. Center for Applied Studies in Forestry, College of Forestry, Stephen F. Austin State University, Nacogdoches, Texas, USA.

Stangel, P.W., M.R. Lennartz and M.H. Smith. 1992. Genetic variation population structure of red-cockaded woodpeckers. Conservation Biology 6:283-292.

Wahlenburg, W.G. 1946. Longleaf pine: its use, ecology, regeneration, protection, growth, and management. Charles Lothrop Pack Forestry Foundation and U.S. Forest Service, Washington, D.C., USA.

Walters, J.R. 1990. Red-cockaded woodpeckers: a 'primitive' cooperation breeder. Pp. 69-101 *in* P.B. Stacey and W.D. Koenig, eds. Cooperative breeding in birds. Cambridge University Press, London, UK.

Walters, J.R., L. B. Crowder and J. A. Priddy. 2002b. Population viability analysis for red-cockaded woodpeckers using an individual-based model. Ecological Applications 12:249-260.

Walters, J.R., S.J. Daniels, J.H. Carter, III, P.D. Doerr, K. Brust and J.M. Mitchell. 2000. Foraging habitat resources, preferences and fitness of red-cockaded woodpeckers in the North Carolina Sandhills. Fort Bragg Project Final Report. Virginia Polytechnic Institute and State University, Blacksburg, Virginia, USA, and North Carolina State University, Raleigh, North Carolina, USA.

Walters, J.R., P.D. Doerr and J.H. Carter III. 1988. The cooperative breeding system of the red-cockaded woodpecker. Ethology 78:275-305.

Ware, S., C. Frost, and P.D. Doerr. 1993. Southern mixed hardwood forest: the former longleaf pine forest. Pages 447-493 *in* W.H. Martin, S.G. Boyce, and A.C. Echternacht, editors. Biodiversity of the southeastern United States: lowland terrestrial communities. John Wiley and Sons, New York, New York, USA.

Watson, J. C., R. G. Hooper, D. L. Carlson, W. E. Taylor and T. E. Milling. 1995. Restoration of the red-cockaded woodpecker population on the Francis Marion National Forest: three years post Hugo. Pages 172-182 *in* D. L. Kulhavy, R. G. Hooper, and R. Costa, editors. Red-cockaded woodpecker: recovery, ecology and management. Center for Applied Studies in Forestry, College of Forestry, Stephen F. Austin State University, Nacogdoches, Texas, USA.

U.S. Army Corps of Engineers. 2008. Biological Assessment for Proposed Maneuver Center of Excellence Actions at Fort Benning, Georgia. USACE Mobile District, Mobile, AL. 458 pp., plus appendices.

U.S. Fish and Wildlife Service. 2003. Recovery plan for the red-cockaded woodpecker (*Picoides borealis*): second revision. USFWS, Atlanta, GA. 296 pp.

U.S. Fish and Wildlife Service. 2005. Memo from Walsh. 4 May 2005. Implementation procedures for use of foraging habitat guidelines and analysis of project impacts under the red-cockaded woodpecker (*Picoides borealis*) Recovery plan: second revision.

USFWS and National Marine Fisheries Service. 1998. Endangered Species Consultation Handbook, Procedures for conducting consultation and conference activities under Section 7 of the Endangered Species Act.

Zwicker, S. M. and J. R. Walters. 1999. Selection of pines for foraging by red-cockaded woodpeckers. Journal of Wildlife Management 63:843-852