BIOLOGICAL ASSESSMENT

FOR

CASPIAN TERN MANAGEMENT TO REDUCE PREDATION OF JUVENILE SALMONIDS IN THE COLUMBIA RIVER ESTUARY

U. S. Fish and Wildlife Service Jurisdiction Species

INTRODUCTION

Purpose and Need of Proposed Action

In 1999, National Marine Fisheries Service (NOAA Fisheries) called for the U.S. Army Corps of Engineers (Corps) to eliminate Caspian tern (*Sterna caspia*) nesting from Rice Island (located in the upper Columbia River estuary) in an attempt to decrease the number of juvenile salmonids consumed by terns. In 1999, the Corps initiated a pilot project to relocate the Rice Island tern colony to East Sand Island, near the mouth of the estuary, where marine fish (i.e., non-salmon) were abundantly available to foraging terns.

In 2000, the Corps proposed to complete a project to prevent Caspian tern nesting on Rice Island, in the Columbia River Estuary, while attracting terns to nest on East Sand Island, near the mouth of the river. As a result of the proposed actions in 2000, Seattle Audubon, National Audubon, American Bird Conservancy, and Defenders of Wildlife filed a lawsuit against the Corps alleging that compliance with the National Environmental Policy Act for the proposed action of attracting the large colony of Caspian terns from Rice Island to East Sand Island was insufficient, and against the Service in objection to the potential take of eggs as a means to prevent nesting on Rice Island. In 2002, all parties reached a settlement agreement. The settlement agreement stipulates that the Service, Corps, and NOAA Fisheries prepare an Environmental Impact Statement (EIS) to address Caspian tern management in the Columbia River estuary. The purpose of the proposed action is to comply with the 2002 Settlement Agreement by identifying a management plan for Caspian terns in the Columbia River estuary that reduces resource management conflicts with ESA-listed salmonids while ensuring the conservation of Caspian terns in the Pacific Coast/Western region.

Alternative C, the Preferred Alternative of the draft EIS (DEIS) (Service *et al.* 2004b), would reduce tern predation on juvenile salmonids in the Columbia River estuary by managing habitat to redistribute the tern colony on East Sand Island throughout the Pacific Coast/Western region. This redistribution would be achieved by creating new or enhanced tern nesting habitat throughout the region and reducing the tern nesting site on East Sand Island to 1 to 1.5 acres. To ensure a suitable network of sites is available for terns on a regional scale, the Action Agencies propose to manage nesting habitat for terns in the region to replace twice the amount of occupied nesting habitat that would be lost on East Sand Island.

The proposed habitat creation or enhancement projects are proposed in eight locations in Washington, Oregon, and California: Dungeness National Wildlife Refuge (NWR) in Washington; Fern Ridge Lake, Summer Lake Wildlife Management Area, and Crump Lake in southern Oregon; and Brooks Island, Hayward Regional Shoreline, and Don Edwards NWR, in San Francisco Bay, California. For additional information on the background of the proposed project, the DEIS can be viewed at the following internet site: http://migratorybirds.pacific.fws.gov/CATE.htm.

Listed Species and Critical Habitat – This Biological Assessment (BA) will evaluate species listed under the authority of the Endangered Species Act (ESA) that are under the U.S. Fish and Wildlife Service's (Service) jurisdiction for project related impacts at East Sand Island, Oregon and eight locations in three western states – Washington (1); Oregon (4); and California (3).

Concurrence Request

We are requesting concurrence from the Service that the following species and critical habitat that may be effected, are not likely to be adversely affected by the proposed action: bald eagle (*Haliaeetus leucocephalus*), marbled murrelet (*Brachyramphus marmoratus*), Oregon chub (Oregonickthys crameri), Bradshaw's lomatium (*Lomatium bradshawii*), Kincaid's lupine (*Lupinus sulphureus kincaidii*), Fender's blue butterfly (*Icaricia icarioides fenderi*), California brown pelican (*Pelecanus occidentalis californicus*), California clapper rail (*Rallus longirostris obsoletus*), California least tern (*Sterna antillarum browni*), western snowy plover (*Charadrius alexandrinus nivosus*), delta smelt (*Hypomesus transpacificus*), and salt marsh harvest mouse (*Reithrodontomys raviventris*).

Formal Consultation Request

We are requesting formal consultation, due to likelihood of unavoidable adverse effects, for the following species: bull trout (*Salvelinus confluentus*) and Warner sucker (*Catostomus warnerensis*).

Organization of the Biological Assessment

This BA addresses species listed under the authority of the ESA that are under the Service jurisdiction. This BA will evaluate the impacts to each listed species and critical habitat that may be affected by the proposed action at eight locations in three western states – Washington (1); Oregon (4); and California (3). Project descriptions will be provided for each location where Caspian tern habitat management is proposed. This will be followed by a description of Caspian tern life history and conclusions of potential effects will follow for each species.

DESCRIPTION OF THE PROPOSED ACTION

Dungeness National Wildlife Refuge, Washington

Project Description

Caspian terns nested on the Dungeness NWR in 2003 and 2004 on Dungeness Spit in an area currently designated closed to public use. The proposed action is to improve protection of this tern colony in order to provide suitable habitat conditions for additional Caspian tern pairs that may arrive at this location as a result of management efforts that aim to reduce tern predation on juvenile salmonids by redistributing the large tern colony at East Sand Island in the Columbia River estuary. The principal means to afford greater protection for the Dungeness NWR Caspian tern colony would be management of potential human disturbance (e.g., placement of additional signs to mark existing closed areas, increased outreach) and predators (e.g., fence around the colony). No habitat modifications are proposed at this site.

Research activities to monitor the Caspian tern colony occurred in 2004. It is anticipated that research activities will continue to occur at the colony location after implementation of the preferred alternative of the EIS. Research activities include construction of a blind near the colony site, prior to the arrival of nesting terns (early April), and personnel accessing the site daily via all terrain vehicles and by foot on the existing public use trail from early April through September. Research activity is accomplished from observations while sitting in the observation blind. Certain research activities require personnel to be outside the blinds for extended periods of time.

The site-specific species list (Service REF: 1-3-04-SP-0661) provided by the Service dated March 24, 2004 identified endangered and threatened species that may occur in the vicinity of the proposed project. These species include the bald eagle, bull trout, and marbled murrelet.

East Sand Island, Oregon

Project Description

Caspian tern habitat management activities at East Sand Island would continue in the manner that they have been implemented since 2000. Specifically, site tillage at the designated tern nesting area is implemented in late March – early April to provide a bare sand nesting substrate for Caspian terns. All heavy equipment will be barged to and from the site. Equipment transported by tug and barge may embark and disembark from Port facilities from Portland, Oregon downstream to Chinook, Washington, depending on the contractor and equipment to be used. Typically 1-2 transit days are required for each leg of the trip to get equipment on and off East Sand Island. Tillage entails disking the site (several replications) followed by roughly leveling the site with a heavy drag harrow. This action occurs as the first birds of the colony begin to arrive on location. Periodically, sand will be added to the site to replace substrate material lost via either wind or water erosion (e.g., every 3-5 years). Replacement sand is borrowed from the upper beach zone using a tracked excavator to load the material on an off-road truck and transport it onto the colony location. A tractor with front-end loader is then used to roughly level the piles of sand and then tillage operations are used to further level the site.

Herbicide treatment is used in the fall to control European beachgrass and American dunegrass on the tern colony location. Herbicide (Rodeo) is applied per label instructions. Typically Rodeo is applied using an ATV mounted sprayer. All herbicide application occurs in an upland situation and during timeframes of low wind velocity to minimize the potential for drift to waters of the Columbia River.

Avian research activities will occur on East Sand Island during the course of the Caspian tern management actions. They entail researchers who camp and/or access the island via boat from late March into September. Most research activity is accomplished from blinds. Certain research activities require personnel to be outside the blinds for extended periods of time. These activities include set up and Operations and Maintenance (O&M) of speaker systems, capture and banding of terns, collection of terns for food habits analysis, brown pelican surveys and miscellaneous research activities.

Based on discussions with Service personnel during previous consultation efforts (BO log #:1-7-01-F-463, file # 8330.4633[01] and FWS reference log #: 1-7-03-I-269) associated with avian predation management and research activities in the lower Columbia River, we have established conservation measures that are employed to minimize the potential impacts to the following endangered and threatened species that occur in the vicinity of, and may be affected by the proposed project: bald eagle and California brown pelican. Please refer to the above referenced documents for a description of these conservation measures.

Fern Ridge Lake, Oregon

Project Description

A 1-acre island is proposed for construction to provide nesting habitat for Caspian terns at Fern Ridge Lake. Island construction would occur during the fall-winter (e.g., October through end of January) timeframe to take advantage of annual drawdown for flood storage. Hardened roadbeds, i.e., former county roads now within the pool boundaries, would be used to access the site during the construction period. These roadbeds have previously been used for hauling materials for construction of a dike adjacent to the proposed island location, thus their capabilities to handle heavy construction traffic are known.

The island's location in open lake waters necessitates that riprap be used to armor the shoreline to prevent wave erosion of island materials. It is estimated that Class III riprap would be sufficient for shoreline armoring and would be used as a shell around the island. Quarry waste and/or material borrowed from the dry bed of the reservoir adjacent to the island location would be used to form the bulk of the island. Borrowed material would be tested for contaminants prior to excavation. A six-inch lift of 1.5 inch minus would be placed atop the quarry waste/borrow material and a one-foot layer of sand/pea gravel placed atop the 1.5 inch minus to provide nesting substrate for Caspian terns. Sand/pea gravel, gravel and rip rap can be obtained locally from gravel companies.

The project island location in the lake is at an average elevation of 369 feet. Full pool occurs at elevation 373.5 feet with flood storage potential to elevation 375.1 feet. It is proposed to construct the island to at least elevation 376 feet initially and concede that some settling of base material will occur. This should keep the island surface well above normal full pool elevation. Flood storage surcharge to elevation 375.1 feet is very unusual and would be short-term in nature with such events typically occurring in winter.

The proposed island would be square with side dimensions of approximately 208.7 feet at its crest. Island shape is for simplicity of construction. Approximately 5.5 feet of base fill would equate to 8,872 cubic yards (cy) of quarry waste/borrow material from the lakebed. A six-inch lift of 1.5 inch minus rock would be placed atop quarry rock to prevent sand from sifting downward into the base material; this equates to 807 cy. A one-foot sand layer would require 1,613 cy of material. Revetment yardage is 800 cy of Class III stone.

Future O&M requirements to maintain Caspian tern nesting habitat at the Fern Ridge Lake Island are anticipated to be minimal. Shoreline revetment would be installed to prevent erosion from wave action and is expected to have a greater than 50-year project life. A sand surface material may require periodic replenishment due to wind erosion; use of pea gravel would negate wind-erosion of surface material. Replacement material could be placed as needed in fall or winter after drawdown occurs. It is anticipated that weeds would have to be removed from the site annually, either by hand pulling or spot application of herbicide (Rodeo).

Avian research activities may occur at the Fern Ridge Lake nesting island during the course of the Caspian tern management actions, if the tern colony reaches a size (e.g., 500 pairs) identified in the EIS Adaptive Management and Monitoring Plan (to be completed after the EIS is finalized) that would trigger research actions. Should research activities occur, they would entail researchers accessing the island location from late March into September via boat. Most research activity is accomplished from blinds. Certain research activities require personnel to be outside the blinds for extended periods of time.

The site-specific species list (Service Ref#1-7-04-SP-0241) provided by Service dated April 22, 2004 identifies the following endangered and threatened species that occur in the vicinity of, and may be affected by the proposed project: bald eagle, Oregon chub, Bradshaw's lomatium, Kincaid's lupine, and Fender's blue butterfly.

Conservation Measures

1. Fenced exclusion areas will be established and occupied habitat will be flagged prior to proposed construction and maintenance activities and monitored during these activities, to avoid destruction or degradation of Kinkaid's lupine and Bradshaw's lomatium habitat that occur in the project area.

2. Construction, maintenance and monitoring activities will be restricted to Royal Avenue and the lake.

3. The following analysis will be conducted for fill materials obtained from an area with prior history of agriculture and possible releases from mining sites (upstream sources than drain into the lake or reservoir): (1) an organochlorine scan with total PCB or Aroclor PCB value; (2) a metal scan, with mercury prepared for (and determined by) cold vapor atomic absorption spectroscopy; and (3) grain size analysis could be determined as well as total organic carbon and percent moisture for sediments. Sediment samples would consist of surface grabs (composite samples are recommended) and cored samples to estimate concentrations at depth (represented concentrations that will remain on the surface after dredging or moving the sediment). These efforts will be conducted by the Action Agencies in coordination with the Service.

Summer Lake Wildlife Management Area, Oregon

Project Description

Caspian tern nesting habitat would be prepared at the Summer Lake Wildlife Management Area (WMA) and would entail the development of three 0.5-acre islands within existing diked wetland impoundments. The locations of these islands and construction details are provided in the following text.

East Link Location

The East Link location at Summer Lake is a diked, rectangular impoundment with water control features (vertical risers with flashboards connected to corrugated metal pipes that run through the dikes) to allow for ingress and egress of water to the unit. The soil is very silty sand with an alkaline crust when dry. Heavy equipment, such as cats, can run on this soil when it has been dry for a considerable length of time. However, subsurface soil can be wet and very mucky even when the surface is dry. Care must be taken when operating equipment on this soil.

To obtain the proper dry soil conditions, prior coordination with Marty St. Louis, Manager of the Summer Lake WMA, will be required, to attain water shutoff to the unit in late November-early December of the year preceding the late-July through September construction period for the Caspian tern nesting island. Water would not be applied to the unit until construction has been completed under this scenario.

For construction purposes the island will be 150 feet x 150 feet. The island will rise three feet above the normal surface elevation of the unit and require 2,500 cy of material to accomplish this objective. The nesting island will be centered in the unit offshore of the second (northern) water control structure. Centering the island will result in it being placed approximately 585 feet off either north-south running dike; the unit is 1,320 feet wide from east to west.

A construction access road will be required for gravel trucks to reach the constructed island for placement of gravel for erosion protection along the island shoreline and for placement of six inches of fine gravels as a surface coat on the island. The access road will be 585 feet in length by 12 feet crest width; sideslopes are 1:2. The construction access road will be formed using a tracked excavator to borrow soil from either side of the roadway to form a raised roadbed (threc feet). The tracked excavator may have to operate on mats if subsurface soil conditions cannot support operating equipment. Approximately 1,000 cy of material will be required to form this roadbed. There will be no turnouts. The raw material will be compacted by tracked equipment running over it and a one-foot layer of quarry waste (~260 cy) placed atop the material to support gravel trucks. The access road, including the quarry waste, will be removed upon completion of the island and sidecast back into the borrow pits from which it was constructed. Quarry waste will be obtained from a nearby Oregon Department of Fish and Wildlife (ODFW) quarry and is large in size, principally rock of various sizes with a minimum amount of fines, which are principally rock dust, and is excavated from a pit on the side of a sagebrush covered hill not subject to agricultural or urban wastes. Thus the likelihood of the presence of contaminants is absent.

The 2,500 cy of material required to construct the island will come from either of two methods. If site conditions are suitable, cats will be used to push material to the island from adjacent land. Given a 600-foot overall perimeter length and two foot borrow depth; cats would have to scalp soil in a 60 feet width to obtain the desired volume of material. A one foot borrow depth would push the borrow width to approximately 115 feet in width. The soil would be pushed into the island perimeter, leveled and compacted.

The second method to obtain the necessary borrow material for construction of the nesting island entails the use of tracked excavators and trucks. Dry soil formerly sidecast from excavation of the East Link canal will be borrowed and trucked to the site and dumped. To facilitate hauling and dumping of this soil to form the island, a dump pad 150 feet by 50 feet will be constructed from either on-site materials excavated by tracked excavators, compacted and rocked, or by material hauled onto site and placed and compacted by cats, then rocked. The material placed will form part of the nesting island. Approximately 280 cy of quarry waste will be used to form a surface from which the trucks will operate. Soil placed for island construction will be pushed into place, leveled and compacted by cats. All soils used for island construction will be tested for contaminants prior to excavation.

Once the island is completed, a top dressing of approximately 450 cy of relatively fine gravels (~pea-size minus) obtained from the ODFW quarry will be placed on the island. This material is intended to provide a suitable nesting substrate for terns versus the native soil that can become quite sticky and interfere with egg rotation or ensnare young tern chicks. The quarry site, also the location for quarry waste materials, is approximately 8 miles from the island location.

A one-foot layer of bedding gravel comprised of approximately 150 cy of relatively fine gravels (~pea-size minus) obtained from the ODFW quarry will be placed on the island perimeter. Cobbles (~150 cy of 8-12 inch minus) obtained from the ODFW quarry will be placed atop the bedding material in a one-foot lift to provide erosion protection to the island.

An estimated one-mile of single lane dike road will have to be maintained during construction. These roadbeds are approximately 10 feet wide. It is anticipated that a 6-inch top dressing (420 cy of 3/4 inch minus to 1.5 inch minus gravels) will have to be placed to maintain these dike roadways. Gravel would be available from the ODFW pit although sorting would be required to obtain the preferred size.

Windbreak and Gold Dike Nesting Island Locations

Two islands will be constructed, one off Windbreak Dike and one off Gold Dike. The first location is off Windbreak Dike approximately 1,000 feet north of the junction with Gold Dike and approximately 1,000 feet west of Windbreak Dike. The second location is off Gold Dike, west of the Ana River, and approximately 1,000 feet north of Gold Dike.

The Windbreak Dike and Gold Dike locations at Summer Lake are within a diked impoundment with water control features to allow for ingress and egress of water to the unit. The soil is very silty sand with an alkaline crust when dry. Heavy equipment, such as cats, can run on this soil when it has been dry for a considerable length of time. However, subsurface soil can be wet and very mucky even when the surface is dry. Care must be taken when operating equipment on this soil. The Windbreak Dike and Gold Dike locations are typically wetter than the East Link location thus operation of equipment on the soil surface will be more difficult and prone to problems.

To obtain the proper dry soil conditions, prior coordination with Marty St. Louis, Manager, Summer Lake WMA, will be required to attain water shutoff to the unit in late November-early December of the year preceding the late July – September construction period for the Caspian tern nesting island. Water would not be applied to the unit until construction has been completed. It is anticipated the these two islands would be constructed in one season but a year apart from the island to be constructed in the East Link unit.

Development of Caspian tern nesting sites at these locations will require construction of two islands, approximately 0.5 acre each; for construction purposes the islands will each be 150 feet x 150 feet. The islands will rise three feet above the normal surface elevation of the unit and require 2,500 cy of material each to accomplish this objective. The nesting islands will be approximately 1,000 feet off either dike.

A construction access road will be required in order for gravel trucks to reach each of the constructed islands for placement of gravel and cobbles for erosion protection along the island shoreline and for placement of six inches of fine gravels as a surface coat on the islands. Each access road will be 1000 feet in length by 12 feet crest width; sideslopes are 1:2. The construction access road will be formed using a tracked excavator to borrow soil from either side of the roadway to form a raised roadbed (three feet). The tracked excavator may have to operate on mats if subsurface soil conditions cannot support operating equipment. Approximately 1,700 cy of material will be required to form this roadbed. There will be no turnouts. The raw material will be compacted by tracked equipment running over it and a one-foot layer of quarry waste (~450 cy) will be placed on each roadbed to support gravel trucks. The access roads including quarry waste will be removed upon completion of the island and sidecast back into the borrow pits from which it was constructed. Quarry waste will be obtained from a nearby ODFW quarry and would not be tested for contaminants prior to deposition at the construction site. Quarry waste is large in size, principally rock of various sizes with a minimum amount of fines, which are principally rock dust, and is excavated from a pit on the side of a sagebrush covered hill not subjected to agricultural or urban wastes. Thus the likelihood of the presence of contaminants is absent.

The 2,500 cy of material required to construct each island will come from either of two methods. If site conditions are suitable, cats will be used to push material to the islands from adjacent land. Given an overall 600-foot perimeter length and two foot borrow depth; cats would have to scalp soil in a 60 feet width to obtain the desired volume of material. A one foot borrow depth would push the borrow width to approximately 115 feet in width. The soil would be pushed into the island perimeter, leveled, and compacted.

The second method to obtain the necessary borrow material for construction of the nesting islands entails the use of tracked excavators and trucks. Dry soil formerly sidecast from excavation of the East Link canal will be borrowed and trucked to the site and dumped. To facilitate hauling and dumping of this soil to form the island, a dump pad 150 feet by 50 feet will be constructed from either on-site materials excavated by tracked excavators, compacted and rocked, or by material hauled onto site and placed and compacted by cats, then rocked. The material placed for the dump pad will form part of the nesting island. Approximately 280 cy of quarry waste will be used to form a surface from which the trucks will operate. Soil placed for island construction will be pushed into place, leveled and compacted by cats. All soil used will be tested for contaminants prior to excavation.

Once the island is completed, a top dressing of approximately 450 cy of relatively fine gravels (~pea-size minus) obtained from the ODFW quarry will be placed on each island. This material is intended to provide a suitable nesting substrate for terns versus the native soil that can become quite sticky and interfere with egg rotation or ensnare young tern chicks. The quarry site, also the location for quarry waste materials, is approximately 8 miles from the island location.

A one-foot layer of bedding gravel comprised of approximately 150 cy of relatively fine gravels (~pea-size minus) obtained from the ODFW quarry will be placed on the perimeter of each island. Cobbles (~150 cy of 8-12 inch minus) obtained from the ODFW quarry will be placed atop the bedding material in a one-foot lift to provide erosion protection to each island.

An estimated 2 miles of single lane dike roads will have to be maintained during construction to access the constructed roadway leading to the island off Windbreak Dike. An additional one mile of dike roadbed will have to be maintained to access the constructed roadway leading to the island off Gold Dike. These roadbeds are approximately 10 feet wide. It is anticipated that a 6-inch top dressing of 3/4 inch minus to 1 and 1/2 inch minus gravels will have to be placed to maintain these dike roadways. Approximately 2,000 cy of these gravels would be required to maintain the roadway leading to the Windbreak Dike access road. An additional 1,000 cy would be required from that point to the access road location on Gold Dike. Gravel would be available from the ODFW pit although sorting would be required to obtain the preferred size. These quarry materials are routinely used for road rock on the WMA.

Future O&M requirements to maintain Caspian tern nesting habitat at the Summer Lake islands for Caspian tern nesting are anticipated to be minimal. Shoreline revetment would be installed to prevent erosion from wave action and is expected to have a greater than 50-year project life. A sand surface material may require periodic replenishment due to wind erosion; use of pea gravel would negate wind-erosion of surface material. Replacement material could be placed as needed after drawdown occurs on these impoundments where the islands are located and the islands can be easily accessed. It is anticipated that weeds would have to be removed from the site annually, either by hand pulling or spot application of herbicide.

Avian research activities may occur at the Fern Ridge Lake nesting island during the course of the Caspian tern management actions, if the tern colony reaches a size (e.g., 500 pairs) identified in the EIS Adaptive Management and Monitoring Plan (to be completed after the EIS is finalized) that would trigger research actions. Should research activities occur, they would entail researchers accessing the locations from late March into September using existing access roads (currently open to the public) and a canoe to reach the islands. Most research activity is accomplished from blinds. Certain research activities require personnel to be outside the blinds for extended periods of time.

The site-specific species list (Service REF: 1-7-04-SP-0241a) provided by the Service dated April 22, 2004, identified the endangered and threatened species that may occur in the vicinity of the proposed project. These species are bald eagle and Warner sucker.

Conservation Measures

1. The following analysis will be conducted for fill materials obtained from an area with prior history of agriculture and possible releases from mining sites (upstream sources that drain into the lake or reservoir): (1) an organochlorine scan with total PCB or Aroclor PCB value; (2) a metal scan, with mercury prepared for (and determined by) cold vapor atomic absorption spectroscopy; and (3) grain size analysis could be determined as well as total organic carbon and percent moisture for sediments. Sediment samples would consist of surface grabs (composite samples are recommended) and cored samples to estimate concentrations at depth (represented concentrations that will remain on the surface after dredging or moving the sediment). These efforts will be conducted by the Action Agencies in coordination with the Service.

Crump Lake, Oregon

<u>Project Description</u>

Crump Island, in Crump Lake, was formerly a natural island located approximately mid-lake and north of the peninsula that nearly bisects the lake. Surface elevation of Crump Lake is 4,475 feet. Previous human disturbance led to erosion of the island to lakebed level, eliminating its use by colonial nesting birds. An effort led by ODFW in the 1990's was partially successful in restoring the island. Working when the lake had dried up due to drought, ODFW pushed material up with cats to reform the island. Unfortunately, the island height did not exceed high water levels and thus is inundated or nearly so during higher water periods. Erosion of soil placed to form the island was also an issue.

Crump Island is approximately 1.25 miles offshore and is situated in waters 2-10 feet in depth. A rudimentary boat ramp exists in the southern part of the lake. The island is too far offshore for construction of a causeway to haul materials into place. Potentially the island could be reconstructed during a future drought but there is no certainty when such a situation will occur or if it will last long enough for the lakebed to support heavy equipment and dump trucks. There is also the issue of stabilizing or armoring the island shoreline to prevent future erosion.

Utilization of a "mudcat," a small hydraulic dredge that places material to form the island, appears to be logistically possible and would allow for construction to occur most years. Retention of material pumped to the location over the short- and long-term remains the most pressing problem. The Corps has preliminarily explored means to contain dredged material at the location and provide erosion protection. These include conventional stone revetment, an interlocking, plastic sheet pile wall, terracells stacked upon one another and filled with rock, and geotubes consisting of geotextile tubes filled with dredged material.

No specific design has been selected. Logistically, placement of conventional stone revetment would be physically impracticable and prohibitively expensive. The use of terracells would also require a substantial amount of imported material such as rock or sand, which presents the same logistical problems as stone revetment. The installation of a sheet pile wall does not appear capable of withstanding the pressure generated by the material placed interior to the wall. Thus at this point in time, the use of geotubes to form the island's perimeter and to retain dredged material placed interior to them appears to be the most likely scenario for construction of Crump Island. A wall comprised of 40-sheet pile would be used to form a 20 feet x 20 feet cell to serve as a settling pond. A 2-4 feet space in the perimeter wall at the settling pond location will be installed one foot lower than the balance of the perimeter wall in order to decant water to the outside from the settling pond. The width of this opening can be increased by the contractor to ensure that outflow is sufficient to prevent overtopping of the perimeter wall by water during construction of the island via hydraulic dredge.

Construction of the island would occur in June when water levels should be at their highest. The hydraulic dredge would excavate material for placement within the island perimeter. Dredging activities will occur 200 feet or greater from the island site to be constructed. For construction purposes, we are assuming that the water depth at the island location is 10 feet, although remnant parts of the former island are present. Thus, an estimated 19,400 cy of material are required to form an island that rises uniformly two feet above full pool level. This should leave two feet of freeboard on the perimeter walls. Dredged material will be pumped to the point furthest from the settling pond location and then moved closer as material accumulates.

To stabilize the surface of the constructed island and to reduce the risk of dense vegetation encroachment, the dredged material would be capped with gravel and fines. Gravel graded as ¾-inch minus to 1.5-inch minus would be placed atop the island in an approximately 6-inch lift (820 cy). It is assumed that the material will have to be placed on site via helicopter. A quarry is located on private lands approximately 1.5 miles west of Crump Island. The 820 cy would have to be purchased and crushed/graded to attain the proper size. Estimated weight of the material is 1,230 tons at 1.5 tons per cubic yard. The material from the private quarry would not be tested for contaminants prior to deposition at the construction site. Quarry waste is large in size, principally rock of various sizes with a minimum amount of fines, which are principally rock dust, and is excavated from a pit on the side of a sagebrush covered hill not subjected to agricultural or urban wastes. Thus, the likelihood of the presence of contaminants is absent.

Placement via helicopter would occur in November-December when the dredged material has had sufficient time to dry and helicopters would be available post-fire season. An estimated 22 hrs of helicopter work spread over 3-5 days, depending upon weather conditions, would be required to place this material.

Future O&M requirements to maintain Caspian tern nesting habitat at Crump Island are anticipated to be minimal. Shoreline revetment/protection would be installed to prevent erosion from wave action and is expected to have a greater than 50-year project life. A sand surface material may require periodic replenishment due to wind erosion; use of pea gravel would negate wind-erosion of surface material. Replacement material would be placed as needed in fall or winter after nesting birds have left the area. It is anticipated that weeds would have to be removed from the site annually, either by hand pulling or spot application of herbicide.

Avian research activities may occur at the Fern Ridge Lake nesting island during the course of the Caspian tern management actions, if the tern colony reaches a size (e.g., 500 pairs) identified in the EIS Adaptive Management and Monitoring Plan (to be completed after the EIS is finalized) that would trigger research actions. Should research activities occur, they would entail researchers accessing the locations from late March into September via a John boat. Most research activity is accomplished from blinds. Certain research activities require personnel to be outside the blinds for extended periods of time.

The site specific species list (Service REF: 1-7-04-SP-0241a) provided by the Service dated April 22, 2004 identified the endangered and threatened species that may occur in the vicinity of the proposed project. These species are bald eagle and Warner sucker.

Conservation Measures

1. The following analysis will be conducted for fill materials obtained from an area with prior history of agriculture and possible releases from mining sites (upstream sources that drain into the lake or reservoir): (1) an organochlorine scan with total PCB or Aroclor PCB value; (2) a metal scan, with mercury prepared for (and determined by) cold vapor atomic absorption spectroscopy; and (3) grain size analysis could be determined as well as total organic carbon and percent moisture for sediments. Sediment samples would consist of surface grabs (composite samples are recommended) and cored samples to estimate concentrations at depth (represented concentrations that will remain on the surface after dredging or moving the sediment). These efforts will be conducted by the Action Agencies in coordination with the Service.

Brooks Island, California

Project Description

The actions proposed at Brooks Island entail primarily vegetation removal and enhancement of the substrate. Vegetation removal includes hand or mechanical removal of non-native plants to provide a bare surface for nesting. Substrate enhancement may entail the addition of sand, pea gravel, or other suitable material for nesting Caspian terns. Predator control may also occur to facilitate maintenance of a Caspian tern colony at this location. Predator management is already implemented as needed to remove red foxes and raccoons or other comparable predators that consume eggs, young, and/or adult colonial nesting birds. Predator management associated with this proposed action falls within the current program design. Future O&M requirements to maintain Caspian tern nesting habitat at this site are anticipated to be minimal and include the above described actions as needed.

Avian research activities may occur at the Brooks Island location during the course of the Caspian tern management actions. Should research activities occur, they would entail researchers accessing the locations from early March into September via boat to the island and then walking to the colony site. Most of the research activity would be accomplished from a blind. Certain research activities require personnel to be outside the blinds for extended periods of time. Research activities would be coordinated with the East Bay Regional Parks District and a park permit will be acquired if necessary.

The site specific species list provided by the Service dated March 24, 2004 identified the following endangered and threatened species may occur in the vicinity of, and may be affected by the proposed project: California brown pelican, California least tern, bald eagle, delta smelt, and designated critical habitat for delta smelt.

Hayward Regional Shoreline, California

Project Description

The actions proposed at Hayward Regional Shorelines entail removal of vegetation and enhancement of the substrate on existing islands within a series of freshwater ponds. Management for Caspian tern habitat is proposed for islands 2, 6, and/or 7. Water within these ponds includes a mixture of treated wastewater and bay water (salt water).

Vegetation removal would be accomplished via mechanical means, either light construction equipment, and/or personnel using hand tools (no heavy equipment required); herbicides would be potentially used if necessary. Removal of vegetation is required in order to provide a bare surface for tern nesting. The site may also be saturated with salt water to inhibit future vegetation growth. Substrate enhancement may entail the addition of a filter fabric over the surface area of the island that would subsequently be covered with sand, pea gravel, or other suitable material for nesting Caspian terns. Sand and other materials would be transported to these islands via boat and/or helicopter. The proposed action is similar to previous enhancement action at these ponds done for California least terns, including spraying vegetation, laying down rock salt, covering the surface with fabric cloth, and finally laying down 110 tons of sand.

Social facilitation, i.e., the use of tern decoys and a sound-system for continuous playback of tern colony vocalizations, will also be used at this location. Predator management may also occur to facilitate establishment and maintenance of a Caspian tern colony at this location. Predator control is already implemented as needed at Hayward Regional Shoreline to remove red foxes and raccoons or other predators that consume eggs, young and/or adult birds of herons, egrets, Forster's terns, and shorebirds. Predator management associated with this proposed action falls within the current program design.

Future O&M requirements to maintain Caspian tern nesting habitat at the Hayward Regional shorelines are anticipated to be minimal and would include the above described actions as needed. Shoreline revetment would not be installed as the islands are within diked ponds. A sand surface material may require periodic replenishment due to wind erosion; use of pea gravel would negate wind-erosion of surface material. Replacement material could be placed as needed in fall or winter after terns and other nesting birds have finished nesting and probably left the area. It is anticipated that weeds would have to be removed from the site annually, either by hand pulling or spot application of herbicide.

Avian research activities may occur at the Fern Ridge Lake nesting island during the course of the Caspian tern management actions, if the tern colony reaches a size (e.g., 500 pairs) identified in the EIS Adaptive Management and Monitoring Plan (to be completed after the EIS is finalized) that would trigger research actions. Should research activities occur, they would entail researchers accessing the locations from early March into September via kayak or observations from the dike. Research activities would be coordinated with the East Bay Regional Parks District and a park permit will be acquired if necessary.

The site specific species list provided by the Service dated March 24, 2004 identified the following endangered and threatened species may occur in the vicinity of, and may be affected by, the proposed project: California brown pelican, California clapper rail, California least tern, bald eagle, western snowy plover, delta smelt, designated critical habitat for delta smelt, and salt marsh harvest mouse.

Conservation Measures

1. To avoid adverse impacts to listed species, particularly the California clapper rail and salt marsh harvest mouse, associated with ingress and egress at the project site, a map indicating vehicle access route and helicopter flight path restrictions during construction, maintenance and monitoring activities will be provided to the Sacramento Fish and Wildlife Office for review and approval prior to initiating construction related activities at the site. If it is determined at the time that, for reasons not considered here, adverse impacts cannot be avoided to one or more species, we will reinitiate consultation at that time.

Don Edwards National Wildlife Refuge, California

Project Description

The actions proposed on Don Edwards NWR entail enhancement of the substrate on existing levees within Ponds N1/N9. These lie within active salt ponds currently managed by Cargill for salt production. Substrate enhancement may entail the addition of a filter fabric over the surface area of the levees that would subsequently be covered with sand, pea gravel or other suitable material for nesting Caspian terns. It will be necessary to do construction and O&M work between November and February or early March. Terns and other nesting birds start prospecting by mid-to-late March in the Bay Area. Social facilitation, i.e., the use of tern decoys and a sound-system for continuous playback of tern colony vocalizations, may also be used at this location. Predator control and/or gull harassment/control may also occur to facilitate establishment and maintenance of a Caspian tern colony at this location. Predator control is currently implemented as needed on the NWR to protect listed species. Typically red foxes and raccoons or other comparable predators consume eggs, young and/or adult birds or the clapper rail or snowy plover. Predator management associated with this proposed action falls within the current program design. Substrate enhancement and predator management would need to be repeated as needed in the future.

Future O&M requirements to maintain Caspian tern nesting habitat at the Don Edwards NWR are anticipated to be minimal. Shoreline revetment would not be installed as the islands occur within diked areas. A sand surface material may require periodic replenishment due to wind erosion; use of pea gravel would negate wind-erosion of surface material. Replacement material could be placed as needed in fall. It is anticipated that weeds would have to be removed from the site annually, either by hand pulling or spot application of herbicide.

Avian research activities may occur at the Fern Ridge Lake nesting island during the course of the Caspian tern management actions, if the tern colony reaches a size (e.g., 500 pairs) identified in the EIS Adaptive Management and Monitoring Plan (to be completed after the EIS is finalized) that would trigger research actions. Should research activities occur, they would entail researchers accessing the locations from early March into September via kayak, foot and/or vehicle. Most research activity is accomplished from blinds. Certain research activities may require personnel to be outside the blinds for extended periods of time. Research activities would be coordinated with the Service and a refuge Special Use Permit will be acquired if necessary.

The site specific species list provided by the Service dated March 24, 2004 identified the following endangered and threatened species may occur in the vicinity of, and may be affected by the proposed project: California brown pelican, California clapper rail, California least tern, bald eagle, western snowy plover, and delta smelt.

Conservation Measures

1. To avoid adverse impacts to listed species, particularly the California clapper rail and salt marsh harvest mouse, associated with ingress and egress at the project site, a map indicating vehicle access route and helicopter flight path restrictions during construction, maintenance, and monitoring activities will be provided to the Sacramento Fish and Wildlife Office for review and approval prior to initiating construction related activities at the site. If it is determined at the time that, for reasons not considered here, adverse impacts cannot be avoided to one or more species, we will reinitiate consultation at that time.

CASPIAN TERN BIOLOGY

Species Range

Caspian terns breed at widely scattered sites across North America. Wires and Cuthbert (2000) described five disjunct breeding regions in North America. Caspian terns breeding in the Columbia River estuary are in the Pacific Coast/Western Pacific Coast region. This region includes coastal Alaska, southwestern British Columbia, Washington, Oregon, California, Baja California, and Sinaloa, Mexico; and interior Washington, Oregon, California, southern Idaho, Montana, Wyoming, western Nevada, and northern Utah.

Pacific Coast Region Overview

Since the beginning of the 20th Century, the Pacific Coast regional population has shifted from nesting in numerous small colonies associated with freshwater marshes in interior California and southern Oregon, to primarily larger colonies along the coast extending into the State of Washington (Gill and Mewaldt 1983). Caspian terns adapt to spatial and temporal variability of breeding habitat and prey, leading to highly variable colony locations and sizes within the region.

In recent years, terns were documented to have nested on about 60 sites scattered throughout the Pacific Coast region, including Alaska. This habitat base serves as a network of sites, which individually may vary in suitability from one year to the next but collectively provide a suite of locations for terns on a regional scale. Colonies in the interior are characteristically small in size and are subject to substantial shifts in location, quantity, and quality corresponding to cycles of flood and drought. Interior sites may also be subject to intensive management such as the control of reservoir and irrigation water. Larger colonies (e.g., many hundreds to thousands of terns) have been documented primarily along the Pacific Coast.

Coastal nesting habitat can be managed or natural and is typically subject to erosion and vegetation changes over time. Although ocean conditions may affect prey availability, coastal prey resources are typically more diverse, abundant, and stable in comparison to prey resources at interior sites which are highly variable from year to year. For a detailed review of current, historic, and potential tern nesting habitat throughout the Pacific Region see: *A Review of Caspian Tern Nesting Habitat: A Feasibility Assessment of Management Opportunities in the U.S. Fish and Wildlife Service Pacific Region* (Seto *et al.* 2003).

Habitat Requirements

Caspian terns nest in single-species colonies or in multi-species assemblages with other ground nesting waterbirds (gulls, skimmers, other terns, and cormorants). Caspian terns breed in a variety of habitats ranging from coastal estuarine, salt marsh, and islands. Terns typically nest in open, barren to sparsely vegetated areas, but also among or adjacent to driftwood, partly buried logs, rocks, or tall annual weeds. Nest substrates vary from sand, gravel, spongy marshy soil, or dead or decaying vegetation to hard soil, shell banks, limestone, or bedrock. Nests range from simple depressions in a bare substrate to nests lined with debris, such as shells, crayfish chelipeds, dried grasses and weed stems, wood, or pebbles.

Diet

Breeding terns eat almost exclusively fish, catching a diverse array of species with shallow plunge dives, usually completely submerging themselves underwater (Cuthbert and Wires 1999). The sizes of fish caught and diet composition are largely determined by geography and annual and seasonal prey availability, but most fish are between 5 to 25 cm and occur near the surface of the water. In the Columbia River estuary, diet studies of the tern colonies on Rice and East Sand islands documented that terns nesting on Rice Island (1999 to 2000) had an average of 83 (77 to 90) percent juvenile salmonids in their diet (Roby et al. 2002), while on East Sand Island (1999 to 2004), terns had an average of 33 (17 to 47) percent juvenile salmonids in their diet (Collis et al. 2002a, 2002b, 2003a, 2003b, K. Collis pers. comm.). From 1999 to 2003, the tern diet on East Sand Island, closer to the mouth of the Columbia River than Rice Island, was primarily non-salmonids, including northern anchovy, herring, shiner perch, sand lance, sculpins, smelt, and flatfish (Roby et al. 2002, Collis et al. 2002b and 2003a). As ocean conditions improved, and thus, ocean productivity, the percentage of juvenile salmonids in the diet of terns in the estuary has continued to decline in recent years.

Salmonid composition at other study sites were found to be variable. For example, in Grays Harbor, Washington, chum and coho salmon were found in the tern diet in low numbers (14 to 21 percent) and primary prey taken were shiner perch and northern anchovy (Penland 1976). At Dungeness NWR, salmonid composition of the tern diet was observed to be the second most important prey species (31 percent of tern diet) in 2004 (Roby et al. 2004). Both of these sites in Washington differ from that observed in Commencement Bay, a location south of Dungeness NWR in Puget Sound, Washington. In 2000, terns in Commencement Bay were observed to have an average of 52 percent salmonids in their diet (Thompson et al. 2002). It is possible that these observed differences in diet composition is because Grays Harbor and Dungeness NWR contain a greater diversity and/or abundance of marine prey species than found in Commencement Bay due to the adjacent marine waters in these two locations.

In San Francisco Bay, diet studies conducted in 2003 and 2004 found that the tern diet varied among the various nesting locations in the bay, but primary prey species included anchovy, surf perch, silversides, herring, sunfish, gobies, and toadfish (Roby et al. 2003a and 2004). In 2003, salmonids (not including trout from reservoirs) were found in the diets of four out of five nesting colonies, ranging from 0.1 (Agua Vista Park and Baumberg Pond) to 8.7 (Knight Island) percent of prey items (Roby et al. 2003a). In 2004, juvenile salmonids were more prevalent in the tern diets, ranging from 1.4 (Agua Vista Park) to 26.1 (Knight Island) percent, and consisted primarily of non-ESA-listed species (Roby et al. 2004). The higher prevalence of salmonids in the tern diet was apparently due to a lower availability of marine fish during that year (e.g., northern anchovy and surfperch, Roby et al. 2004).

In interior Oregon (Summer and Crump lakes), a study conducted in 2003 found tui chubs to be the primary prey of nesting terns (Roby et al. 2003a). In San Diego, food habits of terns were studied in 1995, 1997, and 1998. These studies consistently found terns to feed primarily on sardines, anchovies, and topsmelt (Horn et al. 1996, Horn and Dahdul 1998 and 1999).

Migration

Caspian terns migrate singly or in groups that can be as large as thousands (Shuford and Craig 2002). Most terns congregate for migration at traditional foraging locations along marine coasts and major rivers or freshwater lakes about a month after young have fledged (Shuford and Craig 2002). Timing of migration varies with region; fall movement typically occurs between mid-July and mid-September along the Pacific Coast (Shuford and Craig 2002).

BIOLOGICAL ASSESSMENT

Bald Eagle

Status of the Species

A detailed account of the taxonomy, ecology, and reproductive characteristics of the bald eagle is found in the Pacific States Bald Eagle Recovery Plan (USDI 1986), the final rule to reclassify the bald eagle from endangered to threatened in the 48 contiguous states (USDI 1995), the proposed rule to remove the bald eagle from the Endangered Species List in the 48 contiguous states (USDI 1999), and Stalmaster (1987). History and trends in the status of bald eagle nests in Oregon are tracked annually by Isaacs and Anthony (2002 and 2003), and in Washington by the Washington Department of Fish and Wildlife (WDFW) (WDFW database 2004).

In the Pacific Northwest, bald eagles typically nest in multi-layered, uneven-aged, coniferous stands with old-growth trees that are located within one mile of large bodies of water (Anthony *et al.* 1982). Factors such as tree height, diameter, tree species, and position on the landscape, distance from water, and distance from disturbance appear to influence nest selection. Nest trees usually provide an unobstructed view of the associated water body. Live, mature trees with deformed tops are often selected for nesting. Availability of suitable trees for nesting and perching is critical for maintaining bald eagle populations. Bald eagles often construct several nests within a territory and alternate between them from year to year. Snags, trees with exposed lateral branches, or trees with dead tops are often present in nesting territories and are used for perching or as points of access to and from the nest. Such trees also provide vantage points from which territories can be defended.

The bald eagle was listed as a threatened species in Oregon and Washington under the ESA on February 14, 1978. This status is a result of past and present destruction of habitat, harassment, disturbance, shooting, electrocution, poisoning, a declining food base, and environmental contaminants. Currently, the primary threats to bald eagles are habitat degradation and, in some areas, environmental contaminants.

The Pacific States Bald Eagle Recovery Plan (USDI 1986) established recovery population goals, habitat management goals, and management zones (i.e., Recovery Zones) for a seven-state

Pacific Recovery Region (Recovery Region). It outlined the following criteria for de-listing the bald eagle in the Recovery Region (USDI 1986):

(1) There should be a minimum of 800 pairs nesting in the Recovery Region.
(2) These pairs should be producing an annual average of at least 1.0 fledged young per pair, with an average success rate per occupied territory of not less than 65 percent over a 5-year period.
(3) To ensure an acceptable distribution of nesting pairs, population recovery goals must be met in at least 80 percent of the management zones (i.e., 38 out of 47 Recovery Zones) identified in the Recovery Plan.
(4) Wintering populations should be stable or increasing.

Available information indicates that bald eagle populations are increasing range-wide. The species' status recovered sufficiently to warrant reclassification from endangered to threatened throughout the lower 48 states on July 12, 1995 (USDI 1995); this action did not change the status of the species for Oregon and Washington where it remains listed as threatened. In the Pacific Recovery Region, the number of occupied territories has consistently increased since 1986 and exceeded 800 for five years beginning in 1990 when 861 territories were reported. Although productivity objectives have been met and averaged about 1.03 young per occupied territory since 1990, distribution goals and nesting targets in several Recovery Zones within the Recovery Region have not been met (USDI 1995).

In Oregon, 416 breeding territories were occupied in 2003. Productivity resulted in a 5-year average of 1.01 young per occupied territory. Several Recovery Zones had lower productivity averages below 1.03 young per occupied territory in 2003, indicating that localized regions of poorer reproduction still persist within Oregon. Nesting success resulted in a 5-year average of 64 percent. The net annual population increase was 7.4 percent for 1980-2001, with the average for 2003 at 3.7 percent. It is suggested by Isaacs and Anthony (2002) that population growth may be slowing or survey effort has not detected nesting in new areas. Data gathered during the next two seasons should help determine the trend. Overall, the nesting population continues to grow, and expand into new areas (Isaacs and Anthony 2002).

Of the seven states covered in the Pacific Recovery Area, Washington State supports the largest breeding and wintering populations (USDI 1986). In 2001, 684 nest territories were occupied in Washington (WDFW, 2003, unpub. data). Most nesting territories in Washington are located on the San Juan Islands, along the coastline of the Olympic Peninsula, and along the Straits of Juan de Fuca, Puget Sound, Hood Canal, and the Columbia River. Wintering concentration areas in Washington are along salmon spawning streams and waterfowl wintering areas (Stinson *et al.* 2001).

Abundance and Distribution in the Action Area

Dungeness NWR

Bald eagles use Dungeness Spit year-round. The area is attractive to bald eagles because of its concentration of water birds, seabirds, and anadromous fish species, all of which provide forage for the eagles.

Several active bald eagle nests are known from the action area. Nesting activities typically occur over an extended period from January 1 through August 15. This nesting period overlaps with the time frame that Caspian terns may nest (typically April-July). No bald eagle nesting or perching activity occurs on Dungeness Spit where the existing Caspian tern colony is located. This is attributable to habitat conditions on Dungeness Spit which have precluded establishment of large mature trees for nesting and perching to date. Wintering bald eagles may occur in the vicinity from October 31 through March 31.

East Sand Island

Two bald eagle nest territories occur in the vicinity of East Sand Island around Fort Canby and Cape Disappointment, both 2 miles or more from the proposed project site. These birds are subject to a significant and regular amount of background human activity (boats, barges, ships, etc.) in the action area year-round. Substantial forage resources are available to the bald eagles in this area. Wintering bald eagles may occur in the vicinity from October 31 through March 31.

Fern Ridge Lake

Bald eagles are present at Fern Ridge Lake year-round. A breeding pair nested successfully at Jeans Peninsula (Fern Ridge Lake pair) from 1982 through 1985, producing 5 fledglings in that timeframe (Isaacs and Anthony 2003). Since 1986, the Fern Ridge Lake pair at Jeans Peninsula did not produced any young (Anthony and Isaacs 2003).

Eagles are frequently observed at Fern Ridge and several juveniles and adults are typically present in the winter. The Fern Ridge Wildlife Management Area in the southeast corner of the lake is quite attractive to waterfowl and other waterbirds, providing an available food supply for bald eagles. Fern Ridge Lake also supports a high number of fish, particularly common carp and bullheads. During the winter of 2000/01, as many as 12 adult and sub-adult eagles were observed numerous times perched on shallowly inundated stumps in the lake-bed. Sixty-seven bald eagles were recorded at Fern Ridge Lake in February 2004. This unusual influx of wintering bald eagles was attributed to an abundant food supply comprised of goldfish that had died off due to water temperatures.

Bald eagles also forage at Fern Ridge Lake when the lake is full, and sightings are fairly frequent all along the eastern shoreline. At full pool, fish, waterfowl and other waterbirds continue to provide an adequate food supply for eagles.

Bald eagles would be expected to occur in the vicinity of the proposed nesting island for Caspian terns throughout the year. The construction of the Gibson Island dike within the drawdown zone created a drawdown period shallow reservoir that attracts waterfowl and other waterbirds. When terns are present, however, impounded waters of Fern Ridge Lake would have inundated this shallow reservoir. There are no immediately adjacent perch locations for bald eagles other than osprey nest poles and Gibson Island, which is approximately 2,000 feet distant from the proposed tern island.

Summer Lake Wildlife Management Area

The Big Flat bald eagle pair nested over 1 mile to the west of the proposed project site, however, the nest was lost three years ago (Frank Isaacs, pers. comm. 2004). There are several bald eagle nesting territories located in forested areas on Winter Ridge west of Summer Lake. The closest nest site is several miles from the proposed project site. The Summer Lake WMA provides an extensive area of freshwater marsh habitat and shallow flats. Nesting waterfowl and waterbirds are abundant in the area. Trout and tui chub are available in Ana Reservoir, Ana River and/or Jack's Lake. Big game are present throughout the area along with domestic livestock that provide potential sources of carrion throughout the year.

Wintering bald eagles may occur in the vicinity from October 31 through March 31 with peak numbers in February (Martin St.Louis, ODFW, pers. comm. 2004). Caspian terns are present at this interior basin location by April. There would be little or no overlap of Caspian terns with bald eagles during the late winter timeframe.

Crump Lake

No bald eagle nesting territories are located in the vicinity of the Crump Lake project. No suitable nesting or perching sites are available in the area. Wintering bald eagles may occur in the vicinity of Crump Lake from October 31 through March 31. There would be little or no overlap of Caspian terns with bald eagles during the late winter timeframe. Foraging resources for bald eagles in the Crump Lake area include big game and livestock carrion, waterfowl, various species of waterbirds, and fish, including Warner sucker, brown bullhead, largemouth bass, and white and black crappie.

Effects of the Action

Dungeness NWR

The Service has identified habitat destruction and degradation, human disturbance, and contamination as the major threats to the bald eagle population for the foreseeable future (Stinson *et al.* 2001). For reasons described below, the proposed action at Dungeness Spit for Caspian terns does not entail habitat destruction or degradation that would impact bald eagles, nor will it significantly disrupt normal bald eagle behavior patterns. Human use of refuge lands is controlled by an existing refuge management plan. No contaminants are associated with the proposed action. Nesting, roosting, perching, and/or foraging habitats will not be adversely impacted by the proposed action.

Any increase in the Caspian tern population will be limited by the availability of suitable nesting habitat, avian and mammalian depredation, and by human disturbance. It is estimated that 100 – 3,500 tern pairs would ultimately occupy the Dungeness NWR site (Service *et al.*, 2004b).

Bald eagles appear to be abundant in and around the action area (WDFW 2003, unpubl. Data). Prey resources do not appear to be a limiting factor for the bald eagle population in the project area. Given the diversity and availability of prey species for bald eagles, competition for food resources between bald eagles and Caspian terns is considered minimal. Bald eagles would typically take larger prey (fish) than Caspian terns although there is some overlap projected for fish in the 6-10 inch length class. Based upon the availability and diversity of prey species for bald eagles in the vicinity of the action area, the size and impact of any potential competition for prey species between bald eagles and Caspian terns would likely be insignificant.

Bald eagles have been observed to kleptoparasitize Caspian terns in the Columbia River estuary (G. Dorsey, Corps, pers. obs.). An increase in the Caspian tern population at Dungeness NWR may provide additional seasonal food resources for bald eagles in the Puget Sound area as bald eagles in the Columbia River estuary are known to prey upon Caspian tern adults and young or scavenge their carcasses. Therefore, an increased presence of Caspian terns in the action area may result in a beneficial effect to bald eagles that could substantially offset any effects associated with increased competition for prey between the species.

Human use of refuge lands is controlled by an existing refuge management plan, and no increased use is expected to result from the proposed action. No contaminants are associated with the proposed action. Nesting, roosting, and perching habitats will not be adversely impacted by the proposed action.

Since wintering bald eagles may occur in the vicinity from October 31 through March 31, and Caspian terns typically do not arrive in the area until April and migrate out of the area by September, there would be little or no overlap of Caspian terns with bald eagles during the winter timeframe.

East Sand Island

Bald eagle disturbance and/or avoidance of the island due to construction related activities would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. Barging of equipment to and from the site are also anticipated to result in similar effects. However, the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns.

Given the diversity and availability of prey species for bald eagles, competition for food resources between bald eagles and Caspian terns is considered minimal. Bald eagles would typically take larger prey (fish) than Caspian terns although there is some overlap projected for fish in the 6-10 inch length class. Based upon the availability and diversity of prey species for bald eagles in the vicinity of the action area, the size and impact of any potential competition for prey species between bald eagles and Caspian terns would likely be insignificant.

Caspian terns could also be a prey species for the bald eagle. The decrease in the number of terns nesting in the action area would be considered insignificant to the availability of prey for eagles because of the diversity and abundance of prey species for bald eagles in the Columbia River estuary, and the continued presence of a Caspian tern colony on East Sand Island (estimated to be 2,500 – 3,125 pairs at full implementation).

Since wintering bald eagles may occur in the vicinity from October 31 through March 31, and Caspian terns typically do not arrive in the area until April and migrate out of the area by September, there would be little or no overlap of Caspian terns with bald eagles during the winter timeframe.

Annual operation and maintenance actions associated with the proposed tern nesting island entail minimal human presence. Vegetation management at the island location would take less than one working day to complete; the island surface area is approximately one acre. O&M actions would occur in April through July. O&M would probably entail access via boat. Disking or tilling would be used to manage vegetation. Sand or gravel would be used to enhance tern nesting substrate. Bald eagle disturbance and/or avoidance of the island during these activities would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. However the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns.

The preliminary estimate for the number of Caspian terns occupying this location at full implementation of the proposed action is 2,500 – 3,125 pairs (Service *et al.* 2004b). Research and/or monitoring activities at this location would occur from April to July. Observations would be conducted from blinds to conceal the observers. Access would be via small boat. Researcher/observer disturbance is expected to fall within background limits and is not expected to significantly disrupt normal bald eagle behavior patterns.

Fern Ridge Lake

Construction activities, trucks hauling equipment and materials to the island location and heavy equipment operation at the tern island location may result in periodic disturbance to wintering bald eagles foraging in the general area. The proposed island would be quite distant from the flood control pool (~1.75 miles) where wintering bald eagles concentrate. Waterfowl and waterbirds would be present on the adjacent Gibson Island dike shallow reservoir and the diked marsh restoration areas to the immediate south. The marsh restoration area is greater than 300 acres in extent and waterfowl would be expected to shift away from the northern portions of these diked impoundments during the construction period, thus drawing bald eagles with them. The entire area, including the marsh restoration area is subject to waterfowl hunting and disturbance from hunting efforts to bald eagles would continue to occur during project construction.

Bald eagle disturbance and/or avoidance of the island under construction would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. However, the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns. Fern Ridge Lake and adjacent land holdings, including easement lands, encompasses 12,780 acres. The Fern Ridge Lake Wildlife Management Area (WMA) managed by the ODFW entails more than 5,000 acres in the southeast corner of Fern Ridge Lake. These WMA lands support the bulk of waterfowl resources at Fern Ridge Lake, and in addition to the 1,500 acre minimum flood control pool, represents the principal foraging area for bald eagles. Island construction would be expected to affect waterfowl and waterbird use in approximately 70 acres (12,000 frontage feet by 250 feet interior to dike) of the marsh restoration area. These waterfowl and waterbirds are expected to use other areas within the 5,000-acre WMA and still remain available to bald eagles. No effects to bald eagles perching at Gibson Island are anticipated given the distance to the island from the proposed tern nesting island.

Annual operation and maintenance actions associated with the proposed tern nesting island entail minimal human presence. Vegetation management at the island location would take less than one working day to complete; the island surface area is approximately one acre. O&M actions would occur in late March – early April and possibly in early fall. Spring O&M would probably entail access via boat given that pool elevations would be rising toward full pool elevation at that timeframe. Tillage and/or herbicide (Rodeo) would be used to manage vegetation at that time period. Fall access would be either by boat or vehicle depending upon pool elevation and treatment would be comparable to that used for the Spring period. Bald eagle disturbance and/or avoidance of the island during these activities would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. However, the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns.

It is estimated that 5-300 pairs of Caspian terns could ultimately occupy this location (Service *et al.* 2004b). Should research and/or monitoring activities occur at this location, they would occur from April to July/August. Observations would be conducted from blinds to conceal the observers. Access would be via small boat or canoe from Royal Avenue. Recreational use, via small boats and canoes, including fishing and bird observation already occurs in this area of Fern Ridge Lake. Birders and hikers have access to the dikes in the marsh restoration area to the south of the proposed tern island. Thus, researcher/observer disturbance is expected to fall within background limits and is not expected to significantly disrupt normal bald eagle behavior patterns.

Since wintering bald eagles may occur in the vicinity from October 31 through March 31, and Caspian terns typically do not arrive in the area until April and migrate out of the area by September, there would be little or no overlap of Caspian terns with bald eagles during the winter timeframe.

Summer Lake Wildlife Management Area

The Big Flat bald eagle pair nested over 1 mile to the west of the proposed project site. The nest was lost three years ago (Frank Isaacs, pers. comm. 2004). The bald eagles nesting around Winter Ridge are several miles from the proposed project site.

Bald eagle disturbance and/or avoidance of the island under construction would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. However, given the abundant and diverse sources of forage available to the bald eagle in the Summer Lakes area, the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns. Furthermore, construction of three 0.5 acre islands in diked impoundments, would not adversely impact bald eagle habitat, as it would result in an insignificant loss of forage base for the eagle. No effects to nesting bald eagles are expected at this location.

Forage competition for fish prey between nesting bald eagles and Caspian terns would be considered insignificant due to the substantial forage base available to bald eagles in the area, and the significant (spatial) distance between the tern colony and the nearest nesting eagles and foraging perches. Forage competition for fish prey between overwintering bald eagles and Caspian terns would be considered insignificant, due to the minimal, if any, temporal overlap and the abundance of bald eagle prey items in the Summer Lake area.

Annual operation and maintenance actions associated with the proposed tern nesting island entail minimal human presence. Vegetation management at the island location would take less than one working day to complete; the island surface area is approximately one acre. O&M actions would occur in late March – early April and possibly in early fall. Spring O&M would probably entail access via boat given that pool elevations would be rising toward full pool elevation at that timeframe. Tillage and/or herbicide (Rodeo) would be used to manage vegetation at that time period. Fall access would be either by boat or vehicle depending upon pool elevation and treatment would be comparable to that used for the Spring period. Bald eagle disturbance and/or avoidance of the island during these activities would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. However, the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns.

Research actions, if implemented, to monitor and evaluate Caspian terns at the Summer Lake WMA would not significantly disrupt normal bald eagle behavior patterns, because of the limited nature (spatially) of the research actions, the use of blinds, and the extensive area of available foraging habitat for bald eagles.

Crump Lake

No effects to nesting bald eagles is anticipated as nests or suitable nest sites are not known from the action area. Wintering bald eagles may occur in the vicinity of Crump Lake from October 31 through March 31. The proposed construction window falls outside of this timeframe, thus, no effects to wintering bald eagles are anticipated from the proposed action. There would be little or no overlap of Caspian terns with bald eagles during the late winter timeframe, and no suitable perching sites are available in the area. Foraging resources for bald eagles in the Crump Lake area are substantial.

Annual operation and maintenance actions associated with the proposed tern nesting island entail minimal human presence. Vegetation management at the island location would take less than one working day to complete; the island surface area is approximately one acre. O&M actions would occur in late March – early April and possibly in early fall. Spring O&M would probably entail access via boat given that pool elevations would be rising toward full pool elevation at that timeframe. Tillage and/or herbicide (Rodeo) would be used to manage vegetation at that time period. Fall access would be either by boat or vehicle depending upon pool elevation and treatment would be comparable to that used for the Spring period. Bald eagle disturbance and/or avoidance of the island during these activities would be minimal in nature, resulting in affected bald eagles shifting their foraging areas slightly. However, the likelihood of displacement and the magnitude of such displacement are not expected to significantly disrupt normal bald eagle behavior patterns.

Research actions, if implemented, to monitor and evaluate Caspian terns at Crump Lake would occur during April through August. Since no nesting bald eagle territories are known in this area, no effects are expected from research activities.

Conclusion

The proposed action may effect, but will not likely adversely affect, bald eagles at Dungeness NWR, East Sand Island, Fern Ridge Lake, Summer Lake Wildlife Management Area, or Crump Lake. We make this determination for the following reasons: (1) proposed construction activities will not result in destruction or degradation of bald eagle habitat, nor will they significantly disrupt normal bald eagle behavior patterns; (2) forage competition for fish prey between nesting bald eagles and Caspian terns would be considered insignificant due to the substantial forage base available to bald eagles in the area, and the significant (spatial) distance between the tern colony and the nearest nesting eagles and foraging perches; (3) forage competition for fish prey between overwintering bald eagles and Caspian terns would be considered absent, due to the lack of temporal overlap in presence of both species; (4) the likelihood and magnitude of displacement resulting from annual maintenance activities are not expected to significantly disrupt normal bald eagle behavior patterns; and (5) research actions to monitor and evaluate Caspian terns in the action area would not significantly disrupt normal bald eagle behavior patterns because of the limited nature (spatially) of the research actions, the use of blinds, and the extensive area of available foraging habitat for bald eagles.

Bull Trout

Status of the Species

On November 1, 1999, the Service (USDI 1999) listed five distinct populations segments (DPSs) of bull trout within the coterminous United States as threatened: (1) Coastal-Puget Sound DPS, (2) Columbia River DPS, (3) Jarbidge River DPS, (4) St. Mary-Belly River DPS, and (5) Klamath River DPS. Factors contributing to the decline of bull trout populations identified in the listing rule include restriction of migratory routes by dams and other unnatural barriers; forest management, grazing, and agricultural practices; road construction; mining; introduction of nonnative species; and residential development resulting in adverse habitat modification, overharvest, and poaching (Bond 1992, Thomas 1992, Rieman and McIntyre 1993, Donald and

Alger 1993, WDFW 1997). Critical habitat for the Coastal-Puget Sound DPS was proposed by the Service on June 25, 2004 (USDI 2004).

Life History

Bull trout and Dolly Varden occur together only within the area of the Coastal-Puget Sound DPS and in British Columbia, Canada. Although these two species of native char were previously considered a single species, based on distribution, body type, and measurements bull trout and the Dolly Varden were formally recognized as two separate species in 1980 (Cavender 1978, Robins *et al.* 1980, Bond 1992). In the Coastal-Puget Sound DPS, Dolly Varden tend to be generally distributed in isolated tributary populations above natural anadromous barriers (as in the Dungeness and Nooksack core areas), while bull trout are generally distributed below these barriers (WDFW 1998, Spruell and Maxwell 2002). All anadromous char sampled have been genetically identified as bull trout.

Bull trout in the Coastal-Puget Sound DPS demonstrate all known migratory life history patterns (i.e., anadromous, adfluvial, and fluvial) for this species, as well as nonmigratory, or resident, life history patterns. These diverse life history types are important to the stability and viability of bull trout populations (Rieman and McIntyre 1993).

Within the range of bull trout in the coterminous United States, anadromy, or technically "amphidromy," is unique to the Coastal-Puget Sound DPS. Unlike strict anadromy, amphidromous individuals often return seasonally to freshwater as subadults, sometimes for several years, before returning to spawn (Wilson 1997). Subadult bull trout in the Coastal-Puget Sound DPS can move into marine waters to forage or migrate and return to freshwater to take advantage of seasonal forage provided by salmonids eggs, smolts, or juveniles.

Migratory forms appear to develop when habitat conditions allow movement between spawning and rearing streams and larger rivers and lakes and marine and estuarine waters where foraging opportunities are enhanced (Kraemer 1994; Frissell 1999). Benefits to migratory bull trout include greater growth in the more productive waters of larger streams, lakes, estuaries, and nearshore marine areas; greater fecundity resulting in increased reproductive potential; and dispersal of the population across space and time.

Anadromous and fluvial life history forms typically have widely distributed foraging, migration, and overwintering habitat. Migratory bull trout use nonnatal (habitat outside of their spawning and early rearing habitat) watersheds to forage, migrate, and overwinter (Brenkman and Corbett, *in litt.* 2003a, b). Larger juvenile and subadult bull trout can migrate throughout a core area looking for feeding opportunities, or they can move through marine areas to access independent tributaries (tributaries that connect directly to marine waters) to forage or, potentially, to take refuge from high flows in their core areas (Brenkman and Corbett, *in litt.* 2003a, b). Independent tributaries used by bull trout, such as Kalaloch Creek and Raft River, are not believed to support spawning populations of bull trout and are only accessible to bull trout by swimming through marine waters from core areas.

Within the Coastal-Puget Sound DPS, current bull trout distribution has been reduced from historic distribution, most notably in the Hood Canal, and Satsop, Green, and Nisqually Rivers, and population abundance has significantly decreased throughout much of the DPS (USDI 1999). Highly migratory life history forms, such as fluvial and anadromous fish, have been eliminated from many large, productive river systems in the DPS. Isolated, remnant populations, which lack connectivity to migratory populations, often have a low likelihood of persistence because of reduced access to prey and reduced opportunities for recolonization (Rieman and McIntyre 1993, Rieman and Allendorf 2001).

For all life history types, the juveniles typically rear in tributary streams for 1 to 3 years before migrating downstream into a larger river, lake, or estuary and/or nearshore marine area to mature (Rieman and McIntyre 1993). Bull trout become sexually mature between 4 and 9 years of age and may spawn in consecutive or alternate years (Shepard *et al.* 1984, Pratt 1992). In the Skagit River system, length to age correlations are (1) age 3 subadults are generally 200-250 mm; (2) first year spawners (age 5) are generally 400-450 mm; and (3) mature spawners (age 6 to 7) are generally 550-600 mm (Ed Conner, pers. comm. as cited in Beamer and Henderson 2004). Spawning typically occurs from August through December in cold, low-gradient 1^{st}- to 5^{th}-order tributary streams, over loosely compacted gravel and cobble having groundwater inflow (Shepard *et al.* 1984, Brown 1992, Rieman and McIntyre 1996, Swanberg 1997, MBTSG 1998, Baxter and Hauer 2000). Migratory bull trout may begin their spawning migrations as early as April and have been known to migrate upstream as far as 250 kilometers (km) (155 miles [mi]) to spawning grounds (Fraley and Shepard 1989). Hatching occurs in winter or early spring, and alevins may stay in the gravel for up to three weeks before emerging from the gravel.

Bull trout are apex predators and require a large prey base and home range. Adult and subadult migratory bull trout are primarily piscivorous, feeding on various trout and salmon species, whitefish, yellow perch (*Perca flavescens*), sculpin, and in the case of anadromous forms, surf smelt (*Hypomesus pretiosus*) and sandlance (*Ammodytes hexapterus*). Resident and juvenile bull trout prey on terrestrial and aquatic insects, macrozooplankton, amphipods, mysids, crayfish, salamanders, and small fish (Wyman 1975, Rieman and Lukens 1979 in Rieman and McIntyre 1993, Boag 1987, Goetz 1989, Donald and Alger 1993, Connor *et al.* 1997).

Habitat Requirements

Bull trout have more specific habitat requirements than other salmonids (Rieman and McIntyre 1993). Growth, survival, and long-term persistence are dependent on the following habitat characteristics: cold water, complex instream habitat, a stable substrate with a low percentage of fine sediments, high channel stability, and stream/population connectivity. Stream temperature and substrate type, in particular, are critical factors for the sustained long-term persistence of bull trout. Spawning is often associated with the coldest, cleanest, and most complex stream reaches within basins. However, bull trout exhibit a patchy distribution even in pristine habitats (Rieman and McIntyre 1995) and should not be expected to occupy all available habitats at the same time (Rieman *et al.* 1997). While bull trout clearly prefer cold waters and nearly pristine habitat, it cannot be assumed that they do not occur in streams where habitat is degraded.

For long-term persistence, bull trout populations need a stream temperature regime that ensures sufficient amounts of cold water are present at the locations and during the times needed to complete their life cycle. Temperature is most frequently recognized as the factor limiting bull trout distribution (Dunham and Chandler 2001, Rieman and McIntyre 1993). Probability of occurrence for juvenile bull trout in Washington is relatively high (75 percent) when maximum daily temperatures did not exceed approximately 11 to 12° C (52 to 54° F) (Dunham *et al.* 2001). Optimum incubation temperatures range from 2 to 6° C (36 to 43° F). At 8 to 10° C (46 to 50° F), survival ranged from 0-20 percent (McPhail and Murray 1979). Tributary stream temperature requirements for rearing juvenile bull trout are also quite low, ranging from 6 to 10° C (43 to 50° F) (Buchanan and Gregory 1997, Goetz 1989, Pratt 1992, McPhail and Murray 1979).

Increases in stream temperatures can cause direct mortality, increased susceptibility to disease or other sublethal effects, displacement by avoidance (McCullough *et al.* 2001, Bonneau and Scarnechia 1996), or increased competition with species more tolerant of warm stream temperatures (Rieman and McIntyre 1993, Craig and Wissmar 1993 cited in USDI (1997), MBTSG 1998). Brook trout, which can hybridize with bull trout, may be more competitive than bull trout and displace them, especially in degraded drainages containing fine sediment and higher water temperatures (Clancy 1993, Leary *et al.* 1993). Recent laboratory studies suggest bull trout are at a particular competitive disadvantage in competition with brook trout at temperatures greater than 12° C (54° F) (McMahon *et al.* 2001).

When bull trout migrate through stream segments with higher water temperatures they tend to seek areas offering thermal refuge such as confluences with cold tributaries (Swanberg 1997), deep pools, or locations with surface and groundwater exchanges in alluvial hyporheic zones (Frissell 1999). Water temperatures above 15° C are believed to limit bull trout distribution, which partially explains their generally patchy distribution within a watershed (Fraley and Shepard 1989, Rieman and McIntyre 1995).

Bull trout show a strong affinity for stream bottoms and a preference for deep pools in cold water streams (Goetz 1989, Pratt 1992). Stream bottom and substrate composition are highly important for juvenile rearing and spawning site selection (Rieman and McIntyre 1993, Graham *et al.* 1981, McPhail and Murray 1979). Bull trout of all age classes are closely associated with cover, especially during the day (Baxter and McPhail 1997, Fraley and Shepard 1989). Cover may be in the form of overhanging banks, deep pools, turbulence, large wood, or debris jams. Young bull trout use interstitial spaces in the substrate for cover and are closely associated with the stream bed. This association appears to be more important for bull trout than for other salmonid species (Pratt 1992, Rieman and McIntyre 1993). Large wood in streams enhances the quality of habitat for salmonids and contributes to channel stability (Bisson *et al.* 1987).

Due to the bull trout's close association with the substrate, bed load movements and channel instability can reduce the survival of young bull trout. Maintaining bull trout habitat requires stream channel and flow stability (Rieman and McIntyre 1993). Bull trout are exceptionally sensitive to activities that directly or indirectly affect stream channel integrity. Channel dewatering caused by low flows and bed aggradation has blocked access for spawning fish resulting in year-class failures (Weaver 1992). Timber harvest and the associated roads may cause landslides that affect many miles of stream through aggradation of the streambed.

Migration

The importance of maintaining the migratory life history form of bull trout, as well as migratory runs of other salmonids that may provide a forage base for bull trout, is repeatedly emphasized in the scientific literature (Rieman and McIntyre 1993, MBTSG 1998, Dunham and Rieman 1999, Nelson *et al.* 2002). Isolation and habitat fragmentation resulting from migratory barriers have negatively affected bull trout by (1) reducing geographical distribution (Rieman and McIntyre 1993, MBTSG 1998); (2) increasing the probability of losing individual local populations (Rieman and McIntyre 1993, MBTSG 1998, Nelson *et al.* 2002, Dunham and Rieman 1999); (3) increasing the probability of hybridization with introduced brook trout (Rieman and McIntyre 1993); (4) reducing the potential for movements in response to developmental, foraging, and seasonal habitat requirements (MBTSG 1998, Rieman and McIntyre 1993); and (5) reducing reproductive capability by eliminating the larger, more fecund migratory form from many subpopulations (MBTSG 1998, Rieman and McIntyre 1993). Unfortunately, migratory bull trout have been restricted or eliminated in parts of their range due to stream habitat alterations, including seasonal or permanent obstructions, detrimental changes in water quality, increased temperatures, and the alteration of natural streamflow patterns. Dam and reservoir construction and operations have altered major portions of bull trout habitat in the Skokomish, Elwha, Skagit, Nooksack, and Puyallup core areas.

The estuaries and shoreline areas comprise what is known as the nearshore marine habitat. This nearshore environment supports habitat important to both bull trout and salmon. This habitat provides food production and foraging areas, refuge (from predation, seasonal high flows, winter storms, etc.), and migratory corridors. Bull trout first migrate to tidal areas between age 1 and 3. These juvenile fish may rear in the tidally influenced delta within intertidal marsh, distributary channels, or may pass through into nearshore marine areas. Additional information provided by bull trout acoustic radio telemetry and habitat study projects indicates that bull trout in marine waters are more active at night than during the day, may prefer deeper nearshore habitat than shallow nearshore habitat, can be found at depths as great as 60 to 75 meters, and that bull trout from different freshwater populations may overlap in their use of marine and estuarine waters. Although bull trout are likely to be found in nearshore marine waters year round, the period of greatest use is March through July (Goetz and Jeanes 2004).

In Puget Sound the distribution of bull trout in the nearshore waters has been hypothesized to be correlated to the nearshore distribution of baitfish (Kraemer 1994). It also appears that certain life history stages may utilize different marine prey species. For example, the younger bull trout (age 1-3) that move to marine waters appear to select smaller prey items, such as shrimp. By age 4, the diet of anadromous bull trout has shifted largely to fish. Bull trout from Puget Sound prey on surf smelt, Pacific herring, Pacific sand lance, pink salmon smolts, chum salmon smolts, and a number of invertebrates (Kraemer 1994).

These nearshore marine habitats have been significantly altered by human development (PSWQAT 2000). Construction of bulkheads and other structures have modified the nearshore areas and resulted in habitat loss that has directly affected forage fish for bull trout. Other impacts to the marine environment include alterations to water quality resulting from fish pathogens, nutrients and toxic contaminants, urbanization, and stormwater runoff from basins that feed Puget Sound.

Changes in Status of the Coastal-Puget Sound DPS

Although the status of bull trout in Coastal-Puget Sound DPS has been improved by certain actions, it continues to be degraded by other actions, and it is likely that the overall status of the bull trout in this DPS has not improved since its listing on November 1, 1999. Improvement has occurred largely through changes in fishing regulations and habitat restoration projects. On the other hand, the status of this DPS has been affected by a number of federal and non-federal actions, each having various levels of adverse and beneficial impacts.

Environmental Baseline

Prior to this year, the status of the Dungeness River bull trout population was unknown, due mainly to difficulties in access. However, based on extensive bull trout redd surveys in the Dungeness River this year (Chan, unpublished Service report, 2004), this population may be depressed, based on very low redd counts. It is thought that the 2004 surveys located approximately 90 percent of the redds, because other salmonid species were absent this year that would be otherwise building redds at the same time, so there was little chance of miscounting bull trout redds.

In 2004, in a three mile stretch of lower Dungeness River (presumably the only portion of the river suitable for foraging by terns), there were 73 bull trout encountered in a size range of 0-500mm (0-50 cm). More than half of those encountered were in the smaller size range (large juveniles) that could presumably be captured by Caspian terns (Shelley Spalding, Service biologist, pers. comm. 2004). The bull trout occupied areas to the east (Bell) and west (Morse) of Dungeness Spit, and nearshore marine habitat, are considered to support only foraging, migrating, and overwintering populations of bull trout, and are unlikely to support the smaller fish (Shelley Spalding, Service biologist, pers. comm.2004). There could be some small amount of use by smaller bull trout in this area, but this use is expected to be uncommon or significant.

Effects of the Action

It is difficult to estimate the number of bull trout that could be consumed by Caspian terns. Diet studies for the existing Caspian tern colony have not specifically identified bull trout in the tern diet. Instead an overall salmonid category (which includes bull trout) was used in a study conducted in 2004; an estimate of 31% of the tern's diet was comprised of salmonids (Roby *et al.* 2004). Foraging, migrating and overwintering bull trout known to occur in the action area are expected to be too large for Caspian terns to capture as prey. The fish count for the lower Dungeness River was 73, which was estimated to be 90 percent of the population (Chan, unpublished Service report, 2004). More than half of the bull trout known from the lower Dungeness River estuary are in a size class that would be considered exposed to predation by

Caspian terns (Shelley Spalding, Service, pers. comm. 2004). Bull trout are known to exhibit several behaviors that likely reduce this exposure risk to predation.

As discussed in the Status of the Species section above, migrating bull trout tend to seek out thermal refuge in lower stream segments with higher water temperatures. Bull trout show a strong affinity for stream bottoms and a preference for deep pools in cold water streams, are closely associated with cover (overhanging banks, deep pools, turbulence, large wood, or debris jams), especially during the day, and generally exhibit a patchy distribution within a watershed. Young bull trout are known to use interstitial spaces in the substrate for cover and are closely associated with the stream bed. In nearshore waters bull trout have exhibited a tendency to be more active at night when terns would not be feeding, and may prefer deeper nearshore habitat than shallow nearshore habitat. We expect capture of bull trout by Caspian terns are likely to be extremely rare for the following reasons: (1) young bull trout of the size that could be captured by terns spend most of their time under cover or otherwise exhibit cryptic behavior; (2) young bull trout are patchily distributed in the system and rarely would be expected to enter marine waters; and (3) bull trout have not been observed to be a major forage base of the Caspian tern (tern diet is composed primarily of surfperch and salmonids in general, and various other fish captured in the marine environment.

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. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act. We are unaware of any non-Federal actions affecting bull trout that are reasonably certain to occur in the action area considered by this opinion.

Conclusion

The proposed action is likely to adversely affect bull trout at Dungeness NWR. Based on the above analysis, we believe an incremental, though slight, increase in capture of bull trout by Caspian terns is likely to occur in the action area. Therefore, the action is likely to result in unavoidable adverse affects to bull trout in the form of injury, capture or kill.

Marbled Murrelet

Status of the Species

The marbled murrelet was federally listed as a threatened species in Washington, Oregon, and northern California effective September 28, 1992 (USDI 1992). The final rule designating critical habitat for the murrelet (61 FR 26256; USDI 1996) became effective on June 24, 1996. The species' decline has largely been caused by extensive removal of late-successional and old-growth coastal forests which serve as nesting habitat for murrelets. Additional listing factors included high nest-site predation rates and human-induced mortality in the marine environment from gillnets and oil spills.

The Service recently determined that the California, Oregon, and Washington distinct population segment of the marbled murrelet does not meet the criteria set forth in the Service's 1996 Distinct Population Segment policy (61 FR 4722; USDI 2004). Notwithstanding that finding, the marbled murrelet retains its listing status and receives full protection under the ESA until such time that the original 1992 listing decision is revised through formal rule-making procedures, involving public notice and comment.

The general ecology and recovery needs of the murrelet are addressed below. More detailed discussions are presented in the final rule listing the murrelet as threatened (USDI 1992), Ecology and Conservation of the Marbled Murrelet (Ralph *et al.* 1995), the final rule designating murrelet critical habitat (USDI 1996), the Marbled Murrelet Recovery Plan (USDI 1997), and the evaluation report in the 5-year status review (McShane *et al.* 2004).

Distribution

The range of the murrelet, defined by breeding and wintering areas, extends from the northern terminus of Bristol Bay, Alaska to the southern terminus of Monterey Bay in central California. The listed portion of the species' range extends from the Canadian border south to central California. Murrelet abundance and distribution has been significantly reduced in portions of the listed range, and the species has been extirpated from some locations. The areas of greatest concern due to small numbers and fragmented distribution include portions of central California, northwestern Oregon, and southwestern Washington (USDI 1997).

Life History

Below are some of the more salient aspects of the life history of the murrelet that influence current threats and management. Detailed discussions of the biology and status of the murrelet are presented in the final rule listing the murrelet as threatened (USDI 1992), the Marbled Murrelet Recovery Plan (USDI 1997), Ecology and Conservation of the Marbled Murrelet (Ralph *et al.* 1995), and the final rule designating marbled murrelet critical habitat (USDI 1996).

Murrelets are dependent upon old-growth forests, or forests with an older tree component, for nesting habitat (McShane *et al.* 2004, Hamer and Nelson 1995, Ralph *et al.* 1995). Murrelets generally select nests within 37 miles (60 kilometers (km)) of the coast (Lank *et al.* 2003 *in* McShane *et al.* 2004). Sites occupied by murrelets tend to have a higher proportion of mature forest age-classes than do unoccupied sites (Raphael *et al.* 1995). Murrelets are long-lived, and have high fidelity to nesting areas, but not necessarily specific nest trees or specific branches within a tree crown. They require a sufficiently wide and flat space to retain a single egg.

Adults rotate incubation roles, with exchanges typically occurring under low light conditions at dusk or dawn. Following hatching, adults leave the nest and forage at sea and return daily to feed the chick, again most frequently at dawn or dusk. Nest success appears to be quite low due to predation on eggs and chicks. Nests located near abrupt forest edges appear to be much more susceptible to predation, especially by corvids (Nelson and Hamer 1995, Paton 1994).

When tending active nests during the breeding season (and much of the non-breeding season in southern parts of the range), adult murrelets are restricted to foraging within commuting distance from the nest site. Daily movements of breeding adults between nest sites and foraging areas averaged 10 miles in Prince William Sound, Alaska (McShane *et al.* 2004) and 24 miles in Desolation Sound, British Columbia (Hull *et al.* 2001).

By the end of August, over 90 percent of chicks have fledged (Hammer in litt.). During the non-breeding season, murrelets disperse and can be found farther from shore (Strachan *et al.* 1995). However, little is known about marine-habitat preference during the early spring and fall, but use is thought to be similar to that preferred during the breeding season (Nelson 1997). There is little known about winter use of marine habitats, but there may be a general shift from exposed outer coasts into more protected waters (Nelson 1997). In many areas, murrelets remain associated with the inland nesting habitat during the winter months (Carter and Erickson 1992).

A complete pre-basic molt occurs from mid-July through December (Carter and Stein 1995, Nelson 1997). During the pre-basic molt, murrelets lose all flight feathers somewhat synchronously and are flightless for up to two months (Nelson 1997). In Washington, there is some indication that the pre-basic molt occurs from mid-July through the end of August (Thompson, pers. comm. 2003).

Foraging

Murrelets typically forage in pairs, but have been observed to forage alone or in groups of three or more (Carter and Sealy 1990, Strachan *et al.* 1995, Speckman *et al.* 2003). Paired foraging is common throughout the year, even during the incubation period, suggesting that breeding murrelets may temporarily pair up with other foraging individuals (non-mates) (Strachan *et al.* 1995, Speckman *et al.* 2003).

During the breeding season, murrelets generally forage in shallow waters within 1.25 miles of shore (Strachan *et al.* 1995). Traditional feeding areas (nurseries) are used consistently on a daily and yearly basis (Carter and Sealy 1990). The areas of highest foraging use typically are associated with up-welling areas such as at bay entrances, over underwater sills, tidal rips, narrow passages between islands, shallow banks, and kelp (*Nereocystis* spp.) beds (Ainley *et al.* 1995, Burger 1995, Strong *et al.* 1995, Nelson 1997). Activity patterns and foraging locations are influenced by factors that affect prey availability, such as weather, climate, time of day, season, and light intensity (Speckman 1996).

Juveniles are found closer to shore than adults (rarely greater than 0.625 miles offshore) (Beissinger 1995) and forage without the assistance of adults (Strachan *et al.* 1995). Kuletz and Piatt (1999) found that in Alaska, juvenile marbled murrelets congregated in kelp beds. Kelp beds are often associated with productive waters and may provide protection from avian predators (Kuletz and Piatt 1999). McAllister (unpublished data cited in Strachan *et al.* 1995) found that juveniles were more abundant around bull kelp within 328 feet of shorelines.

Adults and subadults often move away from breeding areas prior to molting and must select areas with predictable prey resources during the flightless period (Carter and Stein 1995, Nelson

1997). In areas where murrelets are not associated with nesting habitat during the winter, adults potentially move great distances. Many marbled murrelets breeding on the exposed outer coast of Vancouver Island appear to congregate in more sheltered waters within the Puget Sound and the Strait of Georgia in the fall and winter (Burger 1995). Murrelets throughout the listed range demonstrate resident movement behavior by not dispersing long distances (McShane *et al.* 2004).

Throughout their range, murrelets are opportunistic feeders and utilize prey of diverse sizes and species. In general, small schooling fish and large pelagic crustaceans are the main prey items. Pacific sand lance *(Ammodytes hexapterus)*, northern anchovy *(Engraulis mordax)*, immature Pacific herring *(Clupea harengus)*, capelin *(Mallotus villosus)*, and smelt (Osmeridae) are the most common fish species taken and are eaten year round. Squid (*Loligo* spp.), euphausiids, mysid shrimp, and large pelagic amphipods are the main invertebrate prey and are primarily eaten during the non-breeding season, but are seldom fed to chicks.

Murrelets usually carry a single fish to their chicks and appear to select a relatively large (relative to the size of the young), energy-dense fish, typically larger sand lance, immature herring, anchovy, smelt and occasionally salmon smolts (Burkett 1995, Nelson 1997). This forces breeding adults to exercise more specific foraging strategies when feeding chicks. As a result, the distribution and abundance of prey suitable for feeding chicks may greatly influence the overall foraging behavior and location(s) during the nesting season. The availability of abundant forage fish during the nestling period may significantly affect the energy demand on adults by influencing both foraging time and number of trips inland required to feed nestlings (USDI 1992).

Murrelets forage at all times of the day, but most actively in the morning and late afternoon (Strachan *et al.* 1995). Some foraging occurs at night (Strachan *et al.* 1995). Speckman *et al.* (2000) observed the abundance of murrelets in Alaska to peak in the morning and in locations where Pacific sand lance were abundant and coincided with high or falling morning tides.

Murrelets forage more frequently in nearshore water generally less than 98 feet (30 m) deep (Strachan *et al.* 1995, Burger 2000). The most common foraging depths are not known. Murrelets typically feed on small schools of fish within 16.4 feet (5 m) of marine waters (Mahon *et al.* 1992), with an estimated maximum diving depth of about 154 feet (47 m) (Mathews and Burger 1998). The deepest dive recorded for murrelets occurred at 89 feet (27 m) in a gill net off of California (Carter and Erickson 1992). Jodice and Collopy (1999) reported typical foraging in Oregon occurs in water less than 33 feet (10 m) deep.

Threats

Threats to murrelets in the marine environment include declines in prey availability, mortality associated with exposure to oil spills, gill net and other fisheries, contaminants suspended in marine waters, and visual or sound disturbance from recreational or commercial watercrafts (USDI 1992, Ralph *et al.* 1995, USDI 1997, McShane *et al.* 2004). However, the consequences of these threats are difficult to quantify.

Many fish populations have been depleted due to overfishing, reduction in the amount or quality of spawning habitat, and pollution. Primary murrelet prey species have little commercial fishery value and, in general, there is little geographic overlap between murrelet distribution and areas of commercial harvest (McShane *et al.* 2004). However, there are several fisheries for herring and surf smelt in Puget Sound and for anchovy in Grays Harbor, Willapa Bay, and along the outer coast (Bargmann 1998). The extent of the effects of these fisheries on murrelets is unknown. In addition to fishing pressure, oceanographic variation can influence prey availability.

Murrelet mortality from oil pollution is a significant conservation issue in Washington (USDI 1997). Most oil spills and chronic oil pollution that can affect murrelets occurs in areas of high shipping traffic, such as the Strait of Juan de Fuca and Puget Sound. However, the estimated annual mortality of murrelets from oil spills in Washington has decreased from 3 to 41 birds per year (1977 and 1992) to 1 to 2 birds per year (1993 and 2003) (McShane *et al.* 2004).

Murrelet mortality from gillnet fishing has been considered a significant conservation issue in Washington (USDI 1997, Melvin *et al.* 1999). Murrelets can also be killed by hooking with fishing lures and entanglement with fishing lines (Carter *et al.* 1995). There is little information available on murrelet mortality from net fishing prior to the 1990s, although it was known to occur (Carter *et al.* 1995). In the mid 1990s, a series of fisheries restrictions and changes were implemented to address mortality of all species of seabirds, resulting in a lower mortality rate of murrelets (McShane *et al.* 2004). In most areas, the threat from gill net fishing has been reduced since 1992, but is still present in Washington nearshore zones (McShane *et al.* 2004).

The primary consequence from the exposure of murrelets to contaminants is reproductive impairment. Reproduction can be impacted by food web bioaccumulation of organochlorine pollutants and heavy metals discharged into marine areas where murrelets feed and prey species concentrate (Fry 1995). However, murrelet exposure is likely a rare event because murrelets have widely dispersed foraging areas and they feed extensively on transient juvenile and subadult midwater fish species that are expected to have low pollutant loads (McShane *et al.* 2004). The greatest risk to murrelets may be regularly feeding near major pollutant sources, such as in Puget Sound (McShane *et al.* 2004).

In coastal and offshore marine environments, vehicular disturbance (e.g., boats, airplanes, personal watercraft) is known to elicit behavioral responses in marbled murrelets of all age classes (Kuletz 1996, Speckman 1996, Nelson 1997). Aircraft flying at low altitudes and boating activity, in particular motorized watercraft, are know to cause birds to dive and are thought to especially affect adults holding fish (Nelson 1997). It is unclear to what extent this kind of disturbance affects the distribution and movements of murrelets. However, it is unlikely this kind of disturbance has decreased since 1992 because the shipping traffic and recreational boat use in the Puget Sound and Strait of Juan de Fuca has continued to increase.

Population Size and Trends

Since the murrelet's listing in 1992, there has been more research on population size, trends and threats. Research in the early 1990s estimated the size of the murrelet population in Washington, Oregon, and California at 18,550 to 32,000 (Ralph *et al.* 1995b). In the early 2000s, the population estimate was 17,700-29,700, with 50 percent of the population occurring in Washington State (Huff *et al.* 2003).

The best available population trend data is derived from demographic modeling by Beissinger 1995, Beissinger and Nur 1997 (cited in USDI 1997a), Cam *et al.* 2003, and McShane *et al.* 2004. Rangewide population declines have been estimated to range from 2 to 15.8 percent (Beissinger and Nur 1997, Beissinger *et al.* 2003, McShane *et al.* 2004). A downward trend of this magnitude indicates the population could be less than three-quarters of its initial population size in 25 years. Unless additional research demonstrates the population survival or fecundity rates are actually higher than estimated or threats to marbled murrelets decrease, extirpation may occur in the future.

Habitat Management

The recovery strategy for the murrelet (USDI 1997) relies on the Northwest Forest Plan (NWFP) as the primary mechanism to achieve recovery on Federal land in Washington, Oregon, and California. The Recovery Plan also addresses the role of non-Federal lands in recovery, including Habitat Conservation Plans (HCPs), state forest practices, and Tribal lands owned by Native American Tribes. The importance of non-Federal lands in the survival and recovery of murrelets is particularly high in recovery zones where Federal lands within 50 miles of the coastline are lacking.

In the short-term, all known-occupied sites of murrelets occurring on U.S. Forest Service (USFS) or Bureau of Land Management (BLM) lands under the NWFP are to be managed as Late Successional Reserves (LSRs). In the long-term, unsuitable or marginally suitable habitat occurring in LSRs will be managed, overall, to develop late-successional forest conditions, thereby providing a larger long-term habitat base into which murrelets may eventually expand. Thus, the NWFP approach offers both short-term and long-term benefits to the murrelet. Scientists predicted implementation of the NWFP would result in an 80 percent likelihood of achieving a well-distributed murrelet population on Federal lands over the next 100 years (USDA and USDI 1994).

Ten HCPs have been completed for private/corporate forest land managers in Washington State and four address murrelets within Washington. Overall in Washington, HCPs affecting murrelets cover approximately 500,000 acres of non-Federal (private/corporate) lands, over 100,000 acres of municipal watersheds, and over 1.6 million acres of State-managed lands. A detailed description of these HCPs was presented in USDI (2002a:68-71).

Under Washington Forest Practices Rules, which apply to all non-Federal lands not covered by a HCP (WFPB 2001), surveys for murrelets are required for harvest of habitat that meets certain platform numbers and stand size criteria. The management strategy of the Bureau of Indian Affairs (BIA) for the murrelet focuses on working with Tribal governments on a government to government basis to develop management strategies for reservation lands and trust resources by assisting tribes in managing habitat consistent with tribal priorities, reserved Indian rights, and legislative mandates.

Habitat Abundance

Due primarily to extensive timber cutting over the past 150 years, at least 82 percent of the old-growth forests existing in western Washington and Oregon prior to the 1840s have been harvested (Booth 1991, Teensma *et al.* 1991, Ripple 1994, Perry 1995). As a result, the loss of nesting habitat (old-growth/mature forest) has generally been identified as the primary cause of the murrelet population decline and disappearance across portions of its range (Ralph *et al.* 1995). Washington State contains approximately 48 percent (1,022,695 acres) of the estimated 2,223,048 acres of remaining suitable habitat in the three-state area (McShane *et al.* 2004). In Washington, murrelets are estimated to annually occupy at least 345,521 acres (16 percent) of the suitable nesting habitat (McShane *et al.* 2004). These totals represent available data but likely are not an accurate estimate of occupied habitat due to extensive missing data, as well as the low quality of much of the data. About 10 percent of pre-settlement old-growth forest remains in western Washington (Norse 1990; Booth 1991). Logging, urbanization, and agricultural development have all contributed to the loss of this habitat. The loss of old-growth habitat is most concentrated at lower elevations.

Environmental Baseline

The action area includes that area of Washington State with the largest density of murrelets. Marbled murrelets nest to the south, southwest and southeast of Dungeness Spit. They are known to forage on the water in the area of Dungeness Spit and tend to avoid large flocks of birds. Several species of seabirds are known to utilize the Dungeness Spit area, and quite a few nesting bald eagles are known from the area. Marbled murrelets compete with other seabirds for forage resources in the area, however, it is unknown how this competition may affect murrelet foraging success.

Effects of the Action

Marbled murrelets, Caspian terns, and many other seabirds already share and depend on the Dungeness Spit area as a significant foraging resource. There is no data to indicate what size of tern colony would cause a significant disruption of forage resource availability, or resource utilization by murrelets in the action area. Murrelets are not likely to be significantly affected by any competition with a larger Caspian tern colony when foraging for themselves, since their prey size selection is greater than that of the terns, and primarily includes small invertebrates and small schooling fish (≤5 cm) up to the size that terns take (≥5 cm or greater).

Prey size selection may force breeding adults to exercise more specific foraging strategies when feeding chicks, so availability of abundant forage fish during the nestling period has the potential to affect the energy demand on adults by influencing both foraging time and number of trips inland required to feed nestlings. Murrelets appear to select a relatively large fish (presumably ≥5 cm) to carry to their young, typically larger sand lance, immature herring, anchovy, smelt and only occasionally salmon smolts. While Caspian terns also capture large fish of the same size class in the Dungeness Spit area, terns primarily forage on surf perch and salmon smolts. Furthermore, murrelets typical dive depth of 5-10m utilizes a significantly larger section of the water column that the shallow diving (e.g., 0.3-1m) Caspian terns. While the two species depend on the Dungeness Spit area as a significant foraging resource, the level of interactions between murrelets and the increased number of terns should not result in a measurable change in habitat condition for the murrelets, due to the lack of a significant overlap in prey items and resource utilization.

Murrelet avoidance of large groups of terns and other seabirds may have a greater influence on murrelet foraging time and success than overall prey availability. However, the proposed action would result only in an increase in the size of an existing tern colony. Assuming the vast foraging resources in the Dungeness Spit area cannot be substantially dominated by additional Caspian terns, and that the baseline condition for murrelets includes avoidance of the Caspian tern colony in the Dungeness Spit area, murrelets will likely be able to continue to successfully avoid large groups of terns and continue to find suitable forage resources nearby. Therefore, any increase in avoidance behavior, given the availability of abundant forage fish in the action area, is not likely to significantly affect energy demand on adults, by measurably influencing foraging time or number of trips inland required to feed nestlings. Subsequently, there should not be a measurable increase in risk to nestlings from predation, the single largest cause of nest failure. If adult foraging time and success are not significantly disrupted by increased Caspian tern presence, marbled murrelet nestlings time at the nest (e.g., risk of predation) should not be significantly increased.

A larger Caspian tern colony may attract bald eagles and other raptors to locations where Caspian terns congregate or forage in the action area, thereby increasing risk of predation on foraging marbled murrelets. However, foraging perches for bald eagles are not available in the immediate area of Dungeness Spit. It would be difficult, if not impossible, to measure if an increase in the size of the existing tern colony would result in a significant increase in the number of raptors (e.g., leading to increased capture of marbled murrelets) in the area. Nevertheless, since marbled murrelets tend to avoid large congregations of seabirds, and murrelets tend to exhibit more sedentary (e.g., cryptic) behavior in the presence of predators, the focus of bald eagle and other raptor foraging activity may trend away from murrelets and toward terns. Therefore, an increased presence of Caspian terns in not likely to measurably increase, and may locally decrease, risk of raptor predation on murrelets.

Conclusion

The proposed action may affect, but will not likely adversely affect marbled murrelets at Dungeness NWR. We make this determination for the following reasons: (1) proposed management activities will not result in destruction or degradation of murrelet habitat, nor will they significantly disrupt normal marbled murrelet behavior patterns; (2) murrelets are not likely to be significantly affected by competition with a larger Caspian tern colony when foraging for themselves, since their prey size selection is different than that of the terns; (3) forage competition for fish prey and foraging space between marbled murrelets and Caspian terns would be considered insignificant due to the substantial forage base available to both species in the area, and the differences in forage resource (water column) utilization between Caspian tern and the murrelets; (4) adult foraging time and success are not likely to be significantly disrupted by increased Caspian tern presence, therefore, marbled murrelet nestlings time at the nest (e.g., risk of predation) will not likely be significantly increased; and (5) increased presence of Caspian terns in not likely to measurably increase, and may locally decrease, risk of raptor predation on murrelets.

Oregon Chub

Status of the Species

The Oregon chub is a small minnow (Family: Cyprinidae) endemic to the Willamette River drainage of western Oregon (Markle *et al.* 1991). This species was formerly distributed throughout the Willamette River Valley in off-channel habitats such as beaver ponds, oxbows, side channels, backwater sloughs, low gradient tributaries, and flooded marshes (Snyder 1908). Historical records show Oregon chub were found as far downstream as Oregon City and as far upstream as Oakridge. Records of Oregon chub collections exist for the Clackamas River, Molalla River, Mill Creek, South Santiam River, North Santiam River, Luckiamute River, Long Tom River, McKenzie River, Calapooia River, Muddy Creek, Mary's River, Coast Fork Willamette River, Middle Fork Willamette River, and the mainstem Willamette River (Markle *et al.* 1991, Scheerer and McDonald 2000).

In the last 100 years, backwater and off-channel habitats have been much reduced due to changes in seasonal flows resulting from the construction of dams throughout the basin, channelization of the Willamette River and its tributaries, removal of snags for river navigation, and agricultural practices. A variety of non-native aquatic species that prey on or compete with Oregon chub were introduced to the Willamette Valley over the same period. Consequently, these activities reduced available Oregon chub habitat, isolated the existing Oregon chub populations, restricted mixing between populations, reduced the probability of successful recolonization by Oregon chub, and introduced new competitors and predators into Oregon chub habitat (Service 1998a, Scheerer 2002).

At present, Oregon chub occur at approximately 27 locations in the North and South Santiam River, McKenzie River, Middle Fork Willamette River, Coast Fork, and several tributaries to the mainstem Willamette River downstream of the Coast Fork Willamette/Middle Fork Willamette confluence Scheerer *et al.* 2003). The ODFW has reintroduced Oregon chub at a number of sites within the Willamette Basin; seven currently sustain a population. In 2002, only nine populations of Oregon chub were larger than 1,000 fish, and eight populations numbered fewer than 100 individuals (Scheerer *et al.* 2003). Oregon chub appear to have been extirpated from at least nine locations at which they were detected in the 1990s (Scheerer *et al.* 2003).

Of the known Oregon chub populations, the sites with the highest diversity of native fish, amphibian, and reptile species have the largest populations of Oregon chub (Scheerer and McDonald 2000). Beavers (*Castor canadensis*) appear to be especially important in creating and maintaining habitats that support these diverse native species assemblages (Scheerer and Apke 1998).

Oregon chub are found in slack water off-channel habitats such as beaver ponds, oxbows, side channels, backwater sloughs, low gradient tributaries, and flooded marshes. These habitats usually have little or no water flow, silty and organic substrate, and considerable aquatic vegetation as cover for hiding and spawning (Pearsons 1989, Markle *et al.* 1991; Scheerer and McDonald 2000). The average depth of Oregon chub habitats is typically less than two meters (six ft) and the summer temperatures typically exceed 16°C (61°F). Adult Oregon chub seek dense vegetation for cover and frequently travel in the mid-water column in beaver channels or along the margins of aquatic plant beds. Larval chub congregate in near shore areas in the upper layers of the water column in shallow areas (Pearsons 1989, Scheerer 1997). Juvenile Oregon chub venture farther from shore into deeper areas of the water column (Pearsons 1989). In the winter months, Oregon chub can be found buried in the detritus or concealed in aquatic vegetation (Pearsons 1989). Fish of similar size classes school and feed together. In the early spring, Oregon chub are most active in the warmer, shallow areas of the ponds.

The Oregon chub evolved in a dynamic network of slack water habitats in the floodplain of the Willamette River. Major alteration of the Willamette River for flood control and navigation improvements has eliminated a large proportion of the river's historic floodplain. This alteration has also impaired or eliminated the environmental conditions in which the Oregon chub evolved. Many of the remaining suitable habitats have been invaded by non-native fish predators and competitors.

The current pattern of distribution and abundance of Oregon chub populations reflects the fundamental alteration in the natural processes under which the species evolved. Sites with Oregon chub can be categorized as having high or low connectivity to the Willamette and its tributaries; those sites with low connectivity tend to have large populations of chub and fewer species of non-native fish (Scheerer 2002). Thus, Oregon chub now thrive only in habitats that are isolated and bear little resemblance to the species' dynamic natural environment. Efforts to restore floodplain function and connectivity may facilitate the introduction of non-native fishes into isolated habitats, which could have devastating effects to populations of Oregon chub (Scheerer 2002).

There is considerable overlap in the length distributions among ages of sampled Oregon chub (Scheerer and McDonald 2000). The largest Oregon chub on record was collected from the Santiam River and measured 89 mm (3.5 inches) (Scheerer *et al.* 1995). Oregon chub spawn from April through September. Before and after spawning season, chub are social and non-aggressive. Spawning activity has only been observed at temperatures exceeding 16°C (61°F). Males over 35 mm (1.4 inches) have been observed exhibiting spawning behavior (Pearsons 1989).

Oregon chub are obligatory sight feeders (Davis and Miller 1967). They feed throughout the day and stop feeding after dusk (Pearsons 1989). Chub feed mostly on water column fauna. The diet of Oregon chub adults collected in a May sample consisted primarily of minute crustaceans including copepods, cladocerans, and chironomid larvae (Markle *et al.* 1991). The diet of juvenile chub also consisted of minute organisms such as rotifers, copepods, and cladocerans (Pearsons 1989).

A variety of factors are likely responsible for the decline of the Oregon chub. These include habitat alteration; the proliferation of non-native fish and amphibians; accidental chemical spills; runoff from herbicide or pesticide application on farms and timberlands or along roadways, railways, and power line rights-of way; the application of rotenone to manage sport fisheries; desiccation of habitats; unauthorized water withdrawals, diversions, or fill and removal activities; sedimentation resulting from timber harvest in the watershed, and possibly the demographic risks that result from a fragmented distribution of small, isolated populations (Service 1993a, Service 1998b).

Based on a 1987 survey (Markle *et al.* 1989) and compilation of all known historical records, at the time of the petition for listing in 1991, viable populations of the Oregon chub occurred in the following locations: Dexter Reservoir, Shady Dell Pond, Buckhead Creek near Lookout Point Reservoir, Elijah Bristow State Park, William L. Finley NWR, Greens Bridge, and East Fork Minnow Pond. These locations represented a small fraction - estimated as two percent based on stream miles - of the species' formerly extensive distribution within the Willamette River drainage.

The decline of Oregon chub has been correlated with the construction of dams. Based on the date of last capture at a site, Pearsons (1989) estimated that the most severe decline occurred during the 1950s and 1960s. Ten of the 13 dams that make up the Willamette Valley flood control system were completed between 1953 and 1969 (Corps 2000). Other structural changes along the Willamette River corridor such as revetment and channelization, diking and drainage, and the removal of floodplain vegetation have eliminated or altered the slack water habitats of tbe Oregon chub (Willamette Basin Task Force 1969, Hjort *et al.* 1984, Sedell and Froggatt 1984, Li *et al.* 1987). Channel confinement, isolation of the Willamette River from the majority of its floodplain, and elimination or degradation of both seasonal and permanent wetland habitats within the floodplain began as early as 1872 and, for example, has reduced the 25-km (15.5-mile) reach between Harrisburg and the McKenzie River confluence from over 250 km (155 miles) of shoreline in 1854 to less than 64 km (40 miles) currently (Sedell and Froggatt 1984, Sedell *et al.* 1990).

The establishment and expansion of non-native species in Oregon have contributed to the decline of the Oregon chub and limits the species' ability to expand beyond its current range. Many species of non-native fish have been introduced to, and are common throughout, the Willamette Valley, including largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieui*), crappie (*Pomoxis* sp.), bluegill (*Lepomis macrochirus*), and western mosquitofish (*Gambusia affinis*). The bullfrog (*Rana catesbiana*), a non-native amphibian, also occurs in the valley and breeds in habitats preferred by the Oregon chub (Willamette Basin Task Force 1969, Hjort *et al.* 1984, Li *et al.* 1984, Scheerer *et al.* 1992). The period of severe decline of the Oregon chub does not coincide well with the initial dates of introduction of nonindigenous species. However, many sites formerly inhabited by the Oregon chub are now occupied by non-native species (Markle *et al.* 1989). Currently, 25 sites are known to contain Oregon chub; over half of these sites are also inhabited by non-native fishes or amphibians (Scheerer and McDonald 2000). Since 1995, non-native fish have been discovered for the first time in six locations containing Oregon chub; the Oregon chub populations have subsequently declined or remained in low abundance in all of these sites (P. Scheerer, pers. comm., 2000). The 1996 flooding in the Santiam River was probably responsible for three of these movements of non-native fish. The other three sites, located in the Middle Fork Willamette River drainage, were likely the result of unauthorized introductions or spread of non-native fish from reservoirs (Scheerer and Jones

1997). Because all remaining population sites are easily accessible, there also continues to be a potential for unauthorized introductions of non-native species, particularly mosquitofish and game fishes such as bass and walleye (*Stizostedion vitreum*).

Specific interactions responsible for the exclusion of Oregon chub from habitats dominated by non-native species is not clear in all cases. While information confirming the presence of Oregon chub in stomach contents of predatory fishes is lacking, many non-native fishes, particularly adult centrarchids (e.g., bass) and ictalurids (e.g., catfish) are documented piscivores (fish eaters) (Carlander 1969, Moyle 1976, Carlander 1977, Wydoski and Whitney 1979, Li *et al.* 1987). These fishes are frequently the dominant inhabitants of ponds and sloughs within the Willamette River drainage and may constitute a major obstacle to Oregon chub recolonization efforts. Adult bullfrogs prefer habitat similar in characteristics (i.e., little to no water velocity, abundant aquatic and emergent vegetation) to preferred habitat for Oregon chub, and are known to consume small fish as part of their diet (Cohen and Howard 1958, Bury and Whelan 1984), although Scheerer and McDonald (2000) did not find any fish in the stomachs of bullfrogs collected from Oregon chub ponds. Non-native fishes may also serve as sources of parasites and diseases; however, disease and parasite problems have not been studied in the Oregon chub.

Observed feeding strategies and diet of introduced fishes, particularly juvenile centrarchids (e.g., bass, crappie), adult mosquitofish (Li *et al.* 1987) and bullfrogs (Cohen and Howard 1958; Kane *et al.* 1992), in many cases overlap with diet and feeding strategies described for Oregon chub (Pearsons 1989). This suggests that direct competition for food between Oregon chub and introduced species may further impede species survival as well as recovery efforts.

Many of the known extant populations of Oregon chub occur near rail, highway, and power transmission corridors and within public park and campground facilities. These populations are threatened by chemical spills from overturned truck or rail tankers; runoff or accidental spills of vegetation control chemicals; overflow from chemical toilets in campgrounds; sedimentation of shallow habitats from construction activities; and changes in water level or flow conditions from construction, diversions, or natural desiccation (Service 1998b). Oregon chub populations near agricultural areas are subject to poor water quality as a result of runoff laden with sediment, pesticides, and nutrients. Logging in the watershed can result in increased sedimentation and herbicide runoff.

Effects of the Action

Two Oregon chub populations are known to occur within foraging range of Caspian terns. However, Caspian terns are not likely to prey on Oregon chub because they are a small (<5cm) fish with very cryptic coloration. Furthermore, chub habitat, as described above, is not particularly conducive to tern foraging. Terns are more likely to forage in open water. Finally, no chubs occur within 5 miles of Fern Ridge Lake, well outside the Fern Ridge Lake area where the vast majority of the tern foraging is expected to occur.

Conclusion

The proposed action may affect, but will not likely adversely affect Oregon chub at Fern Ridge Lake. We make this determination for the following reasons: (1) The cryptic features, small size of the Oregon chub, habitat conditions, and distance from the tern colony site make successful foraging of chub by Caspian terns extremely unlikely to occur, and is therefore discountable.

Bradshaw's Lomatium

Status of the Species

Historic distributions of Bradshaw's lomatium are thought to have been extensive, occupying wetland and upland prairie habitats throughout much of the Willamette Valley (Service 1988, Service 2000). This extensive resource has been dramatically depleted since European settlement from the 1840s to present through conversion of native prairie to agricultural use and urbanization (Boag 1992). Current estimates of the remaining native upland prairie in the Willamette Valley are less than 400 ha (988 acres) (Christy and Alverson 2002). This estimate represents only one-tenth of one percent of the original upland prairie once present. Similar losses have occurred for wetland prairie habitats, but estimates of current acreage are not available.

Native habitat for Bradshaw's lomatium is characterized by seasonally wet conditions and typically called wet prairie or wetland prairie. This relic wetland prairie has been described as the tufted-hairgrass (*Deschampsia caespitosa*) valley prairie and has been studied extensively (Pendergrass 1995, Streatfield and Frenkel 1997, Moir and Mika 1976, Alverson 1990, Meinke 1982). In the wet areas, both species generally occur on the edges of elevated pedestals dominated by tufted-hairgrass or sedge bunches. In the drier areas of wetland prairies, these species are found in the low areas, such as small depressions, trails or seasonal channels, with open, exposed soils.

Threats

A serious long-term threat to all Willamette Valley prairie species is the change in community structure due to plant succession. The majority of Willamette Valley prairies would likely become forested if left undisturbed. Shrub and tree intrusion has been documented on most of the relic prairie sites occupied by Bradshaw's lomatium (Service 1988, Service 2000, ONHP 2003). The natural transition of prairie to forest in the absence of disturbance such as fire will lead to the eventual loss of these prairie sites unless they are actively managed (Clark *et al.* 1993, Franklin and Dyrness 1973, Johansson *et al.* 1971). The presence of tall, fast-growing non-native herbaceous species may speed the conversion of native prairie to dense, rank prairies and shrub lands. Invasion by alien plant species has been documented at most Bradshaw's lomatium sites (ONHP 2003). Non-native grass species aggressive enough to out-compete and suppress native species include velvet grass, orchard grass, false-brome (*Brachypodium sylvaticum*), tall oat-grass (*Arrenatherum elatius*) and bent grasses (*Agrostis* spp.) (Hammond 1996).

Small population size is a threat to the listed plants in the Willamette Valley. Often fence rows, pastures and intervening strips of land along agricultural fields and roadsides currently serve as the only refugia for Bradshaw's lomatium. These small and fragmented populations contribute to the continued existence of these taxa although small populations are more vulnerable to environmental changes than relatively large and contiguous populations. Generally, the direct and indirect effects of small population size on most species, plant and animal, include loss of connectivity for dispersal, a decrease in genetic exchange, a resultant loss of population viability and vigor, and a hastening towards extinction (Gilpin and Soule 1986). The importance of small populations lies in their potential to serve as stepping stones between larger neighboring populations. The loss of small populations and remnant prairie habitats further isolates larger populations and limits opportunities for genetic exchange, migrations and/or re-colonization. The negative impacts of demographic inbreeding depression typically occur in populations that have less than 50 individuals (Falk and Holsinger 1991). The modern use of herbicides by highway departments and agricultural interests for weed control and landscape maintenance further exacerbates the precarious survival of these remnant plant populations.

A number of large sites (greater than 10 ha) of Bradshaw's lomatium are being actively managed and are thus secure from habitat loss and have relatively stable populations. These larger sites provide the greatest potential for long-term persistence of the species if the current condition of these sites can be sustained or improved.

Distribution and Abundance

Formerly a common species on wetland prairies, habitat loss has led to its decline and listing as an endangered species in September 1988 (Service 1988). The species currently ranges from Clark County, Washington, south to Lane County, Oregon.

The species also occurs in Benton, Linn, and Marion Counties in Oregon. A large concentration of Bradshaw's lomatium is found west of Eugene (Service 1993b). The ORNHIC (ONHP 2003) has mapped a total of 1,156,919 m^2 (116 ha; 286 acres) occupied habitat in 100 habitat patches that exist today, and the occupied habitat ranges in size from 2 m^2 to 233,548 m^2.

Population Trends

A number of sites are secure and are being managed to benefit this and other native prairie species. Approximately 65 percent of extant occupied habitat is currently protected at 29 sites including lands managed by the Corps, The Nature Conservancy (TNC), BLM, Lane and Benton Counties, the City of Eugene and in the Service's NWR System.

Life History and Demography

Bradshaw's lomatium is a tap-rooted perennial. A typical population is composed of many more vegetative plants than reproductive plants. Density sampling at Green Mountain in Clark County, Washington in 1996 indicated that 20 to 30 percent of the Bradshaw's lomatium plants were in flower (unpublished data, Service, 2003). Bradshaw's lomatium blooms fairly early in the spring, usually in April and early May. Bradshaw's lomatium is insect pollinated, does not spread vegetatively, and depends exclusively on seeds for reproduction (Kaye 1992). Bradshaw's lomatium produces large fruits and seeds are generally shed by the end of July and the species does not appear to maintain a persistent soil seed bank (Kaye 1992, Kaye and Kirkland 1994).

Compared to widespread *Lomatium* species, Bradshaw's lomatium does not appear to suffer from a lack of genetic diversity (Gitzendanner 2000). While both populations in Clark County, Washington, are quite large, it appears, at least based on the data from the Green Mt. population, that they have reduced levels of genetic diversity. While the isolated populations in Washington appear to have lower levels of diversity, they do not appear to be genetically differentiated from the other populations of the species, consistent with historical gene flow among all populations, and a recent bottleneck in the Washington populations. The Kingston Prairie population near Silverton, Oregon, was the most genetically distant population from all other populations. Studies that include additional sites across the species entire distribution are needed.

Abundance and Distribution in the Action Area

Corps of Engineers Project Lands at Fern Ridge support important populations of Bradshaw's lomatium. Along with West Eugene Wetlands, and Amazon Creek in South Eugene, Fern Ridge is identified in the Recovery Plan as one of three recovery populations in the South Valley area. The Fern Ridge population consists of three distinct sub-populations encompassing nearly 240 acres within remnant wet prairie in the Amazon-Dike 2, Rose Prairie, Royal-Amazon, and Fisher Butte Management Units. High quality prairies within these units have been designated as a Research Natural Area, and are managed to maintain the prairie plant community, while allowing access for research. Bradshaw's lomatium plants within the RNA number in the 10's of thousands, and have responded well to prescribed burning.

In addition to the large intact prairie populations at Fern Ridge, Bradshaw's lomatium is also found in small patches consisting of two to several plants in the Kirk Pond and Amazon-Dike 2 Management units, and scattered along the impoundment dikes in the East and West Coyote Management units. The role of these isolated plants in recovery and maintenance of the species is not clear; accordingly, management efforts are focused on larger intact prairie communities; individual plants found growing outside the communities are protected but not given high priority for management.

Relevant to the proposed project, a small patch of Bradshaw's lomatium was located along the Royal avenue ditch in the early 1990's. In 1996, approximately 300 plants were growing adjacent to the old road, between the road and the drainage ditch, in soil that was pushed up or placed, presumably during original construction of the roadbed. These plants will be protected with fencing to prevent accidental damage from machinery or personnel during use of Royal Avenue access as a haul road.

Effects of the Action

A small patch of Bradshaw's lomatium was located along the Royal Avenue ditch in the early 1990's. In 1996, approximately 300 plants were growing adjacent to the old road, between the road and the drainage ditch, in soil that was pushed up or placed, presumably during original construction of the roadbed.

The small Bradshaw's lomatium population located adjacent to the Royal Avenue haul road will be protected from inadvertent or accidental damage caused by trucks hauling material to the work site. The population will be mapped, flagged, and steel fence posts will be used to delineate the site and prevent contact with the plants. These lomatium occurrences will be monitored during construction, site maintenance, and tern monitoring activities. The use of steel fence posts worked well during construction of the much larger Fern Ridge Marsh restoration project in 1997 and the Gibson Island dike in 2003.

Conclusion

The proposed action may effect, but will not likely adversely affect Bradshaw's lomatium at Fern Ridge Lake. We make this determination on the following reasons: (1) a fenced exclusion area(s) will be established and occupied habitat will be flagged prior to proposed construction and maintenance activities and monitored during these activities, to avoid destruction or degradation of Bradshaw's lomatium habitat; (2) construction, maintenance and monitoring activities will be restricted to Royal Avenue and the lake; and (3) no interactions between Bradshaw's lomatium and Caspian terns are anticipated.

Kincaid's Lupine and Fender's Blue Butterfly

Status of the Species

Fender's blue butterfly was listed as federally endangered, and Kincaid's lupine as threatened, on January 25, 2000 (67 FR 3875). A critical habitat determination has not been made for either species. Much of the following information for these species was extracted from the Final Rule designating endangered and threatened status for these species, from an unpublished "Willamette Basin Overview" report from TNC to the Service (2000), and from summary of current Oregon Natural Heritage Program database (ONHP 2002) and Service database information (current Service database).

Over 80 percent of the remaining upland prairies where these species are known to occur are threatened by agriculture and forest practices, development, grazing, and road construction and maintenance. Kincaid's lupine is the primary host food plant for Fender's blue caterpillars, and the two species are currently known to co-occur at 25 sites on approximately 279 ac across their ranges.

Historic Distribution and Habitat

Kincaid's lupine and Fenders blue butterfly are thought originally to have been widely distributed on upland prairie habitats throughout the Willamette Valley, with the lupine extending into the Umpqua Valley, Oregon. Prior to 1850, there was approximately 685,000 ac of upland prairie in Willamette Valley (Habeck 1961, TNC *et al.*1998). This extensive resource has been dramatically depleted since European settlement began in the 1840's, through fire suppression, agricultural conversion, urbanization (Boag 1992), and the introduction of non-native vegetation (Franklin and Dyrness 1973). Current estimates of the remaining native upland prairie in the Willamette Valley are less than 988 ac (TNC *et al.* 1998). This estimate represents only 0.1 percent of the original upland prairie once present.

A serious long-term threat to all Willamette Valley prairie species is the change in plant community structure due to succession. Without active management, the natural succession of prairie to shrub/forest by the invasion of native species, such as Oregon ash (*Fraxinus latifolia*), Douglas hawthorn (*Crataegus douglasii*), Nutka rose (*Rosa nutkana*) and Douglas spiraea (*Spiraea douglasii*) alone would lead to the eventual loss of these prairie sites (Franklin and Dyrness 1973; Hammond and Wilson 1993; Johannessen *et al.* 1971; Kuykendall and Kaye 1993). The presence of invasive non-native woody species, such as Himalayan blackberry, multiflora rose (*Rosa multiflora*) and Scotch broom, exacerbate this problem. Shrub and tree intrusion has been documented on most of the relic prairie sites occupied by Kincaid's lupine and Fender's blue butterfly.

Often fencerows, pastures, and intervening strips of land along agricultural fields and roadsides serve as the only remaining refugia for native upland prairie endemic plants, which therefore occur in small and fragmented populations. Three large hilltop prairie areas remain despite development pressures on the Willamette Valley floor (Baskett Slough NWR, Coburg Ridge, and McDonald State Forest). Two of these sites, Baskett Slough and Coburg Ridge, are being actively managed for populations of Kincaid's lupine and Fender's blue butterfly. These larger sites provide the greatest potential for long-term persistence of the species if their current condition can be sustained or improved. The importance of small populations lies in their potential to serve as stepping stones between larger, neighboring populations. The loss of small populations and remnant prairie habitats further isolates larger populations and limits opportunities for genetic exchange, migrations and/or re-colonization.

The modern use of herbicides for highway or roadway maintenance, farming practice, or other land uses for weed control and landscape maintenance purposes is further exacerbating the precarious survival of these remnant plant populations.

Kinkaid's lupine

Kincaid's lupine is a perennial forb generally associated with native fescue upland prairies that are characterized by heavier soils, with moderate to slightly dry soil moisture levels. At the southern limit of its range, the subspecies occurs on well-developed soils adjacent to serpentine outcrops where the plant is often found under scattered oaks (Kuykendall and Kaye 1993). Kincaid's lupine is thought to have historically colonized areas along the edge of oak woodlands in upland prairies. Schultz (1997) theorizes that lupine patches were historically distributed no greater than 0.5 km (0.3 mi) apart, allowing dispersal of Fender's blue butterfly between lupine patches.

Kincaid's lupine is a long-lived perennial species with a maximum reported age of 25 years. Individual plants are capable of spreading by rhizomes, producing clumps of plants exceeding 20 m (33 feet) in diameter. The flowering period generally occurs during May and June (Eastman 1990, Hitchcock *et al.* 1961). Self-incompatible, Kincaid's lupine must obtain pollen from another individual plant to produce fertile seeds and is therefore, dependent on solitary bees and flies for pollination. Seed set and seed production are low, with few flowers producing fruit from year to year and each fruit containing an average of 0.3 to 1.8 seeds. Seeds are dispersed from fruits that open explosively upon drying.

Kincaid's lupine occurs in 97 remnant upland prairie patches, averaging 1.454 km^2 (0.561 square miles) in size, scattered from Lewis County, Washington to Douglas County, Oregon (current Service database). Within the Willamette Valley, Kincaid's lupine occupies 86 habitat patches averaging 1.395 km^2 (0.539 square miles) in size. In the Umpqua Valley, Douglas County, Oregon, Kincaid's lupine occupies eight small patches, averaging 0.057 km^2 (0.022 square miles) in size, and in Lewis County, Washington, three tiny patches, averaging 0.002 km^2 (0.0008 square miles) in size.

Fender's Blue Butterfly

Fender's blue butterfly is a Willamette Valley endemic subspecies that was considered to be extinct until rediscovered by Dr. Paul Hammond in 1989 in McDonald Forest, Benton County, Oregon. The historical distribution of Fender's blue butterfly is not precisely known, due to the limited information collected on this species prior to its description in 1931. Recent surveys have determined that Fender's blue butterfly is confined to 33 habitat patches in Yamhill, Polk, Benton, and Lane counties, Oregon. One population at Willow Creek TNC preserve in Eugene, Lane County, Oregon is found in wet Deschampsia-type prairie, while the remaining sites are generally found on drier upland prairies characterized by fescue species. The Willow Creek aggregate of populations is the largest of the south valley sites.

Fender's blue butterfly is known to use Kincaid's lupine as its primary larval food plant but is also known to use spur lupine *(Lupinus laxiflorus = L. arbustus)* and sickle-keeled lupine *(L. albicaulis)* as secondary host plants. Female Fender's blue butterfly lay their eggs on lupine foliage in late May or early June; and larvae emerge to feed on foliage during late June. In July, larvae crawl to the base of the plant and enter diapause. From this point until the larvae emerge and begin feeding on foliage again the following April, the larvae remain at the base of the senescent plant, or in the litter immediately adjacent to the lupine stem.

Fender's blue butterfly density has been positively correlated with the number of Kincaid's lupine flowering racemes, and more recently, to nectar production in native flowering species used as nectar sources by Fender's blue butterflies. The abundance of exotic grasses can effectively preclude butterflies from using a Kincaid's lupine patch (Hammond 1996).

Recent research (Schultz and Dlugosch 1999) indicates that native wildflowers in the Willamette Valley prairies provide more nectar than nonnative flowers for adult butterflies, and that Fender's blue butterfly population density is positively correlated with the density of native wildflowers.

Tall oatgrass and other non-native grasses can out-compete native prairie species (Hammond 1996).

The flight season for Fender's blue butterfly occurs for approximately 3 weeks, generally beginning and ending sometime in May-June (K. Pendergrass, Service, pers. comm. 2004). Anecdotal evidence indicates that under ideal conditions adult Fender's blue butterflies may disperse as far as 5-6 km (3.1 to 3.7 miles) from their natal lupine patches (Hammond and Wilson 1992; and Schultz 1994). According to Schultz (1997), adult dispersal of this magnitude is not likely anymore. Schultz (1997) found that the butterflies are generally found within 10 m (32.8 feet) of lupine patches, although they might disperse more than 2 km (1.2 miles) between lupine patches. Hammond (1998) reports recolonization of a site by Fender's blue butterfly from a distance of approximately 3 km (1.9 miles). Schultz (1997) further theorizes that Fender's blue originally would have had a high probability of dispersing between patches, which were historically located, an average of 0.5 km (0.3 miles) apart. Current distribution of lupine patches range well beyond this distance, and barriers to migration between close sites may be present.

Today, remnant upland prairie acreage is extremely fragmented and remaining Fender's blue butterfly populations so small that migration processes are not expected to maintain the population over time. Extirpation of remaining small populations is expected from localized events and low genetic diversity of very small populations. The low availability of host lupine patches and fragmentation of habitat are seen today as the major ecological factors limiting reproduction, dispersal, and subsequent colonization of new habitat (Hammond 1994, Hammond and Wilson 1993 &1992, Schultz 1997, Schultz and Dlugosch 1999),

Censuses of Fender's blue butterfly were started in 1991; most of the 22 census units have been surveyed every year since 1993 (Fitzpatrick and Schultz 2001, Hammond 2001, 1998, 1996 and 1994, Hammond and Wilson 1993, Schultz 1994-1998).

Total range-wide population numbers (once most sites were monitored) of Fender's blues have ranged from a low of 1,384 in 1998 to a high in 2000 of 3,492. Although population size appears to have increased between 1998 and 2000, this could be a result of poor weather conditions in 1998, and thus poor flight conditions, and it could also be an artifact of increasing survey effort at these sites. However, some of this increase may be attributed to habitat enhancement activities such as tree and shrub removal from lupine sites.

Abundance and Distribution in the Action Area

Kinkaid's lupine and Fender's blue butterfly

Butterfly censuses have been conducted at Fern Ridge since 1997. In 2002, a total of 195 adults (compared to 17 in 1998 and 122 in 2001) were counted at 11 Fern Ridge sites. Subsampling of lupine patches indicates that this species occupied slightly more than 690m^2 and produced 18,000 racemes in 2002. All prairie sites in the vicinity of Shore, Spires and Eaton lanes, Green Oaks, and in the Fisher Butte unit were mown, most for the fourth year in a row. In addition to this late season mowing, the South Green Oaks patch was treated with "weed eaters" in July to reduce Himalaya blackberry (*Rubus armeniacus*), and seed set by meadow knapweed (*Centaurea pratensis*). The southern portion of the North Green Oaks site was treated with weed-eaters to control tall oatgrass (*Arrhenatherum elatius*), blackberry, and meadow knapweed. No lupine or nectar species were planted in the 2002 season. The Royal Amazon unit prescribed burn of September 28, 2002, included a single lupine patch where Fender's butterflies have never been recorded. Lupine collected under section 10 permit TE-041672-0 and planted at Horkelia Prairie in 2001 was monitored through the growing season. Although 76 seedlings were found on June 18, 2002, high mortality was expected during the first dormant season.

Effects of the Action

Kinkaid's lupine

Butterfly censuses were conducted according to the methods employed at Fern Ridge since 1997 (Corps 2003). In 2002, these surveys counted a total of 195 adults (compared to 17 in 1998 and 122 in 2001) for all Fern Ridge sites. Recent management activities for lupine patches in the area have included late-season treatments for weed encroachment by prescribed fire, mowing or weed-trimmer.

In the vicinity of Royal Avenue, there are three known Kinkaid's lupine occurrences, one patch in the Fisher Butte site immediately to the south of Royal Avenue, and two smaller patches located a few hundred meters to the north within the Royal Amazon site.

Although Kinkaid's lupine occurs nearby to the south of Royal Avenue, proposed construction, maintenance, and monitoring related activities in uplands within the action area are restricted to Royal Avenue, and the lupine patch will be mapped, protected with fencing and monitored to prevent accidental damage from machinery, staging areas, or human activities during use of Royal Avenue access as a haul road and during construction related activities, and will, therefore, not adversely affect this species. No effects to Kinkaid's lupine are expected to result from the increased presence of Caspian tern.

Fender's blue butterfly

The flight season of Fender's blue butterfly (approximately May through June) does not overlap with the timing of proposed construction activities (from October through January) or maintenance activities (late-Fall to late-Winter, and prior to the arrival of Caspian terns to the site in March). Therefore, no effect to flying butterflies is anticipated from these elements of the proposed action.

The nearest occupied lupine site is located to the south of Royal Avenue (Fisher Butte site). This lupine patch, which may include Fender's blue butterfly eggs or larvae, will be marked on maps, flagged and monitored to prevent accidental damage from machinery, staging areas, or human activities during use of Royal Avenue access as a haul road and during construction related activities. An adequate staging area for construction, maintenance and monitoring activities occurs at the very end of Royal Avenue. This area is surrounded by suitable lupine habitat and will be fenced to prevent human access related impacts to the habitat. In addition, the private drive immediately prior to the end of Royal Avenue (driveway turns to the south) will be marked both on maps as off limits to construction activities (including turn-arounds). The Fisher Butte site is located near the end of this road. Necessary steps will be taken to avoid impacts to habitat containing nectar plants. If any Fender's blue butterfly habitat, known to contain nectar plants, are identified as potentially threatened by construction activities, they will also be protected with fencing, marked on maps and monitored to prevent accidental damage from machinery, staging areas, or human activities.

Caspian tern monitoring activities will occur between March and September, within the flight season of the butterfly. It is assumed that monitors would use a vehicle to access the island via Royal Avenue. The potential for a butterfly to be killed or injured by direct impact with vehicles, although not entirely discountable, would be extremely unlikely to occur.

No direct or indirect effects to Fender's blue butterflies are expected to result from the increased presence of Caspian Tern.

Conclusion

Kinkaid's lupine

The proposed action may affect, but will not likely adversely affect Kinkaid's lupine at Fern Ridge Lake. We make this determination for the following reasons: (1) fenced exclusion areas will be established and occupied habitat will be flagged prior to proposed construction and maintenance activities and monitored during these activities, to avoid destruction or degradation of Kinkaid's lupine habitat; (2) construction, maintenance and monitoring activities will be restricted to Royal Avenue and the lake; and (3) no interactions between Kinkaid's lupine and Caspian terns are anticipated.

Fender's blue butterfly

The proposed action may effect, but will not likely adversely affect Fender's blue butterfly at Fern Ridge Lake. We make this determination on the following reasons: (1) fenced exclusion areas will be established and occupied habitat areas will be flagged prior to proposed construction and maintenance activities and monitored during these activities, to avoid destruction or degradation of Fender's blue butterfly habitat, eggs, larvae and habitat; (2) construction, maintenance and monitoring activities will be restricted to Royal Avenue and the lake; and (3) no interactions between Fender's blue butterfly and Caspian terns are anticipated.

Warner Sucker

Status of the Species

Species/critical habitat description

The Warner sucker is endemic to the Warner Valley in southeast Oregon, an endoreic (closed) sub-hasin of the Great Basin area. The valley contains a dozen lakes and many potholes during wet years, but only the three southernmost lakes are semi-permanent. In addition, three permanent creeks drain into the valley (Honey, Deep, and Twentymile Creeks).

The Warner sucker is a slender-bodied species that attains a maximum recorded fork length (the measurement on a fish from the tip of the nose to the middle of the tail where a V is formed) of 456 millimeters (17.9 inches). Bond and Coombs (1985) listed the following characteristics of the Warner sucker that differentiate it from other western species of *Catostomus*: dorsal fin base short, its length typically less than, or equal to, the depth of the head; dorsal fin and pelvic fins with 9 to 11 rays; lateral line (microscopic canal along the body, located roughly at midside) with 73-83 scales, and greater than 25 scales around the caudal peduncle (rear, usually slender part of the body between the base of the last anal fin ray and the caudal fin base); eye small, 0.035 millimeter (0.0013 inch) Standard Length (straight-line distance from the tip of the snout to the rear end of the vertebral column) or less in adults; dark pigmentation absent from lower 1/3 of body; in adults, pigmented area extends around snout above upper lip; the membrane-covered opening between bones of the skull (fontanelle) is unusually large, its width more than one half the eye diameter in adults.

The Service listed the Warner sucker as a threatened species and designated critical habitat on September 27, 1985 (Service 1985). Warner sucker critical habitat includes the following areas: Twelvemile Creek from the confluence of Twelvemile and Twentymile Creeks upstream for about six stream kilometers (four stream miles); Twentymile Creek starting about 14 kilometers (nine miles) upstream of the junction of Twelvemile and Twentymile Creeks and extending downstream for about 14 kilometers (nine miles); Spillway Canal north of Hart Lake and continuing about three kilometers (two miles) downstream; Snyder Creek, from the confluence of Snyder and Honey Creeks upstream for about five kilometers (three miles); Honey Creek from the confluence of Hart Lake upstream for about 25 kilometers (16 miles). Warner sucker critical habitat includes 16 meters (50 feet) on either side of these waterways.

Warner sucker life history

Much of the information on the distribution and life history is taken from the Recovery Plan for the Threatened and Rare Native Fishes of the Warner Basin and Alkali Subbasin (Service 1998a). Information from research and observations since completion of the recovery plan has been added.

The distribution of Warner sucker is well known, but limited information is available on stream habitat requirements and spawning habits. Relatively little is known about feeding, fecundity, recruitment, age at sexual maturity, natural mortality, or interactions with introduced gamefishes. In this account, "larvae" refers to the young from the time of hatching to transformation into juvenile (several weeks or months), and "juvenile" refers to young that are similar in appearance to adults. Young of year refers to members of age-group 0, including transformation into juvenile until January 1 of the following year.

Observations indicate that Warner sucker grow larger in the lakes than they do in streams (White *et al.* 1990). The smaller stream morph (development form) and the larger lake morph are examples of phenotypic plasticity within metapopulations of the Warner sucker. Expressions of these two morphs in Warner sucker might be as simple as the species being opportunistic. The larger, presumably longer-lived, lake morphs are capable of surviving through several continuous years of isolation (e.g., drought or other factors) from stream spawning habitats. Similarly, stream morphs probably serve as sources for recolonization of lake habitats in wet years following droughts, such as the refilling of the Warner Lakes in 1993 following their desiccation in 1992. The loss of either lake or stream morphs to drought, winter kill, excessive flows and a flushing of the fish in a stream, in conjunction with the lack of safe migration routes and the presence of predaceous exotic fishes, may strain the ability of the species to rebound (White *et al.* 1990, Berg 1991).

Lake morph Warner sucker occupy the lakes and, possibly, deep areas in the low elevation creeks, reservoirs, sloughs and canals. Recently, only stream morph suckers have exhibited frequent recruitment, indicated by a high percentage of young of year and juveniles in Twelvemile and Honey Creeks (Tait and Mulkey 1993a,b). Lake morph suckers, on the other hand, were skewed towards larger, older adults (8-12 years old) with no juveniles and few younger adult fish (White *et al.* 1991) before the lakes dried up in 1992. Since the lakes refilled, the larger lake morph suckers have reappeared. Captured lake suckers averaged 267 millimeters (10.5 inches) SL in 1996 (Chris Allen, TNC, Fishery Biologist, Portland, Oregon, pers. comm., 1996), 244 millimeters (9.6 inches) SL in 1995 (Allen *et al.* 1995a) and 198 millimeters (7.8 inches) SL in 1994 (Allen *et al.* 1995b). Stream caught fish averaged 138 millimeters (5.4 inches) SL in 1993 (Tait and Mulkey 1993b).

Warner sucker recovered from an ice induced kill in Crump Lake were aged to 17 years old and had a maximum fork length of 456 millimeters (17.9 inches) (White *et al.* 1991). Sexual maturity occurs at an age of three to four years (Coombs *et al.* 1979), although in 1993, captive fish at Summer Lake Wildlife Management Area, Oregon, successfully spawned at the age of two years (White *et al.* 1991).

The feeding habits of the Warner sucker depend to a large degree on habitat and life history stage, with adult suckers becoming more generalized than juveniles and young of year. Larvae have terminal mouths and short digestive tracts, enabling them to feed selectively in midwater or on the surface. Invertebrates, particularly planktonic (having weak powers of locomotion) crustaceans, make up most of their diet. As the suckers grow, they develop subterminal mouths, longer digestive tracts, and gradually become generalized benthic (living on the bottom) feeders on diatoms (small, usually microscopic, plants), filamentous (having a fine string-like appearance) algae, and detritus (decomposed plant and animal remains). Adult stream morph suckers forage nocturnally over a wide variety of substrates such as boulders, gravel, and silt. Adult lake morph suckers are thought to have a similar diet, though caugbt over predominantly muddy substrates (Tait and Mulkey 1993a,b).

Spawning usually occurs in April and May in streams, although variations in water temperature and stream flows may result in either earlier or later spawning. Temperature and flow cues appear to trigger spawning, with most spawning taking place at 14-20 degrees Celsius (57-68 degrees Fahrenheit) when stream flows are relatively high. Warner sucker spawn in sand or gravel beds in slow pools (White *et al.* 1990, 1991, Kennedy and North 1993). Allen *et al.* (1996) surmise that spawning aggregations in Hart Lake are triggered more by rising stream temperatures than by peak discharge events in Honey Creek. Tait and Mulkey (1993b) found young of year were abundant in the upper Honey Creek drainage, suggesting this area may be important spawning habitat and a source of recruitment for lake recolonization. The warm, constant temperatures of Source Springs at the headwaters of Snyder Creek (a tributary of Honey Creek) may provide an especially important rearing or spawning site for Warner sucker (Coombs and Bond 1980).

In years when access to stream spawning areas is limited by low flow or by physical in-stream blockages (such as beaver dams or irrigation diversion structures), Warner sucker may attempt to spawn on gravel beds along the lake shorelines. In 1990, Warner sucker were observed digging nests in 40+ centimeters (16+ inches) of water on the east shore of Hart Lake at a time when access to Honey Creek was blocked by extremely low flows (White *et al.* 1990).

Warner sucker larvae are found in shallow backwater pools or on stream margins where there is no current, often among or near macrophytes. Young of year Warner sucker are often found over deep, still water (from midwater to the surface) but also move into faster flowing areas near the heads of pools (Coombs *et al.* 1979).

Warner sucker larvae venture near higher velocities during the daytime to feed on planktonic organisms but avoid the mid-channel water current at night. This aversion to downstream drift may indicate that spawning habitats are also used as rearing grounds during the first few months of life (Kennedy and North 1993). None of the studies conducted thus far have succeeded in capturing Warner sucker younger than two years old in the Warner lakes, and it has been suggested that Warner sucker do not migrate down from the streams for two to three years (Coombs *et al.* 1979). The absence of young Warner sucker in the Warner lakes, even in years following spawning in the lakes, could be due to predation by introduced gamefishes (White *et al.* 1991).

Juvenile suckers (one to two years old) are usually found at the bottom of deep pools or in other habitats that are relatively cool and permanent such as near springs. As with adults, juvenile Warner sucker prefer areas of the streams that are protected from the higher velocities of the main stream flow (Coombs *et al.* 1979). Larval and juvenile mortality over a two month period during the summer has been estimated at 98 percent and 89 percent, respectively, although accurate larval Warner sucker counts were hampered by dense macrophyte cover (Tait and Mulkey 1993b).

White *et al.* (1991) found in qualitative surveys that, in general, adult suckers used stretches of stream where the gradient was sufficiently low to allow the formation of pools greater or equal to 50 meters (166.6 feet. These pools tended to have undercut banks, large beds of aquatic macrophytes (usually greater than 70 percent of substrate covered), root wads or boulders, a surface to bottom temperature differential of at least two degrees Celsius (at low flows), a maximum depth greater than 1.5 meters (5 feet), and overhanging vegetation (often Salix spp.). About 45 percent of these pools were beaver ponds, although there were many beaver ponds in which Warner sucker were not observed. Warner sucker were also found in smaller or shallower pools or pools without some of the above mentioned features. However, they were only found in such places when a larger pool was within approximately 0.4 kilometer (0.25 mile) upstream or downstream of the site.

Submersed and floating vascular macrophytes are often a major component of Warner sucker-inhabited pools, providing cover and harboring planktonic crustaceans which make up most of the young of year Warner sucker diet. Rock substrates such as large gravel and boulders are important in providing surfaces for epilithic (living on the surface of stones, rocks, or pebbles) organisms upon which adult stream resident Warner sucker feed, and finer gravels or sand are used for spawning. Siltation of Warner sucker stream habitat increases the area of soft stream bed necessary for macrophyte growth, but embeds the rock substrates utilized by adult Warner sucker for foraging and spawning. Embeddedness, or the degree to which hard substrates are covered with silt, has been negatively correlated with total Warner sucker density (Tait and Mulkey 1993a).

Habitat use by lake resident Warner sucker appears to be similar to that of stream resident Warner sucker in that adult Warner sucker are generally found in the deepest available water where food is plentiful. Not surprisingly, this describes much of the habitat available in Hart, Crump, and Pelican Lakes, as well as the ephemeral lakes north of Hart Lake. Most of these lakes are shallow and of uniform depth (the deepest is Hart Lake at 3.4 meters (11.3 feet) maximum depth), and all have mud bottoms that provide the Warner sucker with abundant food in the form of invertebrates, algae, and organic matter.

Warner sucker population dynamics

A population estimate of Warner sucker in streams was conducted in 1993 on the Honey Creek and Twentymile Creek drainages (Tait and Mulkey 1993b). Approximately 20 percent of available stream habitat in the Honey Creek drainage was sampled. The population within the area sampled was estimated at 77 adults, 172 juveniles, and 4,616 young of year. Approximately 60 percent of the available stream habitat in the Twentymile Creek drainage was also sampled. The population estimates within this area sampled was 2,563 adults, 2,794 juveniles, and 4,435 young of year.

As of 1996, the Hart Lake Warner sucker population was estimated at 493 spawning individuals (95 percent confidence intervals of 439-563) (Allen *et al.* 1996). Although this is the only quantified population estimate of Warner sucker ever made for Hart Lake, it is likely well below the abundances found in Hart Lake prior to the drought.

In 1997, Bosse *et al.* (1997) documented the continued existence, but reduced numbers, of Warner sucker in the Warner Lakes. The number of Warner sucker, as measured by catch per unit effort, had declined 75 percent over the 1996 results. The reduction in sucker numbers was offset by a sharp increase in the percentage composition of introduced game fish, especially white crappie and brown bullhead.

Hartzell and Popper (2002) indicated a continued reduction of Warner sucker numbers and an increase of introduced fish in Warner Lakes. The greatest number of Warner sucker captured was in Hart Lake (96% of total Warner sucker catch) with only a few Warner sucker captured in the other Warner Lakes, including Crump Lake. Suckers represented a greater percentage of the catch in relation to introduced and other native fish compared to the efforts of 1997, although a smaller total number of sucker were captured than in 1997. This was the first year since 1991 that native fish made up a smaller percentage of the catch than introduced fish.

Distribution

Between 1977 and 1991, eight studies examined the range and distribution of the Warner sucker throughout the Warner Valley (Kobetich 1977, Swenson 1978, Coombs *et al.* 1979, Coombs and Bond 1980, Hayes 1980, White *et al.* 1990, Williams *et al.* 1990, White *et al.* 1991). These surveys have shown that when adequate water is present, Warner sucker may inhabit all the lakes, sloughs, and potholes in the Warner Valley. The documented range of the sucker extended as far north into the ephemeral lakes as Flagstaff Lake during high water in the early 1980's, and again in the 1990's (Allen *et al.* 1996). The Warner sucker population of Hart Lake was intensively sampled to salvage individuals before the lake went dry in 1992.

Stream resident populations of Warner sucker are found in Honey Creek, Snyder Creek, Twentymile Creek and Twelvemile Creek. Intermittent streams in the drainages may support small numbers of migratory suckers in high water years. No stream resident Warner sucker have been found in Deep Creek since 1983 (Smith *et al.* 1984, Allen *et al.* 1994), although a lake resident female apparently trying to migrate to stream spawning habitat was captured and released in 1990 (White *et al.* 1990). The known upstream limit of the Warner sucker in Twelvemile Creek is through the Nevada reach and back into Oregon (Allen *et al.* 1994). However, the distribution appears to be discontinuous and centered around low gradient areas that form deep pools with protective cover. In the lower Twentymile Slough area on the east side of the Warner Valley, White *et al.* (1990) collected adult and young suckers throughout the slough and Greaser Reservoir. This area dried up in 1991, but because of its marshy character, may be important sucker habitat during high flows. Larval, young-of-year, juvenile and adult Warner sucker captured immediately below Greaser Dam suggest either a slough resident population, or lake resident suckers migrating up the Twentymile Slough channel from Crump Lake to spawn (White *et al.* 1990, Allen *et al.* 1996).

While investigating the distribution of Cowhead Lake tui chub, Scoppettone and Rissler (2001) discovered a single juvenile Warner sucker in West Barrel Creek. West Barrel Creek is a tributary to Cow Head Slough that eventually enters Twelvemile Creek at the known upper extension of suckers in the Twelvemile drainage. This discovery of a Warner sucker in the Cowhead Lake drainage is a significant range extension for Warner sucker.

Threats that currently occur to Warner sucker

Warner sucker were listed due to reductions in the range and numbers, reduced survival due to predation by introduced gamefishes in lake habitats, and habitat fragmentation and migration corridor blockage due to stream diversion structures and agricultural practices. Since the time of listing, it has been recognized that habitat modification, due to both stream channel degradation and overall reduced watershed function has worsened and the status and viability of the Warner sucker has declined. Signs of stream channel and watershed degradation are common in the

Warner Valley, and include fences hanging in mid-air because stream banks have collapsed beneath them, high cutbanks on streams, damaged riparian zones, bare banks, and large sagebrush flats where there were once wet meadows (White *et al.* 1991).

Adult suckers that have spawned and are moving downstream can be diverted from the main channel to become lethally trapped in unscreened irrigation canals. Larval, postlarval, young of year, and juvenile suckers are probably also lethally diverted into unscreened irrigation canals. In high water years, the amount of water diverted from Warner Valley streams may be only a small portion of the total flow, but in drought years, total stream flows often do not meet existing water rights, and so entire streams may be diverted. Over a series of drought years, reduced flows can cause drops in lake levels and sometimes, especially in conjunction with lake pumping for irrigation, cause complete dry-ups, as was the case with Hart Lake in 1992.

Although the native species composition in the Warner basin included some piscivorus fishes, like the Warner Valley redband trout (*Oncorhynchus mykiss sp.*), the introduction of exotic gamefish disrupted this balance and the native ichthyofauna has suffered. In the early 1970s, ODFW stocked white crappie (*Pomoxis annularis*), black crappie (*P. nigromaculatus)*, and largemouth bass (*Micropterus salmoides*), in Crump and Hart Lakes. Prior to this, brown bullhead (*Ameiurus nebulosus*) and non-native rainbow trout were introduced into the Warner Valley. The adults of all five piscivorus fish species feed on Warner sucker to varying degrees.

Brown bullhead are bottom oriented omnivores (Moyle 1976) that may compete directly with Warner sucker for the same food sources. Bullhead may also prey on sucker eggs in the lower creek or lake spawning areas, as well as on sucker larvae and juveniles. Young crappie probably eat many of the same zooplankton and other small invertebrates that young suckers depend on. Habitat use by young Warner sucker remains poorly understood, but there may be competition between suckers and other fishes for what scarce cover resources are available.

With few exceptions, designated Warner sucker critical habitat is excluded from grazing and other land use authorizations analyzed in the Lakeview Resource Area Resource Management Plan. The one exception is on the Deppy Creek/ Honey Creek confluence where a water gap allows stock access. The other exception is in the 0207 allotment on Twentymile Creek. This area is not occupied by Warner sucker and is an intermittent, rock-armored channel. These areas area covered by the ten-year programmatic opinion on grazing issued by the Service in 1997.

<u>Environmental Baseline</u>

The Lakeview Resource Area Resource Management Plan action area is considered to be the Honey/Deep/Twentymile Creeks watershed. This area has discontinuous land ownership and the wide range of human impacts across ownership boundaries. The Warner sucker Environmental Baseline summary is based on the Lakeview Resource Area Resource Management Plan BA.

Status of Warner sucker within the action area

Within the Management Plan action area, Warner sucker inhabit lakes, sloughs, and potholes in the Warner Valley, including the canal north of Hart Lake, Hart Lake, Crump Lake, Anderson

Lake, Swamp Lake, Mugwump Lake, Greaser Reservoir, Honey Creek, Snyder Creek, Twentymile Creek and Twelvemile Creek. A majority of Warner sucker habitat is located in waterways managed by the Lakeview BLM.

Between 1987 and 1991, five consecutive drought years prompted resource agencies to plan a Warner sucker salvage operation and establish a refuge population of Warner sucker at the Service's Dexter National Fish Hatchery and Technology Center (Dexter), New Mexico. Salvage operations consisted of intensive trap netting in Hart Lake to collect Warner sucker, then transportation of the captured fish to a temporary holding facility at ODFW's Summer Lake Wildlife Management Area (Summer Lake). The suckers were held at Summer Lake until September 1991, when 75 adults were recaptured and transported to Dexter.

While being held at Summer Lake, Warner sucker spawned successfully, leaving an estimated 250+ young in the Summer Lake holding ponds. The young suckers survived, growing approximately 85 millimeters (3.3 inches) during their first summer and reaching sexual maturity at the age of only two years. Warner sucker larvae were observed in the ponds during the summer of 1993, just over two years after the original wild suckers from Hart Lake were held there. Approximately 30 of the two year-old suckers were captured and released in Hart Lake in September 1993. In June 1994, over 100 10-17.5 centimeter (4-7 inch) Warner sucker were observed in the Summer Lake ponds. In 1996, nine adult fish were observed in these ponds along with about 20 larvae.

The suckers taken to Dexter were reduced from 75 to 46 individuals between September 1991 and March 1993, largely due to *Lorna* (anchor worm) infestation. In March 1993, the 46 survivors (12 males and 34 females) appeared ready to spawn, but the females did not produce any eggs. Between March 1993 and March 1994, *Lorna* further reduced the population to 20 individuals (5 males and 15 females) (Service 1998a). In May 1994, the five males and seven of the females spawned, producing a total of approximately 175,000 eggs. However, for reasons that are not clear, none of the eggs were successfully fertilized. The remaining 20 fish at Dexter died in 1995 (Service 1998a). In November of 1995, approximately 65 more suckers from Summer Lake were transferred to Dexter for spawning purposes but as yet no attempts to spawn these fish have occurred.

Fish passage improvements

In 1991, BLM installed a modified steep-pass Denial fish passage facility on the Dyke diversion on lower Twentymile Creek. The fishway is intended to re-establish a migration corridor, and allow access to high quality spawning and rearing habitats. The Dyke diversion structure is a 1.2 meter (4 feet) high irrigation diversion that was impassable to Warner sucker and redband trout before the fishway was installed. It blocked all migration of fishes from the lower Twentymile Creek, Twentymile Slough and Greaser Reservoir populations from moving upstream to spawning or other habitats above the structure. To date, no suckers have been observed or captured passing the structure, but redband trout have been observed and captured in upstream migrant traps.

An evaluation of fish passage alternatives has been done for diversions on Honey Creek which identifies the eight dams and diversions on the lower part of the creek that are barriers to fish migration (Campbell-Craven Environmental Consultants 1994). In May 1994, a fish passage structure was tested on Honey Creek. It consisted of a removable fishway and screen. The ladder immediately provided passage for a small redband trout. These structures were removed by ODFW shortly after their installation due to design flaws that did not pass allocated water.

Warner sucker research

Research through 1989 summarized in Williams *et al.* (1990) consisted of small scale surveys of known populations. Williams *et al.* (1990) primarily tried to document spawning and recruitment of the Hart Lake population, define the distributional limits of the Warner sucker in the streams, and lay the groundwork for further studies. White *et al.* (1990) conducted trap net surveys of the Anderson Lake, Hart Lake, Crump Lake, Pelican Lake, Greaser Reservoir, and Twentymile Slough populations. A population estimate was attempted for the Hart Lake population, but was not successful. Lake spawning activity was observed in Hart Lake, though no evidence of successful recruitment was found.

White *et al.* (1991) documented the presence of suckers in the Nevada reach of Twelvemile Creek. This area had been described as apparently suitable habitat by Williams *et al.* (1990), but suckers had not previously been recorded there.

Kennedy and North (1993) and Kennedy and Olsen (1994) studied sucker larvae drift behavior and distribution in streams in an attempt to understand why recruitment had been low or nonexistent for the lake morphs in previous years. They found that larvae did not show a tendency to drift downstream and theorized that rearing habitat in the creeks may be vital to later recruitment.

Tait and Mulkey (1993a,b) investigated factors limiting the distribution and abundance of Warner sucker in streams above the man-made stream barriers. The detrimental effects of these barriers are well-known, but there may be other less obvious factors that are also affecting the suckers in streams. These studies found that general summertime stream conditions, particularly water temperature and flows, were poor for most fish species. Recent studies have concentrated on population estimates, marking fish from Hart Lake and monitoring the recolonization of the lakes by native and non-native fishes (Allen *et al.* 1995a,b, Allen *et al.* 1996).

Federal land management

The Federal agencies responsible for management of the habitat in the Warner Basin have consulted on activities that might impact the Warner sucker. On May 21, 1995, the BLM, USFS, NOAA-Fisheries and the Service signed the Streamlining/Consultation Guidelines to improve communication and efficiency between agencies. In the Warner Basin, the outcome of streamlining has been regular meetings between the Federal agencies conducting and reviewing land management actions that may affect Warner sucker. These meetings have greatly improved the communication among agencies and have afforded all involved a much better understanding of issues throughout the entire watershed. As a result of close coordination, the Forest Service and BLM have modified many land management practices, thus reducing negative impacts, and in many cases bringing about habitat improvements to Warner sucker and Warner Valley redband trout.

Since the listing of Warner sucker as threatened in 1985, the Lakeview Resource Area has completed numerous consultations on BLM actions affecting Warner sucker. The following lists the subject and year the consultation was completed: Habitat Management Plan for the Warner Sucker 1985; Fort Bidwell-Adel County road realignment 1987; Warner Wetlands Habitat Management Plan 1990; relocation of Twentymile stream gauge 1993; Lakeview BLM grazing program 1994; reinitiation of consultation on grazing program 1995; Noxious Weed Control Program 1996; reinitiation of consultation on grazing program 1996; informal consultation on guided fishing activities 1997; reinitiation of consultation on grazing program and consultation on a number of small non-grazing projects 1997; reinitiation of consultation on grazing program 1999; informal consultation on Long Canyon Prescribed Fire 1999; grazing permit renewal concurrence 1999; reinitiation of consultation on grazing program 2000; and reinitiation of consultation on grazing program 2001.

In 1994, Lakeview Resource Area determined that ongoing site-specific livestock grazing actions were likely to adversely affect Warner sucker in the Warner Valley Watersheds and has, to date, consulted under recurring biological opinions with the Service. Present grazing prescriptions and monitoring protocols are in accordance with biological opinions issued by the Service, and results of grazing monitoring appear annually in reports to the Service. Consultation for Lakeview Resource Area's grazing activities has been reinitiated due to changes in the action, changes due to new information, and for failure to comply with terms and conditions of the biological opinions.

Effects of the Action

Effects to Warner sucker from construction activities will include the use of a hydraulic dredge or "mudcat". The dredge will operate more than 200 m from the island under construction. Up to 19,400 cy of material will be dredged from the lake bottom. This effort is expected to impact up to 3-6 ac of lake-bottom, from 2-4 ft deep. We are assuming all Warner suckers using that 6 ac of lake-bottom will be injured or killed if they get sucked through the dredge. Warner sucker density estimates for Crump Lake are not available, making any quantification of potential capture by terns difficult. However, Crump Lake is expected to hold far fewer Warner sucker than Hart Lake, and density estimates conducted at Hart Lake indicate up to 500 suckers may be present. If Crump Lake by itself, surveyed at 12+ miles (7,680+ ac) (USGS 1971), had 500 suckers evenly distributed, there would be approximately 0.07 fish per acre, and 0.4 fish per 6 acres. Although Warner sucker are not likely to be evenly distributed in Crump Lake, it is likely that very few fish would be caught up in the dredge.

Forage surveys for the Crump Lake Caspian tern colony have indicated that Warner sucker is likely an occasional if not rare prey for Caspian terns (Roby *et al.* 2003). Warner suckers are known to exhibit several behaviors that likely reduce this exposure risk to predation. As discussed in the Status of the Species section above, Warner sucker shows a strong affinity for lake and stream bottoms, especially later in the season as the water turns warmer. Nevertheless, a Warner sucker will likely be rarely captured by foraging Caspian terns.

Based on the existing data, it would be difficult to estimate with great confidence the number of Warner suckers that could be captured by Caspian terns. The one foraging study identified only one Warner sucker captured (colony of 71 tern pairs; 1,378 fish captures over 12 weeks), representing only 0.07 percent of the tern diet (Roby *et al.* 2003; Roby, pers. comm. 2004). That data suggests that, over a twelve week period, approximately 10 suckers may have been captured by the 2003 colony of 71 pair of Caspian terns. Extrapolation of this data to estimate the percentage of suckers in the diet of 300 Caspian tern pairs (upper DEIS estimate), would result in an estimate of 42 suckers captured in 12 weeks. However, the study was conducted over 12 weeks and the terns would be present for up to 24 weeks. The Roby *et al.* study may indicate that Warner suckers were not available (began spending most of their time on the bottom) after the first week of the study, which would be consistent with their seasonal behavior as discussed above. Given the density of Warner suckers in Crump Lake, as discussed above, and the tendency for suckers to exhibit more cryptic behavior as the waters warm, it seems highly unlikely that as many as 42 suckers would be consumed, even during the entire 24 weeks that the terns could remain present.

<u>Cumulative Effects</u>

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. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act. We are unaware of any non-Federal actions affecting bull trout that are reasonably certain to occur in the action area considered by this opinion.

<u>Conclusion</u>

The proposed action is likely to adversely affect Warner sucker at Crump Lake. Based on the above analysis, we believe an incremental increase/decrease in capture of Warner sucker relative to the number of Caspian terns present in any year is likely to occur in the action area, and the construction of the island may result in a small amount of harm to suckers. Therefore, the action is likely to result in unavoidable adverse affects to Warner sucker.

California Brown Pelican

<u>Status of the Species</u>

The brown pelican was federally listed as endangered in 1970 (35 *FR* 16047). The brown pelican recovery plan describes the biology, reasons for decline, and the actions needed for recovery of brown pelicans along the Pacific coast (Service 1983).

The California brown pelican (*Pelecanus occidentalis californicus*), one of six recognized subspecies of the brown pelican (Wetmore 1945) which occur in tropical and subtropical waters of the Pacific and Atlantic oceans, is the subspecies that occurs in the action area. The breeding distribution of the California subspecies ranges from the Channel Islands of southern California southward to Isla Isabela, Islas Tres Marias and Isla Ixtapa off the coast of Mexico (Service 1983). Between breeding seasons California brown pelicans range north as far as Vancouver Island, British Columbia, Canada and south to Colima, Mexico. The eastern brown pelican (*P. o. carolinensis*) is the only other subspecies of brown pelican that occurs in North America. The breeding distribution of the eastern subspecies ranges from New Jersey south along the Atlantic coast to Florida and along the coast of the Gulf of Mexico from Florida to Texas. Post-breeding eastern brown pelicans occasionally disperse as far north as New England and Nova Scotia.

The brown pelican is a large marine bird weighing up to 4 kilograms (8 pounds) that is recognized by its large bill, a prominent, unfeathered throat pouch, and a wingspread up to 7 feet (Sykes 1983). Adults in non-breeding plumage have a white head and neck. During the breeding season, the hindneck and nape are dark brown. The body and wings are grayish brown, and the primaries and secondaries are dark brown. The bill is gray, and the throat pouch is black. Immature brown pelicans are mostly brown with a dark neck and head and white belly (Sykes 1983). The California brown pelican can be distinguished from the eastern brown pelican by having a larger size and darker hindneck while in breeding plumage (Wetmore 1945) and a bright red gular pouch during courtship and egg-laying period. They are rather clumsy on land and fly with their necks folded, heads resting on their backs, using slow, powerful wingbeats.

Life History

Brown pelicans feed almost entirely on surface-schooling fish caught by plunge diving in coastal waters. Feeding is often concentrated in relatively shallow waters of less than 50 fathoms (300 feet) (Gress *et al.* 1980). Their diet consists mainly of northern anchovy (*Engraulis mordax*), Pacific sardine (*Sardinopus sagax*), and other surface-schooling fish (Anderson *et al.* 1980, 1982, Anderson and Gress 1984). Sometimes brown pelicans dive from as high as sixty to seventy feet, although a height of thirty feet is more normal. Brown pelicans are often attracted to a particular foraging area by the feeding activities of other species and by other pelicans (Gress *et al.* 1980). As soon as pelicans begin plunging and feeding, they quickly attract the attention of others. Pelicans will often congregate in large numbers at these prey concentrations until the food source is gone (Gress *et al.* 1980). Brown pelicans are rarely found away from salt water and do not normally venture more than 32 kilometers (20 miles) out to sea.

Brown pelicans are social and gregarious. Males and females, juveniles and adults, congregate in large flocks for much of the year. Brown pelicans nest in colonies on small coastal islands that are free of mammalian predators and human disturbance and are associated with an adequate and consistent food supply. Nest sites are generally on steep, rocky slopes. Nests are constructed on the ground or in brush of whatever materials are available, including grasses, sticks, feathers, and seaweed. The nesting chronology of brown pelicans varies from year-to-year, although most nesting activity occurs between February and October. Nesting may be synchronous or may consist of sub-colony units breeding asynchronously over a period of several months (Service 1983).

A brown pelican pair attends a clutch of two or three eggs, which are incubated alternatively by both parents. Young are born altricial and are fed by both parents. Chicks take about 13 weeks to fledge, at which time they weigh about 20 percent more than adults. Brown pelicans reach breeding age in about three to five years.

Offshore habitat associated with island colony sites is also essential habitat for brown pelicans. Brown pelicans are dependent on food resources near the colony site during the breeding season. The offshore zone within 30 to 50 kilometers (18 to 30 miles) of the colony is critical to pelican food supplies, especially when young are being fed (Anderson and Gress 1984). Waters near colony sites are also important for wintering migratory birds and for newly-fledged young when they begin feeding for themselves. Offshore aquatic habitat, including the abundance and availability of brown pelican food resources, is a major factor in determining the population status of brown pelicans and the degree of breeding success (Service 1983).

During the non-breeding season, which varies between colonies but typically extends from July to January, brown pelicans roost communally. Roosting sites and loafing areas are essential habitat for breeding brown pelicans and non-breeding local and Mexican migrants. Brown pelicans are tropically-derived seabirds that have wetable plumage so they must have terrestrial roost sites to dry wet plumage after feeding or swimming (Jaques and Anderson 1987). Roost sites are also important for resting and preening. The essential characteristics of roosts include: nearness to adequate food supplies; presence of physical barriers to predation and disturbance; sufficient surface space for individuals to interact normally; and adequate protection from adverse environmental factors such as wind and surf (Jaques and Anderson 1987). Offshore rocks and islands; river mouths with sand bars; and breakwaters, pilings, and jetties are important roosting sites.

Adult brown pelicans are efficient predators, and therefore many individuals spend a considerable portion of the day on land and all congregate at night roosts during the dark hours (Jaques and Anderson 1987). Pelican concentrations shift in response to prey distributions. The dispersion of suitable roost sites influences bioenergetic considerations, not only for shelter, thermoregulation, and plumage maintenance, but for efficient travel time to food resources (Jaques and Anderson 1987). Communal roosts may also provide increased protection from potential predators, act as centers for social facilitation of food finding, and other functions yet to be identified.

Distribution and Population Status

Although nest predation may be a problem, adult brown pelicans have few natural enemies. Nests are sometimes destroyed by hurricanes, flooding, or other natural disasters; however, the biggest threat to brown pelican survival has historically been related to human activities. Brown pelicans experienced widespread reproductive failures in the 1960s and early 1970s. Much of the failure was attributed to eggshell thinning caused by high concentrations of DDE (dichlorodiphenyldichloroethylene), a metabolite of DDT (dichlorodiphenyltrichloroethane). Other factors implicated in the decline of this species include human disturbance at nesting colonies and food shortages. Brown pelicans have not nested north of the Channel Islands since the species' decline in the late 1950s and early 1960s. In 1972, the Environmental Protection

Agency banned the use of DDT in the U.S. and placed restrictions on the use of other pesticides. Since then, the level of chemical contaminants in pelican eggs has decreased and brown pelican nesting success has subsequently increased. The brown pelican was the first species to apparently recover from the effects of pesticides.

Although food availability, human disturbance at breeding and roosting sites, and chronic levels of DDT in the marine environment may still be suppressing productivity, brown pelican population numbers have increased steadily in the Southern California Bight (SCB). After experiencing nearly complete reproductive failure during the late 1960s to the late 1970s, with nesting attempts often numbering less than 1,000 per year, brown pelican productivity started to improve following the ban on DDT. The increase in productivity that started in 1974 was correlated with an increase in eggshell thickness. Since 1974, food availability has become the most important limiting factor influencing brown pelican breeding success within the SCB and both nesting attempts and productivity can fluctuate greatly year-to-year based on the availability of small surface-schooling fishes.

Nesting colonies of the brown pelican on the Pacific coast range from the Channel Islands in the SCB south to the islands off Nayarit, Mexico. Prior to 1959, intermittent nesting was observed as far north as Point Lobos in Monterey County, California. Currently in southern California, brown pelican colonies are found only on Anacapa and Santa Barbara islands; they do not nest on any of the other Channel Islands. In addition, during 1996 and 1997 a small number of brown pelicans nested near Obsidian Butte at the Salton Sea.

The maximum breeding population of the California brown pelican throughout its range may number about 55,000 to 60,000 pairs. Breeding populations can be differentiated into geographically separate entities that are isolated from each other by long stretches of uninhabited coastline. About 90 to 95 percent of the California brown pelican population breeds on islands off the coast of mainland Mexico, Baja California, and in the Gulf of California (Anderson 1983, Service 1983). In southern California, brown pelican colonies are found only on Anacapa and Santa Barbara islands; they do not nest on any of the other Channel Islands. The breeding population of brown pelicans in southern California is estimated at 4,500 to 6,000 pairs. Some genetic exchange occurs among colonies by the recruitment of new breeders. The largest breeding group is located on the Gulf of California, comprising approximately 68 percent of the total breeding population.

Dispersal between breeding seasons ranges from British Columbia, Canada, to southern Mexico and possibly to Central America. Post breeding dispersal patterns depend largely on oceanographic conditions, which in turn influence food availability (Anderson and Anderson 1976). During the non-breeding season, which varies between colonies but typically extends from July to January, the number of brown pelicans along the Pacific coast of the United States increases. Although only two breeding colonies exist along the coast of California, the majority of brown pelicans seen foraging along coastal California, Oregon, and Washington likely come from Mexico, as those pelicans tend to be more mobile. At this time of year, as many as 75,000-90,000 brown pelicans may occur along the California coast (del Hoyo *et al.* 1992). There are at least 32 important brown pelican roosting areas between Point Reyes and Point Conception (Jaques and Anderson 1988).

Brown pelicans have recently extended their non-breeding range north along the Pacific coast. Range expansion began in the early to mid 1970's but seasonal migration of thousands of pelicans into Washington did not become regular until 1985 (Jaques 1994). Brown pelicans were essentially absent north of 45° north longitude during most of the 20th century, although accounts from early naturalists suggest that brown pelicans were abundant in Washington in the 1800's. The recent range expansion is thought to be due to the recovery of the SCB breeding population and a long-term warm water regime in the north Pacific Ocean beginning in 1976 (Jaques 1994). The Oregon and Washington coastlines have become an important region for non-breeding brown pelicans. Nearly 13,000 brown pelicans were observed there during surveys in 1991 (Jaques 1994).

Environmental Baseline

The factors presenting risks to naturally-reproducing populations of least tern are numerous and varied. A number of documents have addressed the history of human activities, present environmental conditions, and factors contributing to the decline of least tern. For example, the Service has prepared or assisted in the preparation of numerous documents focusing on status and recovery of these species, including the *California Brown Pelican Recovery Plan* (Service 1983), Programmatic Environmental Impact Statement/Report for the CALFED Bay-Delta Program (July 2000), and the Programmatic Environmental Impact Statement for the Central Valley Project Improvement Act (October 1999). All provide excellent summaries of historical and recent environmental conditions in the coastal and San Francisco Bay areas of California. For the purposes of this document, a general description of the environmental baseline for the least tern listed under the Act is based on a summarization of these documents.

Brown pelicans can arrive in northern California, after their breeding season is completed, as early as April or May, but the majority of birds typically arrive in July and stay through September (D. Jaques-Strong *in* Symposium Proceedings). Breakwater Island, located in the offshore waters just south of the western end of Naval Air Station (NAS) Alameda, supports the most significant loafing/night roost for brown pelicans in San Francisco Bay. Typically, Breakwater Island supports more than 400 brown pelicans during the non-breeding season, but in July 1997, the island supported more than 1,000 brown pelicans. Open water around Breakwater Island and in other parts of San Francisco Bay, including areas around Oakland Harbor, provide foraging, loafing and roosting habitat for brown pelicans. According to the *Biological Assessment for the Berths 55-58 and Oakland Harbor Navigation Improvement Projects* prepared by Entrix, Inc. (December 9, 1997, revised April 24, 1998) 16 brown pelicans were recorded near the project site at Oakland Harbor during surveys performed in 1997.

Profound alterations to the estuarine habitat of San Francisco Bay began with the discovery of gold in the middle of the 19th century. Dam construction, water diversion, hydraulic mining, and the diking and filling of tidal marshes soon followed, launching San Francisco Bay into the era of rapid urban development and coincident habitat degradation. In general, the human activities that have affected these avian species and their habitats within the action area consist of: (1) loss of adequate nesting habitat; (2) land use activities such as urban development and landfill that degrade and transform aquatic habitat; (3) pollution; (4) dam construction and water development activities that can affect water quantity, timing, and quality in San Francisco Bay; (5) introduction of non-native species; and (6) ecosystem restoration.

Historically, the tidal marshes of San Francisco Bay were a highly productive estuarine environment providing sufficient fish prey species for the brown pelican. Land use activities since the 1850's associated with urban development, mining, and agriculture have significantly altered habitat quantity and quality in San Francisco Bay, and contributed to ecosystem degradation.

Urbanization has been a major influence on the land surrounding the estuary. In the past 150 years, the diking and filling of tidal marshes have decreased the surface area of San Francisco Bay by 37 percent. More than 500,000 acres of the estuary's historic tidal wetlands have been converted to farms, salt ponds, and urban uses. Less than 45,000 acres of the estuary's historic tidal marshes remain intact, a reduction of 92 percent (San Francisco Estuary Project 1992). Today, nearly 30 percent of the land in the nine counties surrounding San Francisco Bay is urbanized. The increase in urban land reflects the growth of the human population. There are now more than 7.5 million individuals living in the 12 Bay Area counties, making the region the fourth most populous metropolitan area in the United States. These changes have reduced the acreage of valuable farm land, wetlands, and riparian areas, and have increased pollutant loadings to the estuary. Installation of docks, shipping wharves, marinas, and miles of rock rip rap for shoreline protection has also contributed greatly to habitat degradation within the estuary.

Industrial, municipal, and agricultural wastes have been discharged into the waters of San Francisco Bay with major historical point sources including wastes from fish and fruit/vegetable canneries, and municipal sewage. The large-scale pollution of the estuary was partially relieved by the passage of the Clean Water Act in 1972, resulting in the construction of sewage treatment plants in all cities. Non-point sources of pollution, such as urban and agricultural runoff, continue to degrade water quality.

Hydropower, flood control, and water supply dams of the CVP, SWP, and other municipal and private entities have affected water quantity, timing, and quality in San Francisco Bay. These altered streamflows and the resulting inflow to San Francisco Bay have effected the natural ecosystem. Depleted inflow to San Francisco Bay has contributed to higher water temperatures and lower dissolved oxygen levels. Additionally, the seasonal distribution of freshwater inflow differs from historical patterns. The magnitude and duration of peak flows during the winter and spring are significantly reduced by water impoundment in upstream reservoirs. During the summer and early fall, inflow to San Francisco Bay may be greater than historical levels dne to deliveries of municipal and agricultural water supplies. Overall, present day water management practices in the Central Valley reduce natural flow variability by creating more uniform flows year-round that diminish natural channel forming, riparian vegetation growth, and food web functions.

As native fishes became depleted in the late 19th century, non-native species were brought in to the Bay and delta, including American shad, striped bass, common carp, and white catfish. As their populations boomed, those of native fishes declined further. Introduction of non-native species accelerated in the 20th century through deliberate introductions of fish and unintended introductions of competitive invertebrates through ballast water of ships. Establishment of non-native species was probably facilitated by altered hydrologic regimes and reduction in habitats for native species. The introduction and spread of non-native species in San Francisco Bay has affected native species, including listed avian species, by competing with them for food and habitat, and preying on native species.

Preliminary, significant steps towards the largest ecological restoration project yet undertaken in the United States have occurred during the past five years and continue to proceed in California's Central Valley. The CALFED Program and the CVPIA's AFRP, in coordination with other Central Valley and Bay Area efforts, have implemented numerous habitat restoration actions. A few of these restoration projects, primarily land acquisition and wetland restoration, include actions within the San Francisco Bay area. Restoration of wetland areas typically involves flooding lands previously used for agriculture, thereby creating additional wetland areas that could serve as rearing habitat for fish prey species of the brown pelican.

Effects of the Action

The effects of the proposed action on brown pelican may include construction, maintenance and monitoring related impacts. In addition, pelican behavior and forage success may be affected by increased presence of foraging Caspian tern in the 12 mile radius surrounding each of East Sand Island, Brooks Island, Hayward Regional Shoreline, and Don Edwards NWR. Brown Pelicans do not nest in the Columbia River or San Francisco Bay estuaries, but are known to roost on land in these regions from July through September. Their concentrations shift in response to prey distributions; the dispersion of suitable roost sites influences bioenergetic considerations, including efficient travel time to food resources.

East Sand Island supports a large roosting colony of brown pelicans. Up to 10,800 birds were observed roosting on the western end of the island (opposite end from the tern colony site) in 2002. In San Francisco Bay, Breakwater Island, located in the offshore waters just south of the western end of NAS Alameda, supports the most significant loafing/night roost for brown pelicans. Typically, Breakwater Island can support more than 400 - 1,000 brown pelicans during the non-breeding season. Open water around Breakwater Island and in other parts of San Francisco Bay, including areas around Oakland Harbor, provide foraging, loafing and roosting habitat for brown pelicans. These roost sites are miles from proposed construction, maintenance and monitoring activities at Brooks Island, Hayward shoreline, and Don Edwards NWR, and are not likely to be affected by those activities. There are no known roost or loafing sites on Brooks Island, the construction site closest to Breakwater Island. Smaller groups of pelicans loafing adjacent to the construction sites, that may leave their roost in response to construction noise or other activities, are expected to move on to other loafing/foraging areas if a flight response is triggered. None of these impacts are likely to result in significant disruptions of pelican behavior that would lead to its inability to obtain adequate forage or shelter.

Brown pelican and Caspian tern both occur in the Columbia River estuary and San Francisco Bay area and feed on small surface schooling fish such as anchovies. Brown pelicans are often attracted to a particular foraging area by the feeding activities of other birds, and as soon as pelicans begin plunging and feeding, they quickly attract the attention of others. Pelicans are gregarious and efficient predators, so less likely to be outcompeted or otherwise affected by the increased presence of terns for the abundant forage resources in the Columbia River estuary and San Francisco Bay. Because pelicans will not be competing with Caspian terns for nesting areas and because terns will not significantly alter the prey base available for brown pelicans, the proposed project is not likely to adversely affect brown pelicans.

Conclusion

The proposed action may effect, but will not likely adversely affect California brown pelicans at East Sand Island, Brooks Island, Hayward Regional Shoreline, and Don Edwards NWR. We make this determination on the following reasons: (1) Construction, maintenance, and monitoring activities will not significantly affect any significant brown pelican roost sites or foraging behavior; (2) Caspian terns are not likely to out compete California brown pelicans for foraging space or foraging resources; and (3) since pelican do not nest in the Columbia River estuary or San Francisco Bay, there will be no competition for nest sites.

California Clapper Rail

Status of the Species

The California clapper rail was federally listed as endangered in 1970 (35 FR 16047). Critical habitat has not been proposed or designated. This subspecies is one of three subspecies in California listed as endangered under the Act. The other subspecies include the light-footed clapper rail (*R. l. levipes*) which is found in tidal marshes in southern California and northwestern Baja California, and the Yuma clapper rail (*R. l. yumanensis*) which is restricted to the Colorado River basin. A detailed account of the taxonomy, ecology, and biology of the California clapper rail is presented in the *Salt Marsh Harvest Mouse & California Clapper Rail Recovery Plan* (Service 1984) and the references cited therein. The California clapper rail is a fully protected species under California law (See California Fish and Game Code Section 3511).

The California clapper rail is endemic to tidally influenced salt and brackish marshes of California. Historically, the California clapper rail occurred in tidal marshes along California's coast from Morro Bay, San Luis Obispo County, to Humboldt Bay, Humboldt County. Currently, California clapper rails are known to occur in tidal marshes in the San Francisco Estuary (San Francisco, San Pablo, Grizzly, Suisun and Honker bays).

The California clapper rail is distinguishable from other rails by its large body size of 32-47 centimeters from bill to tail, and weighs approximately 250-350 grams. It has an orange bill, a rufous breast, black and white barred flanks, and white under tail coverts (Albertson and Evens 2000). California clapper rails are sexually dimorphic; the males are slightly larger than females (Garcia 1995). Juveniles have a pale bill and dark plumage. California clapper rails are capable of producing several vocalizations, most common of which is a series of keks or claps.

California clapper rails are typically found in the intertidal zone and sloughs of salt and brackish marshes dominated by pickle weed, Pacific cord grass (*Spartina foliosa*), gumplant (*Grindelia spp.*), salt grass, jaumea (*Jaumea carnosa*) and adjacent upland refugia. They may also occupy habitats with other vegetative components, which include, but are not limited to bulrush *(Scirpus americanus* and *S. maritimus*), cattails (*Typha* spp.), and Baltic rush (*Juncus balticus*).

Evens and Page (1983) concluded from research in a northern San Francisco Bay marsh that the California clapper rail breeding season, including pair bonding and nest construction, may begin as early as February. Field observations in south San Francisco Bay marshes suggest that pair formation also occurs in February in some areas (J. Takekawa, pers. comm.). The end of the breeding season is typically defined as the end of August, which corresponds with the time when eggs laid during renesting attempts have hatched and young are mobile. Harvey (1988) and Foerster *et al.* (1990) reported mean clutch sizes of 7.27 and 7.47 for California clapper rails, respectively. The California clapper rail builds a bowl shaped platform nest of marsh vegetation and detritus (DeGroot 1927, Foerster *et al.* 1990, Garcia 1995). The California clapper rail typically feeds on benthic invertebrates, but its diet is wide ranging, and includes seeds, and occasionally small mammals such as the salt marsh harvest mouse.

<u>Environmental Baseline</u>

An estimated 40,191 acres of tidal marshes remained in 1988 of the 189,931 acres of tidal marsh that historically occurred in the San Francisco Bay Estuary; this represents a 79 percent reduction from historical conditions (Goals Project 1999). Furthermore, a number of factors influencing remaining tidal marshes limit their habitat values for California clapper rails. Much of the east San Francisco Bay shoreline from San Leandro to Dumbarton Bridge is rapidly eroding, and many marshes along this shoreline could lose their California clapper rail populations in the future, if they have not already. In addition, an estimated 600 acres of former salt marsh along Coyote Creek, Alviso Slough, and Guadalupe Slough, has been converted to fresh- and brackish-water vegetation due to freshwater discharge from wastewater facilities in the southern part of San Francisco Bay and is of lower quality for California clapper rails. This conversion has at least temporarily stabilized as a result of the drought since the early 1990s. The introduction of non-native, invasive plant species such as smooth cordgrass (*Spartina alterniflora*) and its hybrids into tidal wetlands within the Estuary is potentially impacting California clapper rails by reducing the amount of foraging habitat within tidal channels. The suitability of many marshes for California clapper rails is further limited, and in some cases precluded, by their small size, fragmentation, and lack of tidal channel systems and other micro-habitat features. These limitations render much of the remaining tidal marsh acreage unsuitable or of low value for the species.

Throughout the San Francisco Estuary, the remaining California clapper rail population is impacted by a suite of mammalian and avian predators. At least 12 native and 3 non-native predator species are known to prey on various life stages of the California clapper rail (Albertson 1995). Artificially high local populations of native predators, especially raccoons and skunks, result as development occurs in the habitat of these predators around the San Pablo and San Francisco bay margins (J. Takekawa, pers. comm.). Encroaching development not only displaces lower order predators from their natural habitat, but also adversely affects higher order predators, such as coyotes, which would normally limit population levels of lower order native and non-native predators, especially red foxes (Albertson 1995). Hunting intensity and efficiency by raptors on California clapper rails also is increased by electric power transmission lines, which cris-cross tidal marshes and provide otherwise-limited hunting perches and nesting opportunities (J. Takekawa, pers. comm.). Non-native Norway rats (*Rattus norvegicus*) long have been known to be effective predators of California clapper rail nests (DeGroot 1927, Harvey 1988, Foerster *et al.* 1990). Placement of shoreline riprap, levees, buildings, and landfills favor rat populations, which results in greater predation pressure on California clapper rails in certain marshes. Raven (*Corvus corax*) populations have recently increased dramatically within the Estuary and evidence of egg predation by this species has been detected (Joy Albertson, pers. comm.). Feral cats also represent another predation threat on adult and young California clapper rails near residential areas and landfills (Joy Albertson, pers. comm.). These predation impacts are exacerbated by a reduction in high marsh and natural high tide cover in marshes.

The proliferation of non-native red foxes into tidal marshes of southern San Francisco Bay since 1986 has had a profound effect on California clapper rail populations. As a result of the rapid decline and almost complete elimination of California clapper rail populations in certain marshes, the San Francisco Bay NWR implemented a predator management plan in 1991 (Foerster and Takekawa 1991) with an ultimate goal of increasing rail population levels and nesting success through management of red fox predation. This program initially was successful in increasing the overall south San Francisco Bay populations from an all-time low (see below); however, it has been difficult to effectively conduct predator management over such a large area as the south San Francisco Bay, especially with the many constraints associated with conducting the work in urban environments (J. Takekawa, pers. comm.).

Predator management for California clapper rails is not being regularly practiced in San Pablo and Suisun bays, and California clapper rail populations in this area remain susceptible to red fox predation. Red fox activity has been documented along Sonoma Creek and in the bayshore marshes between Sonoma Creek and the Petaluma River (Evens 2000) and along Dutchman Slough and in Guadalcanal Village on the west side of the Napa River (J. Collins, pers. comm.). Red fox activity also have been documented along the levees at Carl's Marsh and in baylands on the east side of the Petaluma River (Peter Baye, pers. comm.). Along Wildcat Creek near Richmond, where red fox activity was observed in the mid-1990's, the rail population level in one tidal marsh area declined considerably after 1987 (J. Evens, pers. comm.). Red fox predation may be a major reason for recent decreases in California clapper rail populations within certain parts of San Pablo Bay.

Mercury accumulation in eggs is perhaps the most significant contaminant problem affecting California clapper rails in the Estuary, with south San Francisco Bay containing the highest mercury levels. Mercury is extremely toxic to embryos and has a long biological half-life. The Service collected data from 1991 and 1992 on mercury concentrations in rail eggs in the southern portion of the estuary and found that the current accumulation of mercury in rail eggs occurs at potentially harmful levels. The percentage of non-viable eggs ranged from 24 to 38 percent (mean = 29 percent) (Service, unpubl. data).

The California clapper rail was listed as endangered primarily as a result of habitat loss. The factors described above have contributed to the more recent population reduction, which has occurred since the mid-1980s. Although Gill (1978) may have overestimated the total California clapper rail population in the mid-1970s at 4,200 to 5,900 birds, surveys conducted by the California Department of Fish and Game (CDFG) and the Service estimated that the California clapper rail population approximated 1,500 birds in the mid-1980s (Harvey 1988). In 1988, the total rail population was estimated to be 700 individuals, with 400 to 500 rails in south San Francisco Bay (Foerster 1989). The total rail population reached an estimated all-time historical low of about 500 birds in 1991, with about 300 rails in south San Francisco Bay (Service unpubl. data). In response to predator management, the south San Francisco Bay rail population rebounded from this lowest population estimate to an estimated 650 to 700 individuals in 1997-98 (Service unpubl. data). Subsequently, the south San Francisco Bay population declined again the following year to about 500 individuals and remained at that level through early 2002 (Service unpubl. data). However, the south San Francisco Bay population declined further in 2002-2003 and is now estimated to be 400-500 individuals (Service unpubl. data), which represents the lowest estimated population level in this area since the late 1980's and early 1990's. A conservative estimate of the north San Francisco, San Pablo, and Suisun bay population, was 195 to 282 pairs based on a synoptic survey conducted in 1992-93 (Collins *et al.* 1994). Since then, several population centers in San Pablo Bay have declined precipitously. The population in the White Slough tidal marshes on the west side of the Napa River declined from an estimated 16-23 pairs as recent as 2000 to an estimated 2-5 pairs in 2002, while the population in the Sonoma Creek marshes declined from 13 pairs in 1992 to 0-1 pair in 2000 (Avocet Research Associates 2003). Although recent survey data are lacking for other marshes within San Pablo Bay, these areas also may have declined. As a result of declines in areas within San Pablo and San Francisco bays, the overall population may be at its lowest recorded level reached previously during the late 1980's and early 1990's. Although many factors are at work, predation by native and non-native predators, in conjunction with historic habitat loss and fragmentation, are the current known primary threats. With historic populations at Humboldt Bay, Elkhorn Slough, and Morro Bay now extirpated, the San Francisco Estuary represents the last stronghold and breeding population of this subspecies.

Effects of the Action

An important clapper rail colony occurs a few hundred yards from the Hayward Regional Shoreline and Don Edwards NWR management sites. Construction related noise and site access appear to be the largest concerns. Predation of clapper rails is not anticipated to increase due to the increase in Caspian tern presence, given that predator use of the area is already high, and existing predator control efforts are expected to avoid any new predation effects to clapper rails. To avoid adverse impacts to listed species associated with ingress and egress at the project site, a map indicating vehicle access route and helicopter flight path restrictions during construction, maintenance and monitoring activities will be provided to the Sacramento Fish and Wildlife Office for review and approval prior to initiating construction related activities at the site. For the purpose of this analysis, we are assuming that avoidance of adverse impacts can be achieved by these means.

The California clapper rail typically feeds on benthic invertebrates, but its diet is wide ranging, and includes seeds, and occasionally small mammals such as the salt marsh harvest mouse. Nevertheless, Caspian terns, who feed primarily in open water on small schooling fish, and clapper rails do not utilize the same forage base, so increased tern presence will not effect clapper rail foraging behavior or forage resource availability.

Conclusion

The proposed action may affect, but will not likely adversely affect California clapper rail. We make this determination for the following reasons: (1) Construction, maintenance, and monitoring activities, in particular vehicle and helicopter travel will not occur where they would be a threat to California clapper rail; (2) because predator control efforts will avoid increased predation pressure on California clapper rail that may result from the proposed action, and (3) because Caspian terns are not a direct predation threat to California clapper rail.

California Least Tern

Status of the Species

The least tern was federally protected as endangered on October 13, 1970 (35 FR 16047). A detailed account of the taxonomy, ecology, and biology of the least tern is presented in the approved Recovery Plan for this species (Service 1980). Supplemental or updated information is provided in the Service's July 16, 1993, Biological Opinion on the Federal Aviation Administration's authorization for proposed facilities improvements at San Diego International Airport, California, which is hereby incorporated by reference.

Least terns typically arrive at NAS Alameda in mid to late April, but have arrived as early as April 6, and depart in mid to late August each year. During this time period, least tern adults mate and select nest sites; lay, incubate, and hatch eggs; and raise young to fledglings prior to migrating south for the rest of the year. Hatchlings are typically fed from June through mid-August. Since 1977, the majority of nesting activities have occurred in the 4-acre, fenced "traditional" colony site on the western end of NAS Alameda, but prior to 1987, least tern nesting also occurred in other areas within the proposed refuge outside the traditional site area (L. Collins *in* Symposium Proceedings). Furthermore, least terns have moved their young to various locations within the buffer zone surrounding the main colony site during several breeding seasons (and on one occasion as far as about 4,000 feet northwest of the main colony site), apparently to avoid predator pressure at the main colony site. While at NAS Alameda during the breeding season, least terns forage for fish in the open water offshore of the western end of NAS Alameda, which contains extensive, generally productive foraging habitat areas. Foraging intensity has varied between different offshore areas, but has occurred in the Oakland Harbor, Seaplane Lagoon at NAS Alameda, and areas southeast, south, and west of the traditional least tern colony site. During the breeding season, least terns are central-place foragers, that is, they return regularly to a central place (the nest) from their foraging trips. Most foraging activity occurs within 2 miles of the nesting site (Atwood 1983). Having foraging places near their nests is beneficial to the terns because it reduces the energy cost of flying to the feeding site and reduces the time needed to bring a load of fish back to the nest.

According to the 1995 Caffrey report, the least tern breeding site at NAS Alameda has played a significant role in recent increases in the number of least terns throughout California. The NAS Alameda site is consistently one of the most successful sites in California. Between 1987 and 1994, the NAS Alameda site supported 5 to 6 percent of the statewide breeding population out of 35 to 40 sites each year, but produced an average of 10.6 percent of the total number of fledglings produced statewide in each of those years. In 1997, an estimated 244 pairs of least terns nested at the colony out of a total population of more than 4,000 nesting pairs at 37 breeding sites along the California and Baja California coasts. In 1997, an estimated 316 young fledged successfully at NAS Alameda; this represented 10.1 percent of the total number of fledglings produced throughout California that year. By consistently producing large numbers of fledglings each year, the colony has added large numbers of potential new breeding birds to the statewide population. Therefore, this site is considered to be one of the most important "source" populations in California serving to balance out losses at many "sink" locations throughout the State. Because of its importance for least terns, the Service plans to establish a NWR on lands at NAS Alameda that will include the traditional 4-acre least tern colony site and surrounding buffer areas.

There are two other minor least tern breeding sites in the San Francisco Bay area, the Oakland Airport and PG&E Pittsburg power plant site. The Oakland Airport site has not been used in years and the Pacific Gas and Electric Pittsburg site supports only one to four pairs each year. Therefore, the NAS Alameda site currently represents the entire San Francisco Bay area population, and is the most northern of least tern breeding colonies by about 178 miles. Because of its northern location, the NAS Alameda site is relatively unaffected during El Nino years when many southern California sites experience pronounced breeding failure resulting from limited food availability. In the most recent previous El Nino year, 1992, the NAS Alameda site supported 6 percent of the statewide number of breeding pairs, but produced 16 percent of the total statewide number of fledglings.

The 1998 season was another El Nino year, one of the most severe recorded, and least tern breeding at NAS Alameda was less successful. Only 90 young were fledged, more than a 70 percent reduction from 1997. Observations of delayed breeding, reduced fish catch, and the highest non-predator mortality of young ever observed (about 50 percent) (L. Collins, pers. comm. 1999) suggest food limitation and associated problems as a cause. In 1999, the number of nesting pairs at NAS Alameda may have declined substantially (K. Sanchez, pers. comm. 1999).

Environmental Baseline

The factors presenting risks to naturally-reproducing populations of least tern are numerous and varied. A number of documents have addressed the history of human activities, present environmental conditions, and factors contributing to the decline of least tern. For example, the Service has prepared or assisted in the preparation of numerous documents focusing on status and recovery of these species, including the *California Least Tern Recovery Plan* (Service 1980), Programmatic Environmental Impact Statement/Report for the CALFED Bay-Delta Program (July 2000), and the Programmatic Environmental Impact Statement for the Central Valley

Project Improvement Act (October 1999). All provide excellent summaries of historical and recent environmental conditions in the coastal and San Francisco Bay areas of California. For the purposes of this document, a general description of the environmental baseline for the least tern listed under the Act is based on a summarization of these documents.

San Francisco Bay

Profound alterations to the estuarine habitat of San Francisco Bay began with the discovery of gold in the middle of the 19th century. Dam construction, water diversion, hydraulic mining, and the diking and filling of tidal marshes soon followed, launching San Francisco Bay into the era of rapid urban development and coincident habitat degradation. In general, the human activities that have affected these avian species and their habitats within the action area consist of: (1) loss of adequate nesting habitat; (2) land use activities such as urban development and landfill that degrade and transform aquatic habitat; (3) pollution; (4) dam construction and water development activities that can affect water quantity, timing, and quality in San Francisco Bay; (5) introduction of non-native species; and (6) ecosystem restoration.

Only three least tern nesting areas currently exist in the San Francisco Bay area: Oakland Airport, PG&E Pittsburg power plant, and NAS Alameda. The Oakland Airport site has not been used in years and the Pacific Gas and Electric Pittsburg site supports only one to four pairs each year. The site at NAS Alameda remains one of the most important "source" populations in California, with an estimated 244 pairs of least terns nesting in the colony.

Historically, the tidal marshes of San Francisco Bay were a highly productive estuarine environment providing sufficient fish prey species for the least tern. Land use activities since the 1850's associated with urban development, mining, and agriculture have significantly altered habitat quantity and quality in San Francisco Bay, and contributed to ecosystem degradation.

Urbanization has been a major influence on the land surrounding the estuary. In the past 150 years, the diking and filling of tidal marshes have decreased the surface area of San Francisco Bay by 37 percent. More than 500,000 acres of the estuary's historic tidal wetlands have been converted to farms, salt ponds, and urban uses. Less than 45,000 acres of the estuary's historic tidal marshes remain intact, a reduction of 92 percent (San Francisco Estuary Project 1992). Today, nearly 30 percent of the land in the nine counties surrounding San Francisco Bay is urbanized. The increase in urban land reflects the growth of the human population. There are now more than 7.5 million individuals living in the 12 Bay Area counties, making the region the fourth most populous metropolitan area in the United States. These changes have reduced the acreage of valuable farm land, wetlands, and riparian areas, and have increased pollutant loadings to the estuary. Installation of docks, shipping wharves, marinas, and miles of rock rip rap for shoreline protection has also contributed greatly to habitat degradation within the estuary.

Industrial, municipal, and agricultural wastes have been discharged into the waters of San Francisco Bay with major historical point sources including wastes from fish and fruit/vegetable canneries, and municipal sewage. The large-scale pollution of the estuary was partially relieved by the passage of the Clean Water Act in 1972, resulting in the construction of sewage treatment plants in all cities. Non-point sources of pollution, such as urban and agricultural runoff, continue to degrade water quality.

Hydropower, flood control, and water supply dams of the CVP, SWP, and other municipal and private entities have affected water quantity, timing, and quality in San Francisco Bay. These altered streamflows and the resulting inflow to San Francisco Bay have effected the natural ecosystem. Depleted inflow to San Francisco Bay has contributed to higher water temperatures and lower dissolved oxygen levels. Additionally, the seasonal distribution of freshwater inflow differs from historical patterns. The magnitude and duration of peak flows during the winter and spring are significantly reduced by water impoundment in upstream reservoirs. During the summer and early fall, inflow to San Francisco Bay may be greater than historical levels due to deliveries of municipal and agricultural water supplies. Overall, present day water management practices in the Central Valley reduce natural flow variability by creating more uniform flows year-round that diminish natural channel forming, riparian vegetation growth, and food web functions.

As native fishes became depleted in the late 19th century, non-native species were brought in to the Bay and delta, including American shad, striped bass, common carp, and white catfish. As their populations boomed, those of native fishes declined further. Introduction of non-native species accelerated in the 20th century through deliberate introductions of fish and unintended introductions of competitive invertebrates through ballast water of ships. Establishment of non-native species was probably facilitated by altered hydrologic regimes and reduction in habitats for native species. The introduction and spread of non-native species in San Francisco Bay has affected native species, including listed avian species, by competing with them for food and habitat, and preying on native species.

Preliminary, significant steps towards the largest ecological restoration project yet undertaken in the United States have occurred during the past five years and continue to proceed in California's Central Valley. The CALFED Program and the CVPIA's AFRP, in coordination with other Central Valley and Bay Area efforts, have implemented numerous habitat restoration actions. A few of these restoration projects, primarily land acquisition and wetland restoration, include actions within the San Francisco Bay area. Restoration of wetland areas typically involves flooding lands previously used for agriculture, thereby creating additional wetland areas that could serve as rearing habitat for fish prey species of the least tern, and could, if associated with other restoration actions, possibly provide nesting habitat for the least tern and other bird species.

Effects of the Action

California least terns nest in the San Francisco Bay area at Alameda Air Force Base but not at the three proposed management sites (Brooks Island, Hayward Regional Shoreline, and Don Edwards NWR), although recent habitat improvement measures on an island in one of the ponds aims to induce nesting. They are known to roost in the fall at Hayward Regional Shorelines. They winter on the Pacific coast of Central America. Their staple diet is small fish, usually less than 5 cm. Many species of fish are consumed but only those with pre-opercular spines, fin spines, or body depths that exceed the birds' gape are apparently suitable (Atwood and Kelly 1984). Caspian terns are significantly larger than California least terns and take larger prey.

Caspian terns prey upon fish greater than 5 cm in length (Cuthbert and Wires 1999). A study on Caspian terns in the Columbia River showed that Snake River sockeye salmon were more vulnerable to Caspian tern predation than mid and upper-Columbia River sockeye salmon because of size differences (Snake River fish averaged 14.2 cm in length and mid and upper-Columbia River fish averaged 10.3 cm) (Collis *et al.* 2001). Finally, predation of least terns is not anticipated to increase due to the increase in Caspian tern presence since least terns are not roosting in the area when Caspian terns are present.

Conclusion

The proposed action may affect, but will not likely adversely affect California least tern. We make this determination for the following reasons: (1) Construction, maintenance, and monitoring activities, including vehicle and helicopter travel will not occur at the site during fall roosting season for the California least tern; (2) because increased predation of California least terns is not likely to occur due to the increase in Caspian tern presence since least terns are not roosting in the area when Caspian terns are present; (3) California least terns do not nest at any of the three proposed management sites and future nesting bitds are not likely to be in competition for nest sites; and (4) prey use of California least terns likely does not overlap with Caspian terns.

Western Snowy Plover

Status of the Species

The snowy plover was federally listed as threatened in 1993 (58 FR 12874) (Service 1993b). A detailed account of the taxonomy, ecology, and biology of the harvest mouse is presented in the *Western Snowy Plover (Charadrius alexandrinus nivosus) Pacific Coast Population Draft Recovery Plan* (Service 2001a) and the references cited therein.

Charadrinus alexandrinus nivosus is a small shorebird distinguished from other plovers (family Charadriidae) by its small size, pale brown upper parts, dark patches on either side of the upper breast, and dark gray to blackish legs. The species was first described in 1758 by Linnaeus (American Ornithologists' Union, 1957).

The Pacific coast population of the snowy plover is defined as those individuals that nest beside or near tidal waters, and includes all nesting colonies on the mainland coast, peninsulas, offshore islands, adjacent bays and estuaries from southern Washington to southern Baja California, Mexico. Habitats used by nesting and non-nesting birds include sandy coastal beaches, salt pans, coastal dredged spoils sites, dry salt ponds, salt pond levees and gravel bars of the Eel River, Humboldt County, California.

Historic records suggest that nesting Western Snowy Plovers were once more widely distributed in coastal California. In coastal California, Snowy Plovers bred at 53 locations before 1970 (Page and Stenzel 1981). Since then, no evidence of breeding birds has been found at 33 of these 53 sites, which represents a 62 percent decline (Page and Stenzel 1981). The greatest losses were in southern California, within the central portion of the Snowy Plover's coastal breeding range. In 1990 only 6 nesting colonies remained, representing a 79 percent decline in active breeding sites.

A total of 20 plover breeding areas currently occur in coastal California (Page *et al.* 1991). Eight areas support 78 percent of the California coastal breeding population: San Francisco Bay, Monterey Bay, Morro Bay, the Callendar-Mussel Rock Dunes area, the Point Sal to Point Conception area, the Oxnard lowland, Santa Rosa Island, and San Nicolas Island (Page *et al.* 1991).

In the habitats remaining for the snowy Plover, human activity continues to be a key factor adversely affecting snowy Plover coastal breeding sites and breeding populations in California. Projects or management activities in plover nesting areas that cause, induce or increase human-associated disturbance during the snowy plover's breeding season (March 1-September 14) adversely impact snowy plovers. These activities may reduce the functional suitability of nesting, foraging and roosting areas. Activities that may adversely affect nesting and/or wintering plovers include beach nourishment (sand deposition, spreading of sand with machinery); beach cleaning (removal of wrack–surfcast kelp and driftwood); construction of breakwaters and jetties (interruption of sand deposition); dune stabilization/restoration using native and nonnative vegetation or fencing (decreased beach width, increased beach slope, reduction in blowouts and other preferred nesting habitat); beach leveling (increased tidal reach, removal of sparse vegetation used by chicks for shelter, destruction of wrackline feeding habitat); and off-road vehicles driven in nesting areas or at night.

<u>Environmental Baseline</u>

Salt ponds of San Francisco Bay and San Diego Bay, which are filled and drained as part of the salt production process, provide breeding and wintering habitat for snowy plovers. Dry salt ponds and unvegetated salt pond levees are used as snowy plover nesting habitat. Ponds with shallow water provide important foraging habitat for plovers. Nesting plovers can be attracted to an area when ponds are drained during the breeding season, but flooding can then destroy the nests when the ponds are refilled. Also human disturbance resulting from maintenance activities associated with the operation of commercial salt ponds (*i.e.* levee reconstruction and maintenance of facilities) can result in the loss of snowy plovers and alteration or disturbance of their habitat. Feeney and Maffei (1991) observed a sizable population of snowy plovers at the Baumberg and Oliver salt ponds during the breeding and nonbreeding seasons, suggesting that these ponds are important to snowy plovers throughout the year. They suspected that these ponds are used by snowy plovers as both a pre-breeding and post-breeding staging area, based on the relatively high numbers of snowy plovers in mid-February and in late August/September, respectively. The conversion of salt ponds, which provide valuable breeding and wintering habitat for snowy plovers, into tidal marshes would result in a loss of suitable nesting and wintering habitat for snowy plovers.

Snowy plovers are occasional nesters at the Hayward Regional Shoreline on marginal substrate around salt ponds, but are relatively abundant nesters at Don Edwards NWR. Surveys conducted at the refuge (Service 2001a) observed 19 active nests and predicted at least 10 more (for a total of 29 nests) were present on the refuge based on additional brood sightings. Of the 19 observed, 74 percent were successful nest attempts. In addition, surveys of adjacent lands resulted in observations of 31 (CDFG Reserve), and 28 (Cargill) nest sites on properties to the north of the refuge. Predation for the survey sites combined was found to be extremely low, with fewer than 9 percent of the nests predated overall.

Effects of the Action

Hayward Regional Shoreline

Western snowy plovers are known to nest south of the proposed project around adjacent salt ponds. Caspian tern relocation could result in an increase in predators in the general vicinity, and additional predator control may be required if plovers are found nesting closer to the tern colony, but since terns are the attractive nuisance, predators are more likely to successfully target terns than the relatively cryptic and solitary plovers. Caspian terns, themselves, are strictly piscivorous and would not be a threat to plover chicks or compete with plovers for food resources.

Construction activities are not expected to affect nesting plovers as no suitable habitat occurs in the immediate area of the proposed habitat enhancement site. Ingress and egress routes, and a no fly zone will be established in coordination with the Sacramento Fish and Wildlife Office, as needed, to ensure that plovers found nesting in the area are not disturbed by helicopter activities.

Habitat improvement work will require vegetation removal and be conducted in areas currently unsuitable for western snowy plover nesting, so direct impacts from construction activities are unlikely. No net decrease in potential western snowy plover nesting habitat will result from implementation of the project. Creation of bare sand habitat for Caspian terns, however, could provide nesting habitat for western snowy plovers after construction is completed. The earliest plover nesting occur on the California coast during the first week of March in some years and by the third week of March in most years (Page and Persons 1995). Earliest nesting of Caspian terns in the San Francisco Bay area occurs later than earliest nesting of western snowy plovers (Service 2003). However, if western snowy plovers begin nesting in maintained habitat areas in future years, they could be displaced by arriving Caspian terns. Future maintenance activities will be timed in coordination with Service personnel to avoid disturbance to nesting plovers.

Don Edwards NWR

Western snowy plovers are common nesters in the area. With 78 active nests monitored by the Service in 2001, and, based on other observations, as many as 100 nest attempts made in 2001, there appears to be substantial suitable habitat for plovers in the area. Caspian tern relocation could result in an increase in predators in the general vicinity of the tern colony, and additional predator control may be required if plovers are found nesting close to the tern colony, but since terns are the attractive nuisance, predators are more likely to successfully target large groups of terns than the relatively cryptic and solitary plovers. Caspian terns, themselves, would not be a threat to plover chicks or adults, and are strictly piscivorous, so would not compete with snowy plovers for food resources.

Construction activities associated with creating habitat for Caspian terns may affect nesting plovers as potentially occupied habitat occurs in the immediate area of the proposed habitat enhancement site. The N1 levee and N9 shoreline contained active nesting areas in 2001 and 2003 (SFBBO 2004). The Caspian tern nesting site will be located to avoid areas historically used by plovers. Additionally, surveys will be conducted prior to initiation of construction related activities, and if plovers are present, timing of construction will be determined in coordination with Service personnel to ensure no nesting plovers are disturbed. Ingress and egress routes, and a no fly zone and/or timing restrictions will be established in coordination with the Sacramento Fish and Wildlife Office, as needed, to ensure that plovers nesting in the general area are not disturbed by vehicle or helicopter activities.

Habitat improvement work will require vegetation removal and substrate enhancement. No net decrease in potential western snowy plover nesting habitat will result from implementation of the project. Creation of bare sand habitat for Caspian terns, however, could provide additional suitable nesting habitat for western snowy plovers after construction is completed. The plover's earliest nesting activities on the California coast occur during the first week of March in some years and by the third week of March in most years (Page and Persons 1995). Earliest nesting of Caspian terns in the San Francisco Bay area occurs later than earliest nesting of western snowy plovers (Service 2003). However, if western snowy plovers begin nesting in maintained habitat areas in future years, they could be displaced by arriving Caspian terns, or by maintenance activities that may occur prior to the arrival of the terns. Maintenance activities will be timed in coordination with Service personnel to avoid disturbance to nesting plovers. Disturbance to nesting plovers by arriving terns would occur early in the nesting season, so would not likely result in abandonment of eggs, and most displaced plovers are expected to immediately make new nest attempts within suitable habitat available at or near the refuge. Therefore, only the occasional displacement event is likely to result in a significant impairment of essential breeding patterns for a pair of plovers.

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. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act. We are unaware of any non-Federal actions affecting bull trout that are reasonably certain to occur in the action area considered by this opinion.

Conclusion

The proposed action may affect, but will not likely adversely affect the western snowy plover. We make this determination for the following reasons: (1) construction, maintenance, and monitoring activities, including vehicle and helicopter travel will not occur at the site during the nesting season for the plover; (2) predator control efforts will avoid increased predation pressure on the plover that may result from the proposed action; (3) potentially displaced plovers would most likely occur early in the season and thus, would be able to find alternative nesting sites nearby; and (4) direct conflicts between Caspian terns and western snowy plovers have not been documented in the Bay Area.

Delta Smelt

Status of the Species

Delta smelt was federally listed as a threatened species on March 5, 1993, (58 FR 12854) (Service 1993a). Critical habitat for delta smelt was designated on December 19, 1994 (59 FR 65256) (Service Operations Manager 110 1994b). The Sacramento-San Joaquin Delta Native Fishes Recovery Plan was completed in 1996 (Service 1996). The Five Year Status Review for the delta smelt was completed on March 3 1,2004 (Service 2004).

Delta smelt are slender-bodied fish that typically reach 60-70 mm standard length (measured from tip of the snout to origin of the caudal fin), although a few may reach 120 mm standard length.

Delta smelt are endemic to the upper Sacramento-San Joaquin estuary. They occur in the Delta primarily below Isleton on the Sacramento River, below Mossdale on the San Joaquin River, and in Suisun Bay. They move into freshwater when spawning (ranging from January to July) and can occur in: (1) the Sacramento River as high as Sacramento, (2) the Mokelumne River system, (3) the Cache Slough region, (4) the Delta, and, (5) Montezuma Slough, (6) Suisun Bay, (7) Suisun Marsh, (8) Carquinez Strait, (9) Napa River, and (10) San Pablo Bay. It is not known if delta smelt in San Pablo Bay are a permanent population or if they are washed into the Bay during high outflow periods. In any month, two or more life stages (adult, larvae, and juveniles) of delta smelt have the potential to be present in Suisun Bay (Department of Water Resources (DWR) and the Bureau of Reclamation (Reclamation) 1994; Molye 1976; and Wang 1991). Delta smelt are also captured seasonally in Suisun Marsh.

Delta smelt are euryhaline (a species that tolerates a wide range of salinities) fish that generally occur in water with less than 10-12 parts per thousand (ppt) salinity. In wet years, when abundance levels are high, their distribution is normally very broad. In Suisun Marsh, delta smelt larvae occur in both large sloughs and small dead end sloughs. Delta smelt of all sizes are found in the main channels of the Delta and Suisun Marsh and the open waters of Suisun Bay where the waters are well oxygenated and temperatures relatively cool. When not spawning, they tend to be concentrated near the zone where incoming salt water and out flowing freshwater mix (mixing zone). At all life stages delta smelt are found in greatest abundance in the top 2 m of the water column and usually not in close association with the shoreline. Delta smelt inhabit open, surface waters of the Delta and Suisun Bay, where they presumably school.

In most years, spawning occurs in shallow water habitats in the Delta. Shortly before spawning, adult smelt migrate upstream from the brackish-water habitat associated with the mixing zone to disperse widely into river channels and tidally-influenced backwater sloughs (Radtke 1966; Moyle 1976, 2002; Wang 1991). Some spawning probably occurs in shallow water habitats in Suisun Bay and Suisun Marsh during wetter years (Sweetnam 1999 and Wang 1991). Spawning has also been recorded in Montezuma Slough near Suisun Bay (Wang 1986) and also may occur in Suisun Slough in Suisun Marsh (P. Moyle, UCD, unpublished data). The spawning season varies from year to year, and may occur from late winter (December) to early summer (July). A recent study of delta smelt eggs and larvae (Wang and Brown 1993 as cited in Water Resources and Reclamation 1994) confirmed that spawning may occur from February through June, with a peak in April and May. Newly hatched delta smelt have a large oil globule that makes them semi-buoyant, allowing them to maintain themselves just off the bottom (R. Mager, UCD, unpublished data), where they feed on rotifers (microscopic crustaceans used by fish for food) and other microscopic prey. Once the swimbladder (a gas-filled organ that allows fish to maintain neutral buoyancy) develops, larvae become more buoyant and rise up higher into the water column. At this stage, 16-18 mm total length, most are presumably washed downstream until they reach the mixing zone or the area immediately upstream of it. Growth is rapid and juvenile fish are 40-50 mm long by early August (Erkkila *et al.* 1950; Ganssle 1966; Radtke 1966). Delta smelt reach 55-70 mm standard length in 7-9 months (Moyle 1976, 2002).

The abrupt change from a single-age, adult cohort during spawning in spring to a population dominated by juveniles in summer suggests strongly that most adults die after they spawn (Radtke 1966 and Moyle 1976,2002). However, in El Nino years when temperatures rise above normal before all adults have spawned, some fraction of the unspawned population may also hold over as two year-old fish and spawn in the subsequent year. These two-year-old adults may enhance reproductive success in years following El Nino events. In a near-annual fish like delta smelt, a strong relationship would be expected between number of spawners present in one year and number of recruits to the population the following year. Instead, the stock-recruit relationship for delta smelt is weak, accounting for about a quarter of the variability in recruitment (Sweetnam and Stevens 1993). This relationship does indicate, however, that factors affecting numbers of spawning adults (e.g., entrainment, toxics, and predation) can have an effect on delta smelt numbers the following year.

Delta smelt feed primarily on (1) planktonic copepods (small crustaceans used by fish for food), (2) cladocerans (small crustaceans used by fish for food), (3) amphipods (small crustaceans used by fish for food) and, to a lesser extent, (4) on insect larvae. Larger fish may also feed on the shrimp.

The amount and extent of suitable habitat for the delta smelt has declined dramatically. The advent in 1853 of hydraulic mining in the Sacramento and San Joaquin rivers led to an increase in siltation and the alteration of the circulation patterns of the Estuary (Nichols *et al.* 1986, Monroe and Kelly 1992). The reclamation of Merritt Island for agricultural purposes, in the same year, marked the beginning of the present-day cumulative loss of 94% of the Estuary's tidal marshes (Nichols *et al.* 1986, Monroe and Kelly 1992). The extensive levee system in the Delta has led to a loss of seasonally flooded habitat and significantly changed the hydrology of the Delta ecosystem, restricting the ability of suitable habitat substrates to re-vegetate. Delta smelt were once one of the most common pelagic (living in open water away from the bottom) fish in the upper Sacramento-San Joaquin estuary, as indicated by its abundance in DFG trawl catches (Erkkila *et al.* 1950; Radtke 1966; Stevens and Miller 1983). Delta smelt abundance from year to year has fluctuated greatly in the past, but between 1982 and 1992 their population was consistently low. During the period 1982-1992, most of the population was confined to the Sacramento River channel between Collinsville and Rio Vista (Sweetnam, DFG unpublished data). The actual size of the delta smelt population is not known. However, the pelagic life style of delta smelt, short life span, spawning habits, and relatively low fecundity indicate that a fairly substantial population probably is necessary to keep the species from becoming extinct.

Recreation in the Delta has resulted in the presence and propagation of predatory non-native fish such as striped bass (*Morone saxatilis*). Additionally, recreational boat traffic has led to a loss of habitat from the building of docks and an increase in the rate of erosion resulting from boat wakes. In addition to the loss of habitat, erosion reduces the water quality and retards the production of phytoplankton in the Delta. In addition to the degradation and loss of estuarine habitat, delta smelt have been increasingly subject to entrainment, upstream or reverse flows of waters in the Delta and San Joaquin River, and constriction of low salinity habitat to deep-water river channels of the interior Delta. These adverse conditions are primarily a result of the steadily increasing proportion of river flow being diverted from the Delta by the Projects, and occasional droughts (Monroe and Kelly 1992). Reduced water quality from agricultural runoff, effluent discharge and boat effluent has the potential to harm the pelagic larvae and reduce the availability of the planktonic food source. The introduction of the Asian clam (*Potamocorbula amurensis*), a highly efficient filter feeder, presently reduces the concentration of phytoplankton in this area.

In summary, the threats of the destruction, modification, or curtailment of its habitat or range resulting from extreme outflow conditions, the operations of the State and Federal water projects, and other water diversions as described in the original listing remain. The only new information concerning the delta smelt's population size and extinction probability indicates that the population is at risk of falling below an effective population size and therefore in danger of becoming extinct. The increases in water demands are likely to result in less suitable rearing conditions for delta smelt, increased vulnerability to entrainment, and less water available for maintaining the preferred salinity. The importance of exposure to toxic chemicals on the population of delta smelt is highly uncertain. In addition, many potential threats have not been sufficiently studied to determine their effects, such as predation, disease, competition, and hybridization.

Effects of the Action

The only exposure to effects on delta smelt that may result from the proposed action appears to be from predation by Caspian terns. The foraging range of the Caspian tern colony at Brooks Island and Hayward Regional Shoreline could include Suisun Bay, the southern end of the range of delta smelt. Delta smelt is a near-annual species that occurs in various life stages throughout the year. Most adults are thought to die after spawning in their first year. Although there may be a slight temporal overlap between foraging terns and the presence of adult size smelt in Suisun Bay, the rare foraging tern (given the distance and geography between the colony and Suisun Bay) would most likely find successful foraging opportunities in the form of abundant populations of other schooling fish species. Furthermore, delta smelt of the adult size class that could be captured by Caspian terns (e.g., 5.5 - 9 mm) are not expected to be present until after August when the terns are expected to begin leaving the area. Therefore, it is extremely unlikely that predation by Caspian terns on delta smelt would occur.

Conclusion

The proposed action may affect, but is not likely to adversely affect delta smelt. We make this determination for the following reasons: (1) Construction, maintenance, and monitoring activities will not affect delta smelt; (2) because of the small size of the delta smelt and temporal separation between foraging terns and adult smelt, and (3) because upon the rare event that Caspian terns from this colony will make their way into Suisun Bay, Caspian terns are not likely to capture delta smelt.

Salt Marsh Harvest Mouse

Status of the Species and Environmental Baseline

The salt marsh harvest mouse was federally listed as endangered in 1970 (35 FR 16047). Critical habitat has not been proposed or designated. A detailed account of the taxonomy, ecology, and biology of the salt marsh harvest mouse is presented in the Recovery Plan (Service 1984) and the references cited therein. The salt marsh harvest mouse is a fully protected species under California law (See California Fish and Game Code Section 4700).

The salt marsh harvest mouse is a rodent endemic to the salt and brackish marshes of the San Francisco Estuary and adjacent tidally influenced areas. The salt marsh harvest mouse closely resembles the western harvest mouse (*R. megalotis*). The salt marsh harvest mouse typically weighs about 10 grams, has a head and body length ranging from 69-74 mm, a tail length ranging from 65-82 mm, and a hind foot length of 17-18 mm (Fisler 1965). As stated in the recovery plan, the salt marsh harvest mouse, when compared to the western harvest mouse, have darker ears, belly and back, and a slightly thicker, less pointed and unicolored tail. The salt marsh harvest mouse is further distinguished taxonomically into the northern and southern subspecies, *R. raviventris halicoetes* and *R. raviventris raviventris*, respectively. Of the two subspecies, *R. r. halicoetes* more closely resembles *R. megalotis*, and can be difficult to differentiate in the field; body color and color of ventral hairs as well as the thickness and shape of the tail have been used to distinguish the two.

The salt marsh harvest mouse has evolved to a life in tidal marshes. Specifically, they have evolved to depend mainly on dense pickle weed (*Salicornia virginica*) as their primary cover and food source. However, salt marsh harvest mice may utilize a broader source of food and cover which includes salt grass (*Distichlis spicata*) and other vegetation typically found in the salt and brackish marshes of this region. In natural systems, salt marsh harvest mice can be found in the middle tidal marsh and upland transition zones. Upland refugia is an essential habitat component during high tide events. Salt marsh harvest mice are highly dependent on cover, and open areas as small as 10 meters wide may act as barriers to movement (Shellhammer 1978, as cited in Service 1984). The salt marsh harvest mouse does not burrow. It has been noted that the northern subspecies may build nests of loose grasses.

As described by Fisler (1965), male salt marsh harvest mice are reproductively active from April through September, but may appear active throughout the year. Females are reproductively active from March to November, and have a mean litter size of approximately four offspring.

The historic range of the species included tidal marshes within the San Francisco and San Pablo bays, east to the Collinsville-Antioch areas. Agriculture and urbanization has claimed much of the former historic tidal marshes, resulting in a 79 percent reduction in the amount of tidal marshes in these areas (Goals Project 1999). At present, the distribution of the northern subspecies occurs along Suisun and San Pablo Bays north of Point Pinole in Contra Costa County and Point Pedro in Marin County. The southern subspecies is found in marshes in Corte Madera, Richmond, and South San Francisco Bay mostly south of the San Mateo Bridge (Highway 92).

Effects of the Action

An occupied salt marsh harvest mouse preserve occurs a few hundred yards to the southeast of the proposed project site at Hayward Regional Shoreline. Other occupied habitat may occur in the area as well since appropriate habitat does exist at Don Edwards NWR. Potential effects to the salt marsh harvest mouse due to construction, maintenance, and monitoring could include roadway interactions and helicopter noise. Ingress and egress routes, and a no fly zone and/or timing restrictions will be established in coordination with the Sacramento Fish and Wildlife Office, as needed, to ensure that salt marsh harvest mouse at Hayward Regional Shoreline and Don Edwards NWR are not disturbed by vehicle or helicopter activities to such an extent that could cause adverse impacts to individual mice.

Establishment of new Caspian tern colonies could result in an increase in predators in the general vicinity of the tern colony. However, since terns are an attractive nuisance, predators are more likely to successfully target large groups of terns than the relatively cryptic and solitary mice. Caspian terns, themselves, would not be a threat to salt marsh harvest mouse, and are strictly piscivorous, so will not compete with them for food resources.

Conclusion

The proposed action may affect, but will not likely adversely affect salt marsh harvest mouse. We make this determination on the following reasons: (1) Construction, maintenance, and monitoring activities, in particular vehicle and helicopter travel will not occur where it would be a threat to salt marsh harvest mouse; (2) because existing predator control efforts will avoid increased predation pressure on salt marsh harvest mouse that may result from the proposed action, and (3) because Caspian terns are not a direct predation threat to salt marsh harvest mouse.

REFERENCES

Ainley, D.G., S.G. Allen, and L.B. Spear. 1995. Offshore occurrence patterns of marbled murrelets in central California. *In*: C.J. Ralph, G.L. Hunt, M.G. Raphael, and J. F. Piatt (Tech. eds.), Ecology and Conservation of the Marbled Murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, California: Pacific Southwest Experiment Station, Forest Service, U.S. Dept. of Agriculture; 420 pp.

Albertson, J.D. 1995. Ecology of the California clapper rail in south San Francisco Bay. A thesis submitted to San Francisco State University. 147 pp.

Albertson, J.D. and J.G. Evens. 2000. California clapper rail. In: Baylands ecosystem species and community profiles: life histories and environmental requirements of key plants, fish and wildlife. Prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. P.R. Olofson, editor. San Francisco Bay Regional Water Quality Control Board, Oakland, California. Pages 332-341.

Allen, C., A. Atkins, and M. Stern. 1994. Status and Recolonization of the Warner sucker (*Catostomus warnerensis*) and other fishes in the Warner Lakes in Southeast Oregon, 1994. Unpublished report to Bureau of Land Management and Oregon Department of Fish and Wildlife. 22 pp.

Allen, C., K. Hartzell, M. Stern and A. Munhall. 1995a. Status of the Warner sucker (*Catostomus warnerensis*) and other fishes in the Warner Basin in Southeast Oregon, 1995. Unpublished report to the Fish and Wildlife Service. 35 pp.

Allen, C., M. Stern, and A. Munhall. 1995b. Monitoring of the Warner sucker and other fishes in the Warner Basin in 1994 and 1995. Unpublished report to the Fish and Wildlife Service. 4 pp.

Allen, C., K. Hartzell, and M. Stern. 1996. Warner sucker progress report - 1996 findings. Unpublished report to Bureau of Land Management. 55 pp.

Alverson, E.R. 1990. A survey for Native Grassland Remnants in the Mid-Willamette Valley, Oregon. Unpub. Report, 33pp.

American Ornithologists' Union (AOU). 1957. The A.O.U. checklist of North American birds. Fifth edition. 168 pp.

Anderson, D.W. 1983. Seabirds: distribution and assemblages in the Gulf of California. Chap. 9, in: T. Case and M. L. Cody, eds., Island biogeography in the Sea of Cortez. University of California Press, Berkeley, California.

Anderson, D.W., and I.T. Anderson. 1976. Distribution and status of brown pelicans in the California current. American Birds 30(1):3-12.

Anderson, D.W., and F. Gress. 1984. Status of a northern population of California brown pelicans. The Condor 85(1):79-88.

Anderson, D.W., F. Gress, K.F. Mais, and P.R. Kelly. 1980. Brown pelicans as anchovy stock indicators and their relationships to commercial fishing. Calif. Coop. Oceanic Fish. Invest. Rep. 21:54-61.

Anderson, S. 2003. Foraging Ecology, Colony Attendance, and Chick Provisioning of Caspian Terns (*Sterna caspia*) in the Columbia River Estuary. Masters thesis, Oregon State University, Corvallis, OR, 101 pages.

Anthony, R.G., R.L. Knight, G.T. Allen, B.R. McClelland, and J.I. Hodges. 1982. Habitat use by nesting and roosting bald eagles in the Pacific Northwest. Trans. N. Am. Wildl. Nat. Res. Conf. 47:332-342.

Atwood, J.L. and P.R. Kelly. 1984. Fish dropped on breeding colonies as indicators of least tern food habits. Wilson Bulletin. 96:34-47.

Avocet Research Associates. 2003. Regional significance of the California clapper rail population at Bahia Lagoon and evaluation of the Bahia Homeowner Association's biological assessment and proposal for dredging and mitigation. Appendix B in *Environmentally Superior Redesign, Bahia Homeowners Boat Lock Project, Novato, California, 22 July 2003, Project No. 1076.* 13 pp.

Bargmann, Greg. 1998. Forage fish management plan: A plan for managing the forage fish resources and fisheries of Washington. Washington Department of Fish and Wildlife. Olympia, Washington. 66 pg.

Baxter, C.V., and F.R. Hauer. 2000. Geomorphology, hyporheic exchange, and the selection of spawning habitat by bull trout (*Salvelinus confluentus*). Can. J. Aquat. Sci. 57: 1470-1481.

Baxter, J.S., and J.D. McPhail. 1997. Diel microhabitat preferences of juvenile bull trout in an artificial stream channel. North American Journal of Fisheries Management 17:975-980.

Beamer, E. and R. Henderson. 2004. *Incomplete draft.* Skagit Delta and Bay habitat use data report. Appendix E: Bull Trout in the nearshore. Prepared for U.S. Army Corps of Engineers, Seattle District, Washington.

Beissinger, S.R. 1995. Population trends of the marbled murrelet projected from demographic analyses. *In*: C.J. Ralph, G.L. Hunt, M.G. Raphael, and J. F. Piatt (Tech. eds.), Ecology and Conservation of the Marbled Murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, California: Pacific Southwest Experiment Station, Forest Service. 420 pp.

Beissinger, S.R, and M.Z. Peery. 2003. Range-wide analysis of juvenile ratios for marbled murrelet monitoring programs: Implications for Demopgraphic Analyses. A report to the Service, Arcata Field Office. 20 August 2003. 29 pp.

Berg, W. J. 1991. Selected observations and interpretations on the life-history of the Warner sucker (*Catostomus warnerensis*): conservation genetic management. Unpublished report to the Warner Sucker Working Group. 11 pp.

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Dolloff, G.B. Grette, R.A. House, M.L. Murphy, K.V. Koski and J.R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. *In* Streamside management: forestry and fishery interactions. E.O. Salo and T.W. Cundy, Editors., Univ. Washington, Institute of Forest Resources : 143-190.

Boag, T.D. 1987. Food habits of bull char (*Salvelinus confluentus*), and rainbow trout (*Salmo gairdneri*), coexisting in the foothills stream in northern Alberta. Canadian Field-Naturalist 101(1): 56-62.

Boag, P.G. 1992. Environment and experience: settlement culture in the nineteenth-century Oregon. University of California Press. Berkeley, California. 209 pages.

Bobzien, Steve. East Bay Regional Parks. 2004. Personal communications.

Bolsinger, C.L. and K.L. Wadell. 1993. Area of old-growth forest in California, Oregon, and Washington. Resource Bulletin PNW-RB-197. Portland, Oregon: Forest Service, Pacific Northwest Research Station. 26 pp.

Bond, C.E. and C.I. Coombs. 1985. Characteristics and life history of *Catostomus warnerensis*. Proc. Desert Fisbes Council. 13:40-53.

Bond, C.E. 1992. Notes on the nomenclature and distribution of the bull trout and the effects of human activity on the species. Pages 1-4 *In*: Howell, P.J. and D.V. Buchanan, eds. Proceedings of the Gearhart Mountain bull trout workshop. Oregon Chapter of the American Fisheries Society, Corvallis, OR.

Bonneau, J. L. and D. L. Scarnecchia 1996. Distribution of juvenile bull trout in a thermal gradient of a plunge pool in Granite Creek, Idaho. Transactions of the American Fisheries Society 125(4): 628-630.

Booth, D.E. 1991. Estimating prelogging old-growth in the Pacific Northwest. Journal of Forestry (October):25-29.

Bosse, S.E., D. Hartzel, C. Allen, M. Stern, and A. Munhall. 1997. Warner Sucker Progress Report-1997 Findings. Unpublished Report to Bureau of Land Management. 78pp.

Brenkman, S., Olympic National Park. 2003a. Additions to recovery plan 13 January 2003.

Brenkman, S., Olympic National Park. 2003b. Protocol development to conduct a population estimate of federally threatened bull trout in the North Fork Skokomish River, Olympic National Park. Submitted to Service, Lacey, Washington, July 2003. Port Angeles, Washington.

Brown, L.G. 1992. On the zoogeography and life history of Washington native char Dolly Varden (*Salvelinus malma*) and bull trout (*Salvelinus confluentus*). Washington Department of Wildlife, Fisheries Management Division Report. Olympia, Washington.

Buchanan, D. M. and S. V. Gregory. 1997. Development of water temperature standards to protect and restore habitat for bull trout and other cold water species in Oregon. Pages 1-8 *in*: Mackay, W.C., M.K. Brewin and M. Monita, editors. Friends of the Bull Trout Conference Proceedings. Bull Trout Task Force (Alberta), c/o Trout Unlimited Calgary, Alberta, Canada.

Burger, A.E. 1995. Marine distribution, abundance and habitat of marbled murrelets in British Columbia. *In*: C.J. Ralph, G.L. Hunt, M.G. Raphael, and J. F. Piatt (Tech. eds.), Ecology and Conservation of the Marbled Murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, California: Pacific Southwest Experiment Station, Forest Service, U.S. Dept. of Agriculture; 420 pp.

Burger, A. E. 2002. Conservation assessment of marbled murrelets in British Columbia, a review of biology, populations, habitat associations and conservation. Pacific and Yukon Region, Canadian Wildlife Service: 168 pp.

Burkett, E. E. 1995. Marbled Murrelet Food Habits and Prey Ecology. In Ralph, C. John; Hunt, George L., Jr.; Raphael, Martin G.; Piatt, John F., Technical Editors. 1995. Ecology and conservation of the Marbled Murrelet. Gen. Tech. Rep. PSW-GTR-152. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture; 420 pp.

California Department of Fish and Game. Internet Site (a) (http://www.delta.dfg.ca.gov/gallery/dsmelt.asp). Central Valley Bay-Delta Branch: Delta Smelt.

California Department of Fish and Game. Internet Site (b) (http://www.elkhornslough.org/sunflower/CDFGreport.pdf). Santa Cruz Tarplant.

Cam, E., L. Lougheed, R. Bradley, and F. Cooke. 2003. Demographic assessment of a marbled murrelet population from capture-mark-recapture and radio telemetry data. Conservation Biology 17(4): 1118-1126.

Campbell-Craven Environmental Consultants. 1994. Draft evaluation of fish passage alternatives for water diversions in Honey Creek, tributary to Hart Lake, Lake County, Oregon. Prepared for Oregon Department of Fish and Wildlife. Portland, Oregon. 16 pp.

Carter, H.R. and R.A. Erickson. 1992. Status and Conservation of the Marbled Murrelet in California, 1892-1987. *In*: H.R. Carter and M.L. Morrison (editors), Status and Conservation of the Marbled Murrelet in North America. Proceedings of the Western Foundation for Vertebrate Zoology 5.

Carter, H.R. and S.G. Sealy. 1990. Daily Foraging Behavior of Marbled Murrelets. Studies in Avian Biology No. 14:93-102, 1990. Condor 88:473-477.

Carter, H.R. and J.L. Stein. 1995. Molts and Plumages in the Annual Cycle of the Marbled Murrelet. *In*: C.J. Ralph, G.L. Hunt, M.G. Raphael, and J.F. Piatt (Tech. eds.), Ecology and Conservation of the Marbled Murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, California: Pacific Southwest Experiment Station, Forest Service, U.S. Dept. of Agriculture; 420 pp.

Carter, H.R., M..L.C. McAllister, and M.E. Isleib. 1995. Mortality of marbled murrelets in gill nets in North America. Pages 271-283 in C.J. Ralph, G.L. Hunt, M. Raphael, and J.F. Piatt (Tech. Eds.). Ecology and conservation of the marbled murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Dept. of Agriculture. 420 pp.

Cavender, T.M. 1978. Taxonomy and distribution of the bull trout, (*Salvelinus confluentus*) (Suckley), from the American Northwest. California Fish and Game 64(3):139-174.

Christy, J.A. and E.R. Alverson. 2002. Vegetation of Willamette Valley, Oregon, in 1850. (In review).

City of Seattle. 2001. Final Cedar River watershed habitat conservation plan for the issuance of a permit to allow incidental take of threatened and endangered species. City of Seattle. Seattle, Washington. April 2000.

Clancy, C.G. 1993. Statewide Fisheries Investigations, Bitterroot Forest Inventory. Helena, MT: Montana Department of Fish, Wildlife, and Parks, Fisheries Division. [not paged]. Job Completion Report. Project F-46-R-4. (As referenced in U.S. Department of Interior 1998d).

Clark, D.L., K.K. Finley and C.A. Ingersoll. 1993. Status report for *Erigeron decumbens var. decumbens*. Report prepared for the Conservation Biology Program, Oregon Department of Agriculture, 635 Capitol Street NE, Salem, Oregon.

Collins, J.N., J.G. Evens, and B. Grewell. 1994. A synoptic survey of the distribution and abundance of the California clapper rail *Rallus longirostris obsoletus* in the northern reaches of the San Francisco Estuary during the 1992 and 1993 breeding seasons. Final report to California Department of Fish and Game. 22 pp. plus appendix.

Collis, K., S. Adamany, D.D. Roby, D.P. Craig, and D.E. Lyons. 1998. Avian Predation on Juvenile Salmonids in the Lower Columbia River. 1998 Annual Report.

Collis, K., D. D. Roby, D. P. Craig, B. A. Ryan, and R. D. Ledgerwood. 2001. Colonial Waterbird Predation on Juvenile Salmonids Tagged with Passive Integrated Transponders in the Columbia River Estuary: Vulnerability of Different Salmonid Species, Stocks, and Rearing Types. Transactions of the American Fisheries Society. 130:385-396.

Collis, K. D., D. Roby, D. E. Lyons, R. M. Suryan, M. Antolos, S. K. Anderson, A. M. Myers and M. Hawbecker. 2002a. Caspian tern research on the Lower Columbia River, Final 2001 Summary. Prepared for Bonneville Power Administration and the Interagency Caspian Tern Working Group, 35 pages.

Collis, K. D., D. Roby, D. E. Lyons, M. Antolos, S. K. Anderson, A. M. Myers, and M. Hawbecker. 2002b. Caspian tern research on the Lower Columbia River, Final 2000 Season Summary. Prepared for Bonneville Power Administration and the Interagency Caspian Tern Working Group, 26 pages.

Collis, K. , D. D. Roby, D. E. Lyons, R. M. Suryan, M. Antolos, S. K. Anderson, A. M. Myers and M. Hawbecker. 2003a. Caspian tern research on the Lower Columbia River, Final 2002 Season Summary. Prepared for Bonneville Power Administration and the Interagency Caspian Tern Working Group, 16 pages.

Collis, K. ,D. D. Roby, C. Couch, G. Dorsey, K. Fischer, D. E. Lyons, A. M. Myers, S. K. Nelson, R. M. Suryan, A. Evans, and M. Hawbecker. 2003b. Caspian tern research on the Lower Columbia River, Draft 2003 Season Summary. Prepared for Bonneville Power Administration and the Interagency Caspian Tern Working Group, 38 pages.

Connor, E., D. Reiser, K. Binkley, D. Paige, and K. Lynch. 1997. Abundance and distribution of an unexploited bull trout population in the Cedar River Watershed, Washington. Pages 403-411 *in* Mackay, W.C., M.K. Brewin, and M. Monita, editors. Friends of the bull trout conference proceedings. Bull Trout Task Force (Alberta), c/o Trout Unlimited Canada, Calgary.

Coombs, C. I., C. E. Bond, and S. F. Drohan. 1979. Spawning and early life history of the Warner sucker (*Catostomus warnerensis*). Unpublished report to Fish and Wildlife Service. 52 pp.

Coombs, C.I., and C.E. Bond. 1980. Report of investigations on *Catostomus warnerensis*, fall 1979 and spring 1980. Unpublished report to Fish and Wildlife Service. 32 pp.

Cuthbert, F.J. and L.R. Wires. 1999. Caspian Tern (*Sterna caspia*). *In* The Birds of North American, No. 403 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Davis, B. and R. Miller. 1967. Brain patterns in minnows of the genus *Hybopsis* in relation to feeding habits and habitat. Copeia 1967:1-39.

DeGroot, D.S. 1927. The California clapper rail: its nesting habitats, enemies, and habitat. Condor. 29:259-270.

del Hoyo, J., A. Elliot, and J. Sargatal eds. 1992. Handbook of the Birds of the World. Volume 1. Lynx Edicions. Barcelona, Spain.

Department of Water Resources and U.S. Bureau of Reclamation, Mid-Pacific Region 1993. Effects of the Central Valley Project and State Water Project on delta smelt. Sacramento County, California. 134 pp.

Donald, D.B. and D.J. Alger. 1993. Geographic distribution, species displacement, and niche overlap for lake trout and bull trout in mountain lakes. Canadian Journal of Zoology 71:238-247.

Dunham, J.B. and G.L. Chandler. 2001. Models to predict suitable habitat for juvenile bull trout in Washington State. Final report to Service, Lacey, WA.

Dunham, J., B. Rieman, and G. Chandler. 2001. Development of field-based models of suitable thermal regimes for Interior Colombia Basin salmonids. Interagency agreement #00-IA-11222014-521, Final report to EPA, Seattle WA.

Eastman, Donald C. 1990. Rare and Endangered Plants of Oregon. Wilsonville, OR: Beautiful America Publishing Company, 194 pages.

Erkkila, L.F., J.F. Moffett, O.B. Cope, B.R. Smith, and R.S. Nelson. 1950. Sacramento-San Joaquin Delta fishery resources: effects of Tracy pumping plant and delta cross channel. U.S. Fish and Wildlife Services Special Report. Fisheries 56. 109 pp.

Evens, J. 2000. Sonoma Creek bridge seismic retrofit and barrier placement project: Sonoma and Solano counties, California clapper rail & California black rail: 2000 breeding season surveys. Final report prepared for California Department of Transportation. 7 pp. plus figures.

Evens, J. and G. Page. 1983. The ecology of rail populations at Corte Madera Ecological Preserve with recommendations for management. Report prepared for Marin Audubon Society. 62 pp.

Falk, Donald A. and Kent E. Holsinger. Editors. 1991. Genetics and Conservation of Rare Plants. Oxford University Press. New York. Pp. 283.

Federal Register. 1978 (April 26). 50 CFR Part 17. Endangered and Threatened Wildlife and Plants. (Discussion of Antioch Dunes evening primrose and Contra Costa wallflower among others).

Federal Register. 1994 (December 19). 50 CFR Part 17. Endangered and Threatened Wildlife and Plants: Critical Habitat Determination for the Delta Smelt.

Federal Register. 1997 (November 20). 50 CFR Part 17. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for Two Tidal Marsh Plants—*Cirsium* *hydrophilum* var. *hydrophilum* (Suisun Thistle) and *Cordylanthus mollis* ssp. *mollis* (Soft Bird's-Bead) from the San Francisco Bay Area of California.

Federal Register. 2000 (October 3). 50 CFR Part 17. Endangered and Threatened Wildlife and Plants; Final Determination of Critical Habitat for the Alameda Whipsnake (*Masticophis lateralis euryxanthus*).

Federal Register. 2002a (November 21). 50 CFR Part 17. Endangered and Threatened Wildlife and Plants; Critical Habitat Designation for Four Vernal Pool Crustaceans and Eleven Vernal Pool Plants in California and Southern Oregon.

Federal Register 2002b. (October 16). 50 CFR Part 17. Endangered and Threatened Wildlife and Plants; Final Designation of Critical Habitat for *Holocarpha macradenia* (Santa Cruz Tarplant).

Feeney, L.R. and W.A. Maffei. 1991. Snowy plovers and their habitat at the Baumberg area and Oliver salt ponds, Hayward, California, March 1989 through May 1990. City of Hayward, Hayward, California. 162 pp.

Fisler, G.F. 1965. Adaptations and speciation in harvest mice of the marshes of San Francisco Bay. University of California Publications in Zoology, Volume 77. University of California Press, Berkeley, California 108 pp.

Fitzpatrick, G.S. and C.B. Schultz. 2001. 2000 status of the Fender's blue butterfly (*Icaricia icarioides fenderi*) in Lane County, Oregon: Population estimates and site evaluations. An unpublished report to Oregon Natural Heritage Program and US Fish and Wildlife Service.

Forester, K.S. 1989. Summary of California clapper rail winter populations in the San Francisco Bay NWR, Nov. 1988 to Jan. 1989. Unpubl. Report. San Francisco Bay NWR.

Foerster, K.S., J.E. Takekawa, and J.D. Albertson. 1990. Breeding density, nesting habitat, and predators of the California clapper rail. Unpubl. Rpt. No. SFBNWR-116400-90-1, prep. for San Francisco Bay NWR, Fremont, California. 46 pp.

Foerster, K.S. and J.E. Takekawa. 1991. San Francisco Bay National Wildlife Refuge Predator Management Plan and Final Environmental Assessment. U.S. Fish and Wildlife Service. San Francisco Bay National Wildlife Refuge, California. 54 pp.

Fraley, J.J. and B.B. Shepard. 1989. Life history, ecology and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake and River system, Montana. Northwest Science 63(4): 133-143.

Franklin, J.F. and C.T. Dyrness. 1973. *Natural Vegetation of Oregon and Washington*. USDA Forest Service Pacific Northwest Forest and Range Experiment Station, Portland, OR. 417 pp. USDA Forest Service Technical Report PNW-8.

Frissell, C.A. 1999. An ecosystem approach to habitat conservation for bull trout: groundwater and surface water protection. Open File Report Number 156-99. Flathead lake Biological Station, The University of Montana, Polson, MT.

Fry, M. 1995. Pollution and Fishing Threats to Marbled Murrelets. In: C.J. Ralph, G.L. Hunt, M.G. Raphael, and J.F. Piatt (Tech. eds.), Ecology and Conservation of the Marbled Murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, California: Pacific Southwest Experiment Station, Forest Service, U.S. Dept. of Agriculture; 420 pp.

Ganssle, D. 1966. Fishes and decapods of San Pablo and Suisun bays. Pp.64-94 in D.W. Kelley, ed.: Ecological studies of the Sacramento-San Joaquin Estuary, Part 1. Calif. Dept. Fish and Game, Fish Bulletin No. 133.

Garcia, E.J. 1995. Conservation of the California clapper rail: An analysis of survey methods and habitat use in Marin County, California. Master Thesis, University of California-Davis. 135 pp.

Gill, R., Jr. 1978. Status and distribution of the California clapper rail (*Rallus longirostris obsoletus*). 21 pp. plus appendix.

Gill, R.E., Jr., and L.R. Mewaldt. 1983. Pacific Coast Caspian terns: Dynamics of an expanding population. The Auk 100: 369-381.

Gilpin, M.E. and M.E. Soulé. 1986. Minimum viable populations: Processes of species extinction. In *Conservation Biology: The Science of Scarcity and Diversity*, ed. M.E. Soulé, pp 19-34. Sinauer, Sunderland, Mass.

Gitzendanner M. A. 2000. Genetic diversity in rare and widespread plant species. Ph.D. dissertation, Washington State University.

Goals Project. 1999. Baylands ecosystem habitat goals. A report of habitat recommendations prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. U.S. Environmental Protection Agency, San Francisco, California, and San Francisco Bay Regional Water Quality Control Board, Oakland, California. 209 pp. plus appendices.

Goetz, F. 1989. Biology of the bull trout, *Salvelinus confluentus*, literature review. Willamette National Forest, Eugene, Oregon.

Goetz, F.A. and E. Jeanes. 2004. *Preliminary draft* Bull trout in the nearshore. U.S. Army Corps of Engineers, Seattle District, Washington.

Graham, P. J., Shepard, B. B., and Fraley, J. J. 1981. Use of stream habitat classifications to identify bull trout spawning areas in streams. Pages 186-190 *in* N. B. Armantrout. Acquisition and utilization of aquatic habitat inventory information. American Fisheries Society. Portland, Oregon.

Gress, F., P.R. Kelly, D.B. Lewis, and D.W. Anderson. 1980. Feeding activities and prey preference of brown pelicans breeding in the Southern California Bight. Administrative report, California Department of Fish and Game, Sacramento, California.

Habeck, J.R. 1961. The original vegetation of the mid-Willamette valley, Oregon. Northwest Science 35(2):65-77.

Hamer, T.E. and S.K. Nelson. 1995. Characteristics of marbled murrelet nest trees and nesting stands. *In*: C.J. Ralph, G.L. Hunt, M.G. Raphael, and J.F. Piatt (Tech. eds.), Ecology and Conservation of the Marbled Murrelet. Gen. Tech. Rept. PSW-GTR-152. Albany, California: Pacific Southwest Experiment Station, Forest Service, U.S. Dept. of Agriculture; 420 pp.

Hamer, T., S.K. Nelson, and T.I. Mohagen II. 2003. Nesting chronology of the marbled murrelet in North America. February 6, 2003. Prepared for the USFWS, Portland, Oregon. 21 pp.

Hammond, P.C. 2001. 2000 Study of the Fender's blue butterfly (*Icaricia icarioides fenderi*) in Benton, Polk, and Yamhill Counties, Oregon. Unpubl. Rept. to Oregon Natural Heritage Program and U. S. Fish and Wildlife Service.

Hammond, P. C. 1998. 1997 Study of the Fender's blue butterfly (*Icaricia icarioides fenderi*) in Benton, Polk, and Yamhill Counties, Oregon. Unpubl. Rept. to Oregon Natural Heritage Program and U. S. Fish and Wildlife Service.

Hammond, P.C. 1996. 1995 Study of the Fender's blue butterfly (*Icaricia icarioides fenderi*) in Benton, Polk, and Yamhill counties. Unpublished report to the Oregon Natural Heritage Program and U.S. Fish and Wildlife Service. 25pp.

Hammond, P.C. 1994. 1993 Study of the Fender's Blue Butterfly (*Icaricia icarioides fenderi*). Unpublished report to the Oregon Natural Heritage Program and U.S. Fish and Wildlife Service. 56pp

Hammond, P.C. and M.V. Wilson. 1993. Status of the Fender's blue butterfly. Unpublished report on file at U.S. Fish and Wildlife Service, Oregon State Office. 66pp.

Hammond, P.C. and M.V. Wilson. 1992. Fender's blue butterfly populations: habitat descriptions and threats to survival. Unpublished report to the Oregon Natural Heritage Program, Portland, OR. 26pp.

Hartzell, K. E, and K. Popper. 2002. Warner Sucker Progress Report-2001 Findings. Unpublished Report to Bureau of Land Management.36pp.

Harvey, T.E. 1988. Breeding biology of the California clapper rail in south San Francisco Bay. 1988 Transactions of the Western Section of the Wildlife Society 24: 98-104.

Hayes, J.P. 1980. Fish of the Warner Valley. Pages 131-137 in C. Gilman and J.W. Feminella, editors. Plants and animals associated with aquatic habitats of the Warner Valley. Oregon State University, Corvallis, Oregon.

Hitchcock. C.L., Cronquist, A., Ownbey, M. and Thompson, J.W. 1961. Vascular Plants of the Pacific Northwest, Part 3: Saxifragaceae to Ericaceae. University of Washington Press. Seattle and London. 330 pp.

Horn, M.H., P.A. Cole, and W. E. Loeffl er. 1996. Prey resource base of the tern and skimmer colonies at the Bolsa Chica Ecological Reserve, Orange County, and the Western Salt Works, south San Diego Bay. U.S. Fish and Wildlife Service unpublished report (Final report on Grant# 14-48-001-95586), Carlsbad, CA, 54 pages.

Horn, M.H. and W.M. Dahdul. 1998. Prey resource base of the tern and skimmer colony at the Western Salt Works, south San Diego Bay, during tbe 1997 breeding season. U.S. Fish and Wildlife Service unpublished report (Final report on Grant# 3 14-48-0001-95586), Carlsbad, CA, 27 pages.

Horn, M.H. and W.M. Dahdul. 1999. Prey resource base of the tern and skimmer colony at the Western Salt Works, south San Diego Bay, during the 1998 breeding season. U.S. Fish and Wildlife Service unpublished report (Final report on Grant# 14-48-0001-95586), Carlsbad, CA, 36 pages.

Huff, M., P. Jodice, J. Baldwin, S. Miller, R. Young, K. Ostrom, C.J. Ralph, M. Raphael, C. Strong, and C. Thompson. 2003. Draft marbled murrelet effectiveness monitoring Northwest Forest Plan 2002 annual summary report. 29 pp.

Hull, C.L., G.W. Kaiser, C. Lougheed, L. Lougheed, S. Boyd, and F. Cooke. 2001. Intraspecific variation in commuting distance of marbled murrelets (*Brachryamphus marmoratus*): ecological and energetic consequences of nesting further inland. Auk 118:1036-1046.

Humboldt State University. 2002. Final Report: 2002 Snowy Plover Breeding in Northern California with Emphasis on Humboldt County.

Isaacs, F.B. and R.G. Anthony. 2002. Bald eagle nest locations and history of use in Oregon and the Washington portion of the Columbia River recovery zone, 1971 through 2002. Oreg. Coop. Wildl. Res. Unit, Oreg. State Univ., Corvallis. 20 pp, 6 tables, 2 figures, 1 appendix.

Jaques, D.L. 1994. Range expansion and roosting ecology of non-breeding California brown pelicans. Unpublished Masters thesis, University of California, Davis, California.

Jaques, D.L., and D.W. Anderson. 1987. Conservation implications of habitat use and behavior of wintering brown pelicans (*Pelecanus occidentalis californicus*). University of California. Davis, California.

BROWN PELICAN

Project related activities are of most concern for brown pelicans. East Sand Island currently represents the largest brown pelican roost location in the Pacific Northwest. Survey results from 2002 (Wright 2003) indicate that well in excess of 10,800 brown pelicans roosted at East Sand Island during a period of maximum use. The western end of East Sand Island represents the area where the majority of brown pelicans roost. However, brown pelicans may occur on the beach areas on the upstream end of the island also. Brown pelicans are in attendance at East Sand Island from the latter part of April into December with largest numbers expected to occur in July, August and September.

The Columbia River estuary and nearshore waters provide a rich foraging ground for brown pelicans. Pelicans are most numerous at the mouth proper with few birds venturing upstream of the Astoria-Megler Bridge. Collis et al. (2000) surveyed five pile dikes in the lower Columbia River plus the North, South and A jetties to record brown pelican occurrence. Dates of the survey were July 31 – October 31, 2000. Brown pelican use of pile dikes was most prevalent for the dike on the upstream tip of West Sand Island (CRM 4.01) and for the downstream dike on East Sand Island (CRM 4.47). These two pile dikes are nearest the main roost location on the western end of East Sand Island. Lesser use occurred at the downstream pile dike on West Sand Island (CRM 3.0), the upstream dike on East Sand Island (CRM 5.15) and the Chinook pile dike (CRM 6.37).

Pelican use of pile dikes was primarily for perching activities. Use was most prevalent from mid-August to mid-September. Some foraging by brown pelicans was observed in waters adjacent to the pile dikes. Distribution of brown pelicans was relatively uniform along the length of pile dikes at CRM 4.47 and 5.15 (East Sand Island). Pelicans primarily used the nearshore portion of the pile dike at CRM 3.0 (West Sand Island) and the offshore portion of the pile dikes at CRM 4.01 (West Sand Island) and 6.37 (Chinook). Mean numbers of pelicans at pile dikes ranged from 3.2 to 18.4 birds. The lower West Sand Island pile dike (CRM 3.0) supported a high weekly mean of 20+ brown pelicans during the week of September 11, 2000 (Collis et al., 2000). The upstream pile dike at West Sand Island (CRM 4.01) supported weekly means of 25-40+ brown pelicans from mid-August to mid-September 2000 (Collis et al., 2000). The downstream pile dike on East Sand Island (CRM 4.47) typically supported a weekly mean of 10-20+ brown pelicans throughout the study period with a high weekly mean of approximately 55 pelicans for the week of August 11, 2000 (Collis et al., 2000). The upstream pile dike at East Sand Island (CRM 5.15) supported a weekly mean of 20 brown pelicans on three occasions during the study period. The Chinook pile dike (CRM 6.37) supported the fewest brown pelicans, typically weekly means were 10 or less birds (Collis et al., 2000).

Cormorant Excluders – Spike Strips

The Corps of Engineers continues to work toward compliance with NOAA Fisheries directives in the BIOP for the Dredged Material Management Plan to reduce cormorant predation on juvenile salmonids. As directed, the Corps initiated emplacement of cormorant excluders on pile dikes in 2000. Placement of cormorant excluders in 2000 occurred in the upper estuary from Miller Sands to Puget Island. For 2001, three pile dikes in the lower estuary (CRM 4.01 (West Sand), 5.15 (East Sand Island upstream dike) and 6.37 (Chinook dike), plus six upstream pile dikes from CRM 24 to CRM 51 had excluders emplaced. The upstream pile dikes are outside the normal range of brown pelicans in the Columbia River estuary.

The pile dikes at CRM 4.01, 5.15 and 6.37 fall in the area of use by brown pelicans. Collis et al. (2000) observed brown pelicans using all three of these pile dikes for perching activities. The Corps placed cormorant excluders on these three pile dikes in May-early June 2001 and O&M of these excluders would mimic that timeframe. Thus, we would attempt to conclude construction activities at these locations prior to brown pelicans arriving in numbers in the general area thus lessening disturbance potential for brown pelicans either perching on the pile dike or roosting on East Sand Island.

To provide for continued pelican use of these pile dikes, the Corps will leave unexcluded 40 piling in the nearshore area of the pile dike at CRM 4.01, 40 piling in four groups of 10 each uniformly distributed throughout the dike at CRM 5.15 and 20 piling in the offshore portion of the pile dike at CRM 6.37 for pelican perch sites. The retention of open pilings, plus king piles and end dolphins that are also not fitted with excluders should leave ample perching space for brown pelicans at each pile dike location. The number selected is based generally on the mean high weekly count for these pile dikes.

Bald eagle kites and eagle silhouette decoys would not be used at the three lower river (CRM 4.01, 5.15 and 6.37) pile dikes.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

East Sand Island Habitat Management Actions (Tillage and Herbicide)

Habitat management actions on East Sand Island, specifically herbicide action (one day) and tillage (~3 days) are focused on the six-acre upland location near the upstream tip of East Sand Island. Tillage would occur in late March to the first week in April, prior to the return of brown pelicans to the roosting location. Herbicide application would occur in early September. Herbicide application in September 2002 resulted in the flushing of 445 brown pelicans or 1.4% of the total observed flushes in 2002 (Wright 2003). The birds flushed were on the upstream beach where equipment was landed to access the tern colony location.

Wright (2003) observed that 60% of the total brown pelicans flushed (4,147) by research activity re-landed within 200 meters of their original location, 36 percent re-landed elsewhere on East Sand Island and the fate of the remaining 4 percent was undetermined. The response to flushing is not considered detrimental to individuals or the species. By comparison, bald eagles accounted for 16,481 flushes or approximately 52 percent of the total flushes observed during the research effort.

The brown pelican monitoring data (Wright 2003) indicates that brown pelicans continued to use East Sand Island in substantial numbers even though substantial disturbance, primarily due to natural factors, was recorded. No abandonment of the island as a night roost location is apparent from her data (Wright 2003). The number of brown pelicans using East Sand Island in 2002 more than doubled compared to 2001 numbers. The maximum level of brown pelicans roosting on East Sand Island was approximately 4,400 in 2001 and increased to 10,800 in 2002 (Wright 2003).

Determination: Habitat management actions involving tillage would have no affect on brown pelicans. Habitat management actions involving herbicide application may affect, but are not likely to adversely affect brown pelicans.

Colonial Water Research Activities at East Sand Island

Caspian tern and double-crested cormorant research activities at East Sand Island will be comparable to previous years efforts. Enclosure 1 describes the proposed colonial waterbird research actions in more detail plus provides an entire section on measures associated with each research task to minimize disturbance to brown pelicans. Disturbance levels for brown pelicans are identified for each research measure. Caspian tern research activities will be focused on the tern colony at the upland site on the upstream end of East Sand Island.

It is anticipated that Caspian tern research activities, e.g., observations from blinds to determine tern numbers, distribution and productivity (Task 1.1), diet composition (Task 1.2), and banding of terns prior to fledging (Task 1.3) occurring at the tern nesting colony on East Sand Island may result in occasional disturbance to brown pelicans loafing on the island, particularly the east beach. These research activities are distant from the western portions of East Sand Island where the vast majority of brown pelicans roost. The researchers would ingress and egress the island according to protocol developed in previous years. Visitors would be briefed as to area closures and the limitations imposed upon their actions.

Completion of Task 1.1, determination of tern numbers, distribution of nesting pairs and productivity of Caspian terns nesting on East Sand Island, is confined to the tern colony area at the upstream (east) end of the island. These research activities may result in occasional disturbance to brown pelicans loafing on the island, particularly the east beach. These research activities are distant from the western portions of East Sand Island where the vast majority of brown pelicans roost. The researchers would ingress and egress the island according to protocol developed in previous years. Visitors would be briefed as to area closures and the limitations imposed upon their actions as has occurred in previous years. The increased population of roosting brown pelicans at East Sand Island has led to their increased presence around the tern colony location, particularly the east beach.

Task 1.1 also requires the presence of two colony monitors on East Sand Island for the duration of the tern nesting season. These monitors estimate colony size on a daily basis and collect data on factors affecting nesting success. Although the vast majority of brown pelicans roost at the opposite end of the island from the Caspian tern colony, occasionally, particularly late in the season and when winds are from the west, dozens or even hundreds of pelicans will roost on the beach adjacent to the tern colony. On occasion, roosting pelicans can make it impossible for colony monitors to access observation blinds next to the tern colony and collect data without causing some pelicans to flush and move down the beach. Also, on occasion small numbers of pelicans roost on the landing beach next to the tern colony, making it difficult for researchers or management agency visitors to land on or leave the island without flushing some pelicans. Under these circumstances, movements to and from the blinds or on and off the island will be minimized to avoid unnecessary disturbance to pelicans on the adjacent beach. If there is a risk of flushing as many as 500 pelicans at one time, activities will be postponed until pelicans have left the area. If > 1,000 pelican flushes occur during a one week period due to research activities, research activities on East Sand Island will cease until new protocols to minimize pelican disturbance are agreed upon with the U.S. Fish and Wildlife Service.

The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

Task 1.2 entails the determination of diet composition and smolt consumption rates of Caspian terns nesting on the East Sand Island colony. This task requires collection of fish identification data from visual observations of bill loads brought into the colony by adult terns. This task is performed by colony monitors from the blinds adjacent to the tern colony. Measures used to minimize pelican disturbance from this activity are as described for Task 1.1. This task also involves the collection of adult terns returning to the colony with bill loads of recently caught fish. These adult terns are collected over land (so that bill loads aren't lost when they drop into the river and sink) using shotguns. This activity can be conducted in the midsection of East Sand Island without causing disturbance to either the Caspian tern breeding colony or the double-crested cormorant breeding colony, because collecting activities occur at least a quarter of a mile from these colonies. On occasion, pelicans roosting on the beach adjacent to the collection site can make it impossible to collect adult terns without flushing some pelicans. On these occasions, every effort will be made to minimize disturbance to roosting pelicans, and if there is a risk of flushing as many as 500 pelicans at one time, activities will be postponed until pelicans have left the area.

The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

Task 1.3 entails the development of a demographic model for the Pacific Coast population of Caspian terns that will predict future population trends from current life table data. This task requires resightings of color-banded adult terns on the colony by observers in blinds adjacent to the colony. Procedures to minimize pelican disturbance from this activity will be as described for Task 1.1. This task also requires the banding of fledgling terns late in the breeding season with unique color combinations of plastic leg bands. Banding fledgling terns on the East Sand Island tern colony can potentially result in flushing brown pelicans if they are roosting on the adjacent beach. In this event, every effort will be made to minimize the number of flushed pelicans associated with this activity, and if there is a risk that as many as 500 pelicans could be flushed, the banding activity will be postponed until pelicans have left the area.

The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

Task 1.4 involves the evaluation of various factors that may influence the vulnerability of juvenile salmonids to predation by Caspian terns nesting at East Sand Island.
This task requires reading smolt PIT tags on the surface of the tern colony once all nesting activity has ceased at the end of the breeding season. This task is accomplished by NOAA Fisheries in the fall, activities that are not supervised by the OSU/CRITFC/RTR/USGS research crew. This task may also include mooring a net pen in the vicinity of East Sand Island as part of a pilot study testing the feasibility of this technique; activities that could potentially flush brown pelicans roosting on East Sand Island will be avoided.

The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

Dan Roby, Oregon State University, and research associates will also be investigating the double-crested cormorant colony on East Sand Island in 2003-2004 (Enclosure 1) to determine productivity and assess factors affecting nesting success. Their research will also include a monitoring effort to determine if research activities are having an adverse effect on brown pelicans (Enclosure 1). Measures to minimize disturbance to brown pelicans arising from double-crested cormorant research and brown pelican monitoring are described in detail in Enclosure 1.

Task 2.1 entails the determination of numbers, distribution, and productivity of double-crested cormorants nesting at the East Sand Island colony. This task requires aerial photography of the cormorant colony, photographs that are taken from a small fixed-wing aircraft flying at altitudes of about 1,000 feet. Observations of the behavioral reactions of brown pelicans roosting on East Sand Island indicate that fixed-wing aircraft flying over the island at altitudes exceeding 500 feet do not cause pelicans to flush (Wright 2003).

This task also requires researchers to access the observation tower adjacent to the cormorant colony in order to monitor success of cormorant nests in productivity plots and collect data on factors affecting nesting success. All cormorant research activities on the East Sand Island colony will be restricted to the observation tower and blinds near the eastern end of the colony (east of the big stump). Most cormorants nest to the west of the big stump and most brown pelicans roost to the west of the stump as well. Once cormorants arrive on the colony (mid-April), the observation tower and other blinds will be accessed in such a manner as to minimize the potential for disturbance of nesting cormorants (Anderson and Keith 1980).

Roosting pelicans arrive on East Sand Island after cormorants initiate nesting and human activities that disturb nesting cormorants also tend to disturb roosting pelicans because of facilitated alarm behavior. Consequently, efforts by cormorant researchers to minimize disturbance to nesting cormorants should also minimize disturbance to pelicans. Access to the observation tower and blinds in the cormorant colony will occur only at low tide. Researchers will approach the cormorant colony from the direction of the tern colony (from the east), following the water's edge along the north shore of the island. This approach minimizes disturbance to nesting gulls (gulls disturbed into flight tend to cause behavioral responses in nesting cormorants and roosting pelicans as well).

Above-ground tunnels constructed of silt fencing will allow researchers to access the tower and observation blinds from the north shore of the island without disturbing nesting cormorants or roosting pelicans. Cormorant nesting success data will be collected from the series of blinds adjacent to the cormorant colony in order to minimize the effects of observer presence on cormorant nesting behavior. This will also ensure that impacts on behavior of roosting pelicans are minimized. Repairs to the pre-existing blinds will be completed prior to the initiation of cormorant nesting, which is well before the first brown pelicans arrive on East Sand Island.

On occasion, particularly late in the cormorant breeding season, brown pelicans roost in significant numbers along the north beach, making it impossible to access or depart the tunnel system and the blinds without causing brown pelicans to flush. This situation is most prevalent when the wind is from the south. In these circumstances, every effort will be made to minimize the number of pelicans flushed during access to or departure from the tunnel system. If there is a risk of flushing as many as 500 pelicans at one time, activities will be postponed until pelicans have left the area.

The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

Task 2.2 entails the determination of diet composition and smolt consumption rates of double-crested cormorants nesting at the East Sand Island colony. This task requires the collection of adult cormorants as they return to the East Sand Island colony from foraging trips. Diet composition is determined by the composition of stomach contents from adults collected throughout the nesting season. Tissue samples from collected adults are also used to assess diet composition over more extended timeframes. Adult cormorants are collected using shotguns, and birds are usually shot over water. Most cormorants are collected near the end of the middle pile dike on East Sand Island as they fly toward the colony to the west, or near the eastern pile dike at West Sand Island. Both of these collection sites are over a quarter of a mile from the cormorant colony on East Sand Island and collection activities do not cause disturbance to cormorants on the colony. On occasion, particularly late in the cormorant breeding season, brown pelicans roost in significant numbers near these collection sites, making it impossible to collect adult cormorants without causing pelicans to flush. In these circumstances, every effort will be made to minimize the number of pelicans flushed, and if there is a risk of flushing as many as 500 pelicans at one time, activities will be postponed until pelicans have left the area.

The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

MONITORING BROWN PELICANS ROOSTING ON EAST SAND ISLAND

The objective of the monitoring plan is to assess the effects of research and monitoring activities on the behavior and numbers of California brown pelicans roosting on East Sand Island in the Columbia River estuary. The western end of East Sand Island is currently the site of the largest night roost of brown pelicans in the Pacific Northwest and California brown pelicans are listed as endangered under the U.S. Endangered Species Act.

Although brown pelicans have not nested on East Sand Island, there is concern that research activities on East Sand Island and other anthropogenic and natural disturbances may cause roosting brown pelicans to partially abandon the roost there. Also, in the last two years brown pelicans roosting on East Sand Island have displayed more and more nesting behavior, including courtship, nest-building, copulation, and broodiness, suggesting that active nesting by pelicans on the island may occur in future years.

The proposed monitoring of brown pelicans is designed to measure (1) effects of research activities on the behavior of roosting brown pelicans, (2) numbers and distribution of brown pelicans using East Sand Island as a roost, (3) behavioral responses by roosting brown pelicans to other potential disturbances (both anthropogenic and natural), and (4) any nesting behavior by brown pelicans on East Sand Island. Products from these brown pelican monitoring activities will include counts of the numbers of California brown pelicans roosting on East Sand Island, documentation of the behavioral responses of brown pelicans to researcher activities, monitoring of the effects of other potential disturbances on brown pelicans nesting on East Sand Island, detection of any potential nesting by brown pelicans on East Sand Island, and monthly reports on findings from the monitoring. Brown pelican monitoring will occur during the nesting seasons of Caspian terns and double-crested cormorants on East Sand Island, from April to August, in 2003 and 2004.

The primary activity in the pelican monitoring program will be the keeping of detailed records of disturbances apparently associated with researcher activities on East Sand Island. In addition, similar detailed records will be kept on observed disturbances to brown pelicans from natural factors (e.g., bald eagles) and anthropogenic factors (other than researchers; e.g., recreational boaters, Coast Guard helicopters). The primary metric of the magnitude of disturbance to pelicans will be the number of pelicans flushed and the number of distinguishable flushing events. Pelicans usually flush in groups of more than one individual because they usually roost in groups and social facilitation frequently induces more pelicans to flush when a nearby individual takes flight. The following data will be recorded for each flushing event: (1) number of pelicans flushed, (2) location on the island where the event took place, (3) researcher activity, natural factor, or other anthropogenic factor that elicited the event, (4) time and date of the event, (5) subsequent behavior of the flushed pelicans (i.e., re-landed on the island < 200 yds from the original roost site, re-landed on the island > 200 yds from the roost site, left the island altogether, or unknown fate). All researcher-related disturbances to brown pelicans and any disturbances caused by other factors and observed during researcher activities will be recorded.

Trained personnel from the OSU/CRITFC/RTR/USGS research crew will conduct weekly, boat-based censuses of roosting brown pelicans throughout East Sand Island. These counts will be conducted late in the evening, when previous research results indicated that maximum numbers of brown pelicans can be counted on the island (Wright 2003). These counts will be used to obtain seasonal trends for each coastline sector of the island. Weather conditions and tide stage during each evening count will be noted. Similar methods have been employed to evaluate night roosting and address questions of disturbance effects at other roost sites on the Pacific Coast (Jaques and Anderson 1987, Jaques and Anderson 1988, Jaques et al. 1994, Jaques and Strong 1996, Jaques 1998).

In addition to boat-based surveys, counts of pelicans will be conducted at low tide from the observation tower next to the cormorant colony a minimum of once each week. Counts of roosting pelicans from the observation tower are incomplete, because some primary pelican roosting areas on East Sand Island are not visible from the tower. Thus it is only possible to obtain a complete census of pelicans roosting on the island using boat surveys. Nevertheless, the area of the island visible from the observation tower is where most pelican nesting activity has been observed, so observations of pelicans from the tower should detect any progressive increase in nesting behavior by pelicans. In 2002, most nesting behavior by brown pelicans was observed immediately adjacent to the observation tower. In 2003 and 2004, any nesting behavior by brown pelicans in the vicinity of the tower will be monitored and if any active nesting (i.e., egg-laying) is detected, it will be reported to the U.S. Fish and Wildlife Service immediately.

Monthly summary reports of numbers and distribution of brown pelicans roosting on East Sand Island (simple spreadsheet format) will be provided to the U.S. Fish and Wildlife Service and the U.S. Army Corps of Engineers. U.S. Fish and Wildlife Service biologists will be welcomed to participate in monitoring as their schedules permit.

Brown pelican research efforts in 2002 accounted for 2,366 flushes (7.5% of total observed flushes). Relative to total research related flushes (4,147), pelican research comprised 57 percent of this total. The response of brown pelicans to disturbance activities was previously discussed in the text for East Sand Island habitat management activities (tillage and herbicide). Our determination for this task relative to brown pelicans is predicated upon the observed responses to disturbance.

Determination: The proposed action may affect, but is not likely to adversely affect brown pelicans.

WESTERN SNOWY PLOVER

Our assessment of plover use at project locations and determination for western snowy plovers relative to FY 2001-2002 actions at East Sand Island, Rice Island, and Miller Sands Spit is identical to that provided in the FY 1999 and 2000 consultation process.

Conclusion: The proposed action will have no effect on western snowy plovers.

OREGON SILVERSPOT BUTTERFLY

None of the proposed actions, or locations where they would be implemented, occurs in habitat occupied by this species. Please reference the October 26, 1998 biological assessment for background information on this species.

Conclusion: The proposed action would have no effect on the Oregon Silverspot Butterfly.

HOWELLIA

None of the proposed actions, or locations where they would be implemented, occur in habitat occupied by this species. Please reference the October 26, 1998 biological assessment for background information on this species.

Conclusion: The proposed action would have no effect on Howellia.

LITERATURE CITED

Anderson, D.W., and J.O. Keith. 1980. The human influence on seabird nesting success: Conservation implications. Biological Conservation 18: 65-80.

Isaacs, F. B. and R. G. Anthony. 2002. Bald eagle nest locations and history of use in Oregon and the Washington portion of the Columbia River Recovery Zone, 1971 through 2002. Oregon Cooperative Fish and Wildlife Research Unit, Oregon State University, Corvallis. 21 pp, 6 tables, 3 figures, 1 appendix.

Jaques, D.L. 1998. Monitoring Brown Pelican night roosts in Willapa Bay, Washington. Unpubl. Report to the U.S. Fish and Wildlife Service, Willapa National Wildlife Refuge. 9 pp.

Jaques, D.L., and D.W. Anderson. 1987. Conservation implications of habitat use and behavior of wintering Brown Pelicans. Unpubl. report. U.C. Davis, Public Service Research and Dissemination program.

____________, and D.W. Anderson. 1988. Brown Pelican use of the Moss Landing Wildlife Management Area; roosting behavior, habitat use, and human disturbance. California Dept. Fish and Game, Nongame Bird and Mammal Section Report. 58 pp.

____________, R.L. Lowe, and D.W. Anderson. 1994. Brown Pelican range expansion in the eastern North Pacific: Roles of tradition and climate change. Unpubl. Masters Thesis, University of California, Davis.

_____________, and C.S. Strong. 1996. Brown Pelican roosting patterns and responses to disturbance at Mugu Lagoon and other non-breeding sites in the Southern California Bight. Technical Report No. 54. USDI, National Biological Service, Arizona. 62 pp.

Wright, S.K. 2003. California brown pelicans on East Sand Island, Columbia River estuary, 2002. Annual Report to the U.S. Fish and Wildlife Service and U.S. Army Corps of Engineers from the USGS-Oregon Cooperative Fish and Wildlife Research Unit, Oregon State University, Corvallis, OR.

Table 2. Endangered Species Determinations for Colonial Waterbird Research, Monitoring and Habitat Management Actions at East Sand Island, 2003-2004 Plus.					
Research, Monitoring or Habitat Management Action	**Bald Eagle**	**Brown Pelican**	**Western Snowy Plover**	**Oregon Silverspot Butterfly**	**Howellia**
Caspian Tern Habitat Management - Tillage	May affect, not likely to adversely affect	No Affect	No Affect	No Affect	No Affect
Caspian Tern Habitat Management - Herbicide	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Cormorant Excluders - Spike strips	May affect, not likely to adversely affect	May affect, not likely to adversely	No Affect	No Affect	No Affect
Cormorant Excluders - Eagle kites and silhouette decoys.	May affect, not likely to adversely affect	No Affect (not used in lower estuary)	No Affect	No Affect	No Affect
Caspian tern hazing - upper estuary islands	May affect, not likely to adversely affect	No Affect	No Affect	No Affect	No Affect
Task 1.1 Determine numbers, distribution and productivity of Caspian terns at East Sand Island	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Task 1.2. Determine diet composition and smolt consumption rates of Caspian terns nesting at East Sand Island.	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Task 1.3. Develop demographic model of Pacific Coast population of Caspian terns (banding)	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Task 1.4. Juvenile salmonid vulnerability test (net pens)	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Task 2.1 Determine numbers, distribution and productivity of double-crested cormorants at East Sand Island	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Task 2.2. Determine diet composition and smolt consumption rates of double-crested cormorants nesting at East Sand Island.	May affect, not likely to adverscly affeet	May affect, not likely to adversely affect	No Affect	No Affect	No Affect
Monitor Brown Pelicans roosting on East Sand Island.	May affect, not likely to adversely affect	May affect, not likely to adversely affect	No Affect	No Affect	No Affect

Table . Estimates for Caspian tern juvenile salmonid predation at East Sand Island and projections for tern colony at Dungeness NWR.

Year	No. of Terns	Mean # Juvenile Salmonids Consumed by Terns	# juvenile salmonids consumed per tern	% Juvenile Salmonids in CATE diet	Estimated Number of fish consumed annually by one adult tern at ESI colony.	Notes:
2000*	**17026**	**6,700,000**	**394**	47	837	Data for East Sand Island colony - Collis et al. 2002. Caspian Tern Research on the Lower Columbia River FINAL 2000 Season Summary
2001*	**17964**	**5,800,000**	**323**	33	978	Note: From Collis et al. 2002b; page 9; 22% of chinook were yearling chinook and 20% were sub-yearlings.
2002*	**19866**	**6,500,000**	**327**	31	1055	Note: From Collis et al. 2003a
2003*	**16,650**	**4,200,000**	**252**	24	1051	Note: From Collis et al. 2003b; page 10; 26% of chinook were yearling chinook and 17% were sub-yearlings.
2004	**19,000**	**3,500,000**	**184**	17	1084	From Roby (2004) AFEP presentation.
Sum - 2000-2001	90506	**26,700,000**	1480	152	5006	
Mean**	**18,101**	**5,340,000**	**296**		**1001**	
Estimate for East Sand Island CATE Colony using data for 2000-2004						
CATE EIS min. pop. est.	**5,000**	**1,480,000**	**296**			
CATE EIS max. pop. est.	**6,250**	**1,850,000**	**296**			
CATE - curr. Avg.	**18,140**	**5,369,440**	**296**			
Estimate for Dungess CATE colony using average number of prey x 31.3% salmon in diet.						
2004	422	**132,218**	**313**	31.3	1001	Low population estimate for 2004 at Dungess NWR (Roby et al. 2004)
2004	586	**183,601**	**313**	31.3	1001	High population estimate for 2004 at Dungess NWR (Roby et al. 2004)
CATE EIS max. pop. est.	2000	**626,626**	**313**	29	1001	

Estimate for Northern Anchovy using average number of prey x 0.8% northern anchovy in tern diet.						
2004	422	3379	8	0.8	1001	158.40825
2004	586	4693	8	0.8	1001	219.96975
EIS Max.	2000	16000	8			750
Estimate for cod spp. using average number of prey x 0.02% cod spp. in tern diet.						
2004	422	84	0.2	0.02	1001	
2004	586	117	0.2	0.02	1001	
EIS Max.	2000	400	0.2			
Estimate for flatfish spp. using average number of prey x 0.2% flatfish spp. in tern diet.						
2004	422	845	2.0	0.2	1001	
2004	586	1173	2.0	0.2	1001	
EIS Max.	2000	4000	2.0			

Table . Endangered Species Act determinations for proposed Caspian tern habitat locations in California, Alternatives C and D.

	Brooks Is. Contra Costa Co.	Hayward Regional Shoreline Don Edwards NWR Alameda Co.
Mammals:		
San Joaquin kit fox (*Vulpes macrotis mutica*) (E)	No Effect	No Effect
riparian (San Joaquin Valley) woodrat (*Neotoma fuscipes riparia*) (E)	No Effect	No Effect
riparian brush rabbit (*Sylvilagus bachmani riparius*) (E)	No Effect	No Effect
salt marsh harvest mouse (*Reithrodontomys raviventris*) (E)	No Effect	May effect, not likely to adversely affect (HRS and DENWR)
Birds:		
California brown pelican (*Pelecanus occidentalis californicus*) (E)	No Effect	No Effect
California clapper rail (*Rallus longirostris obsoletus*) (E)	No Effect	May effect, not likely to adversely affect (HRS and DENWR)
California least tern [*Sterna antillarum* (=*albifrons*) *browni*] (E)	No Effect	No Effect
bald eagle (*Haliaeetus leucocephalus*) (T)	No Effect	No Effect
western snowy plover (*Charadrius alexandrinus nivosus*) (T)	No Effect	May effect, not likely to adversely affect (HRS and DENWR)
Reptiles:		
Alameda whipsnake (*Masticophis lateralis euryxanthus*) (T)	No Effect	No Effect
Critical Habitat for Alameda whipsnake	No Effect	No Effect
Giant garter snake (*Thamnophis gigas*) (T)	No Effect	N/A
Amphibian:		
California red-legged frog (*Rana aurora draytonii*) (T)	No Effect	No Effect
Fish:		
delta smelt (*Hypomesus transpacificus*) (T)	May effect, not likely to adversely affect	No Effect
Critical Habitat for delta smelt	No Effect	N/A
tidewater goby (*Eucyclogobius newberryi*) (E)	No Effect	No Effect

Table ???. Contd.		
Invertebrates:		
Critical Habitat: vernal pool invertebrates	No Effect	No Effect
Bay checkerspot butterfly (*Euphydryas editha bayensis*) (T)	N/A	No Effect
callippe silverspot butterfly (*Speyeria callippe callippe*) (E)	No Effect	No Effect
Lange's metalmark butterfly (*Apodemia mormo langei*) (E)	No Effect	N/A
Valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*) (T)	No Effect	N/A
longhorn fairy shrimp (*Branchinecta longiantenna*) (E)	No Effect	No Effect
vernal pool fairy shrimp (*Branchinecta lynchi*) (T)	No Effect	No Effect
vernal pool tadpole shrimp (*Lepidurus packardi*) (E)	No Effect	No Effect
conservancy fairy shrimp (*Branchinecta conservation*) (E)	No Effect	N/A
Plants:		
Critical Habitat: vernal pool plants	No Effect	No Effect
California sea-blite (*Suaeda californica*) (E)	N/A	No Effect
Presidio clarkia (*Clarkia franciscana*) (E)	N/A	No Effect
Antioch Dunes evening-primrose (*Oenothera deltoids* ssp. *howellii*) (E)	No Effect	N/A
Critical Habitat for Antioch Dunes evening-primrose	No Effect	N/A
Contra Costa goldfields (*Lasthenia conjugens*) (E)	No Effect	No Effect
Contra Costa wallflower (*Erysimum capitatum* ssp. *angustatum*) (E)	No Effect	N/A
Critical Habitat for Contra Costa wallflower	No Effect	N/A
Santa Cruz tarplant (*Holocarpha macradenia*) (T)	No Effect	No Effect
Critical Habitat for Santa Cruz tarplant	No Effect	N/A
large-flowered fiddleneck (*Amsinckia grandiflora*) (E)	No Effect	No Effect
pallid manzanita (*Arctostaphylos pallida*) (T)	No Effect	No Effect
soft bird's-beak (*Cordylanthus mollis* ssp. *mollis*) (E)	No Effect	N/A
palmate-bracted bird's beak (*Cordylanthus palmatus*) (E)	N/A	No Effect
robust spineflower (*Chorizanthe robusta* var. *robusta*) (E)	N/A	No Effect
showy Indian clover (*Trifolium amoenum*) (E)	N/A	No Effect

Species	Dungeness NWR	East Sand Island	Fern Ridge Lake	Summer Lake	Crump Lake
Western snowy plover, Oregon silverspot butterfly, Howellia					
Alternative A	N/A	No effect	N/A	N/A	N/A
Alternative B	N/A	No effect	N/A	N/A	N/A
Alternative C	N/A	No effect	N/A	N/A	N/A
Alternative D	N/A	No effect	N/A	N/A	N/A
Bradshaw's Lomatium					
Alternative A	N/A	N/A	No effect	N/A	N/A
Alternative B	N/A	N/A	No effect	N/A	N/A
Alternative C	N/A	N/A	No effect	N/A	N/A
Alternative D	N/A	N/A	No effect	N/A	N/A
Golden paintbrush, Willamette daisy, Kincaid's lupine, and Fender's blue butterfly					
Alternative A	N/A	N/A	No effect	N/A	N/A
Alternative B	N/A	N/A	No effect	N/A	N/A
Alternative C	N/A	N/A	No effect	N/A	N/A
Alternative D	N/A	N/A	No effect	N/A	N/A
Warner Sucker					
Alternative A	N/A	N/A	N/A	N/A	No effect
Alternative B	N/A	N/A	N/A	N/A	No effect
Alternative C	N/A	N/A	N/A	N/A	May affect, likely to adversely affect
Alternative D	N/A	N/A	N/A	N/A	May affect, likely to adversely affect

Table . Endangered Species Act determinations for proposed Caspian tern habitat locations in Oregon and Washington.					
Species	**Dungeness NWR**	**East Sand Island**	**Fern Ridge Lake**	**Summer Lake**	**Crump Lake**
Bald Eagle					
Alternative A	No effect	May affect, likely to adversely affect	No effect	No effect	No effect
Alternative B	No effect	No effect	No effect	No effect	No effect
Alternative C	No effect	May affect, likely to adversely affect	May affect, not likely to adversely affect	No effect	No effect
Alternative D	No effect	May affect, likely to adversely affect	May affect, not likely to adversely affect	No effect	No effect
Marbled Murrelet					
Alternative A	No effect	N/A	N/A	N/A	N/A
Alternative B	No effect	N/A	N/A	N/A	N/A
Alternative C	No effect	N/A	N/A	N/A	N/A
Alternative D	No effect	N/A	N/A	N/A	N/A
Bull Trout					
Alternative A	No effect	N/A	N/A	N/A	N/A
Alternative B	May affect, likely to adversely affect	N/A	N/A	N/A	N/A
Alternative C	May affect, likely to adversely affect	N/A	N/A	N/A	N/A
Alternative D	May affect, likely to adversely affect	N/A	N/A	N/A	N/A
Brown Pelican					
Alternative A	N/A	May affect, likely to adversely affect	N/A	N/A	N/A
Alternative B	N/A	No effect	N/A	N/A	N/A
Alternative C	N/A	May affect, likely to adversely affect	N/A	N/A	N/A
Alternative D	N/A	May affect, likely to adversely affect	N/A	N/A	N/A