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Linking Ecological Risk Assessment and Economic Benefits

Contract #68-W-99-042
Work Assignment 4-44

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April 22, 2005

Prepared for
National Center for Environmental Economics
Office of Policy, Economics and Innovation
US Environmental Protection Agency

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1. INTRODUCTION

1.1 Background

Ecological risk assessments are used to support an array of decisions across EPA programs, including, for example, setting national air quality standards, establishing site-specific waste clean-up goals, and specifying effluent guidelines for particular industries or limits for particular water bodies. For many of these decisions (depending on the statutory authority), EPA also has a mandate to assess the relative cost and benefits of proposed regulations to society. The current analysis of many rules is, however, sparse in the description of benefits related to ecosystem services and improved ecosystem functions. This is not surprising because describing the intrinsic worth of environmental services represents a significant challenge. In particular, to develop an improved ecological benefits analysis capability, methods must be developed to "translate" ecological assessment endpoints into descriptors that can be understood in terms of their societal value and benefit (whether or not such benefits can be explicitly monetized).

Ecological risk assessments may use a wide range of measures to characterize risks to organisms, populations, communities and ecosystems/ ecological functions. For example, ecologists have developed a suite of indices to measure community health and to measure the level of community impacts from stressors (such as species richness, diversity indices, and dominance by opportunistic species which is characteristic of a disturbed community). In some more detailed assessments, ecologists model changes in populations or, in some cases, changes in ecosystem composition (for example, using relative toxicities to various compartments of the ecosystem and/or using food chain models). Other, even more complex, systems dynamic models are also being developed that represent overall ecological resilience and sustainability.

However, in practice, needed data are often not available or systems are too complex to characterize risk at higher levels of biological organization; therefore, in regulatory contexts, ecological risks often have been characterized using hazard quotient methods that compare current or anticipated exposures to an ecologically relevant benchmark values (e.g., ambient water quality criteria).

While these assessments support the evaluation of risks to environmental values, the particular measures are often not directly or obviously linked to potential consequences for societal, or specifically economic, values. In other cases, the logical link is clear between the assessment endpoint and societal values (such as global biodiversity) but quantifying the link is difficult. Nonetheless, decision makers need ways to describe the value of protecting these environmental entities in order to make rational regulatory decisions.

1.2. Purpose of this Report

In November 2004, EPA issued a draft document entitled "NCEE Draft Ecological Benefits Assessment Strategy" (SAB Review version, November 11, 2004). That document outlines a series of action items to advance ecological benefits assessment within the EPA regulatory context. This report responds to particular actions identified in that report, specifically:

- "*Explore methods for expanding the use of ecological risk assessment information in economic benefits assessments*,"
- "*Create a catalogue of existing population models and develop guidance on model selection and use*,"

The efforts presented in this report will help develop ways to use available risk assessment techniques to describe impacts on ecosystems in terms that can be understood by decision makers and are useful for economic valuation.

With the goal of placing existing risk assessment techniques into context, Section 2 of this report presents a review of accepted ecological risk assessment techniques used by U.S. EPA as well as other government agencies. The techniques vary in their input data requirements, level of biological organization, endpoints, spatial scale, regulatory acceptance, and availability. Selected approaches are evaluated for their potential utility for benefits assessment. For example, an ecological risk model may provide endpoints that are easily assessed in economic benefit terms, but if data to support the model are not easily obtained or the model has not before been used in regulatory settings, than there may be practical limits to its use in benefits assessment. Section 3 discusses the availability of literature on economic valuation that could be applied to the types of endpoints each risk model evaluates. These data can serve as the basis of conducting an economic benefits assessment using the various ecological risk techniques discussed.

Sections 4, 5, and 6 present ecological benefits assessment exercises based on published ecological risk studies. Performing the valuation exercise for these case studies illustrates the type of analyses that can be conducted to inform decision makers of economic impacts for similar ecological risk scenarios. Finally, Section 7 discusses future directions to further develop this work, and to aid in a more unified approach to benefits analysis of ecological risks.

2. EVALUATION OF ECOLOGICAL RISK ASSESSMENT MODELS

A variety of ecological risk assessment techniques are employed in regulatory settings, with different ecosystems and scales of interest, and different goals. This section describes the approaches and models identified, and evaluates the applicability of the ecological models to conducting a range of risk assessments and their ability to provide output that can be economically valued. Examples of the selected approaches are included as a sample of ecological risk scenarios to which these techniques have been applied.

Risk assessment models used in government agencies were found and reviewed through risk assessment guidance documents, model documentation, case studies (where available), journal articles, and interviews with agency staff. Staff were identified and contacted in different programs within EPA, including National Center for Environmental Economics (NCEE), Office of Research and Development (ORD), Office of Pesticide Programs (OPP), Office of Water (OW), Office of Science and Technology (OST), and the Superfund program. In addition, officials from the National Oceanic and Atmospheric Administration (NOAA) and independent researchers affiliated with other government risk assessment efforts were interviewed. The models and case studies reviewed are outlined in Appendix A by the relevant species, ecosystem, and particular endpoints measured. Appendices B and C indicate guidance documents and all EPA and other government agency staff contacted in the course of identifying these approaches and related case studies.

2.1. Background On Ecological Modeling For Risk Assessments And Economic Valuation

Much of the recent literature on ecological risk assessment methods encourages the modeling of ecological effects to populations, communities, or ecosystems, in order to better represent risk at scales that are of interest in environmental management. Ecological models are tools that describe the effects of an environmental insult or perturbation (e.g. toxic chemical exposure, habitat fragmentation) to the levels of organization of interest, i.e. populations, communities, and ecosystems. Above the individual-level, they may be broadly classified into three groups: (1) population models, (2) ecosystem models, and (3) landscape models (Pastorok et al., 2002).

- *Population models* describe the effects on abundance or distribution of one or more species either at a single point in space and time or over a more extended area and period. Population models may be grouped into several categories: (1) scalar abundance, (2) life history, (3) individual-based, and (4) meta-population models. According to Pastorok et al. (2003, p. 945), individual-based models have limited potential for use in today's chemical risk assessments due to their limited applicability to environmental scenarios other than the ones for which they were developed. Among the remaining choices, life history models appear to have the greatest potential for widespread application in chemical risk assessments. An important feature of the life history model is that its typical output, decline in populations, is intuitive to decision makers and amenable to economic valuation.
- *Ecosystem models* take into account species interactions in addition to population abundance information. For example, food web models can show the impact that

decreases in one species will have on the surrounding species.

- *Landscape models* include all the features of population and ecosystem models, but also are spatially explicit. These can be particularly useful in assessments of terrestrial ecosystems that tend to show greater heterogeneity.

Current common methods of risk characterization, such as the use of hazard quotients, stop short at individual-level endpoints, and therefore do not offer the most relevant indicators of risk to environmental managers. Models that evaluate exposure alone (e.g., chemical fate and transport models) are important to the process of risk assessment but do not provide final endpoints that are relevant to risk managers. Figure 2-1 depicts the scale of biological organization measured by risk models, and each level's ecological relevance.

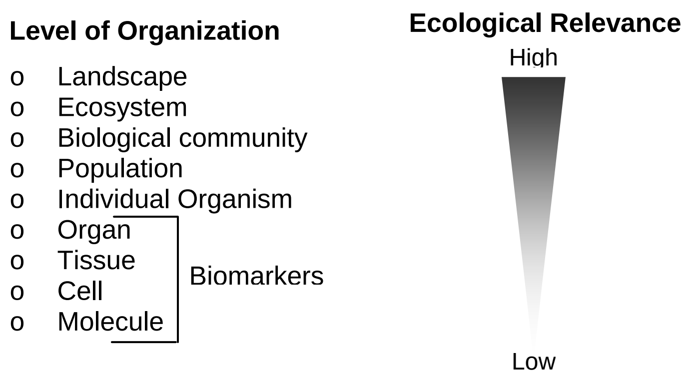

Figure 2-1. Hierarchy of Biological Endpoints, from Pastorok et al. 2002.

According to Pastorok et al. (2003, p. 968), using population models in risk assessments is more cost effective than using ecosystem or landscape models. Also, they are appropriate for chemical risk assessments in particular because the typical measurement endpoints in such an assessment (survival, growth, reproduction) can be easily related to population-level changes (Pastorok et al. 2002, p. 6).

2.2. Review of Ecological Risk Assessment Approaches and Models

Much of the recent work in ecological modeling in regulatory agencies such as EPA (OST, OW, OPP, and ORD), NOAA, and the Department of Energy (DOE), focuses on characterizing the dynamics of populations. EPA's Ecological Committee on FIFRA Risk Assessment Methods (ECOFRAM) and other forums have recommended the use of population modeling approaches in regulatory risk assessments (ECOFRAM, 1999). Interviews with EPA staff also confirmed a growing emphasis on population level modeling, as indicated by ORD's efforts to work with OPP on developing population-level approaches, and an EPA-sponsored workshop in 2004 on using population risk assessment models for the Superfund program. However, research revealed only a few examples to date of practical applications of population approaches, though their use will likely increase over time, as models are developed for a greater range of scenarios.

Table 2-1 shows the thirteen risk assessment approaches/models identified through the literature review and contacts with regulatory agencies, which are discussed below. Some of the techniques are simply approaches or steps to conducting a risk assessment, while others are distinct software packages. As Table 2-1 shows, these approaches encompass a range of ecosystems, required inputs, and scales of modeling. Some approaches are particular to a certain category of resources (e.g., water quality, or populations of terrestrial vertebrates). In addition, some approaches/models are designed for a specific stressor, while others are generally applicable for any environmental stressor. Most of the models are publicly available. A few of the techniques already include an economic assessment component, though they may differ in how benefit values are derived.

Table 2-1. Summary of Selected Ecological Risk Approaches

	Ecosystem	Specific to a Particular Resource	Population Modeling	Demonstrated Practical Uses	Freely Available	Includes Valuation
Salmon Population Modeling	**A**	T	T	T	T	
Acid Deposition Impacts on Brook Trout	**A**	T	T			T
AQUATOX	**A**	T	T	T	T	
NRDAM	**A**	T	T	T	T	T
PATCH	**T**	T	T	T	T	
Matrix Models	**A/T**		T	T	T	
RAMAS Ecotox	**A/T**		T			
RAMAS Metapop/GIS	**A/T**		T	T		
NWPCAM	**A**	T		T		T
EDT	**A**	T		T	T	
Level II Models	**A/T**	T		T	T	
Integrated Watershed Approach	**A**	T		T	T	T
Superfund Risk Assessments	**A/T**			T	T	

Key: A = Aquatic; T = Terrestrial

2.2.1. Population-Level Approaches for Aquatic Resources

Salmon Population Modeling

NOAA's Northwest Fisheries Science Center has performed very detailed modeling on salmon populations in the Pacific Northwest (pers. comm. Paul McElhany). Information on salmon population life history, reproduction, feeding patterns, and migration, is used to estimate risk of endangerment or extinction and to identify causes of risk. Data is also incorporate on stream/ocean temperature and current, availability of food, and loss to predators. A life-stage model is used to identify where maximum mortality is occurring, and given salmonid-species behavior. This allows the Center to identify spatially where the greatest risk lies to salmon populations. In addition to risk, the approach models demographic factors for the salmon species, such as growth rate, productivity, and mortality at different life stages. This model does have regulatory acceptance, as NOAA uses it for their research and population predictions.

Application of NOAA Salmon Population Modeling

Effect of pollution on fish diseases: Potential impacts on salmonid populations. This NOAA research paper investigates the interactions of general pollutant levels and disease susceptibility in salmon. Data are reviewed on juvenile salmon populations and mortality, the incidence of pathogens, and the relation of pollutant exposure to immunosuppression. (Arkoosh et al., 1998)

Acid Deposition Impacts on Brook Trout Fishing

This approach developed by Abt Associates (2002) evaluates water quality effects from different emissions scenarios, and the resulting change in brook trout biomass from acid rain deposition. Data for specific sites (Appalachian mountain streams) were used in its development, and a regression model links brook trout biomass to each stream's acid neutralizing capacity (ANC). Most inputs have already been entered in this model, including ANC, number of stream miles, a dose-response relationship for brook trout and pH changes, recreational use of the streams, and economic value of this recreational use. Changes in monetary value of brook trout fishing due to changes in brook trout populations are also modeled; thus, a valuation component is already included.

AQUATOX

EPA's AQUATOX model was developed as a management tool for addressing changes in freshwater systems from a variety of stressors (U.S. EPA, 2000a). The model functions at the ecosystem level, but it can also predict population-level effects. AQUATOX can predict the environmental fate of chemicals and their impact on several biological parameters such as dissolved oxygen, phytoplankton abundance, and fish populations. Acute and chronic toxicity to populations is evaluated. AQUATOX is flexible, and can be used for simple model ecosystems as well as complex, multi-species, multiple trophic level ecosystems. The model can include both plants and animals, and accommodates several species in each trophic group. Up to 20 toxicants can be input, along with their biodegradation products. Libraries of different chemicals are included, with several parameters for each including toxicity for a variety of animal species. However, users can choose to use their own chemical data. The program also has a link to BASINS, a GIS-based watershed modeling system. No economic component is included, but endpoints related to population level changes or water quality may be more easily assigned economic values because of their value for recreational use. The model is readily available, as it is provided for free from EPA's web site. It also has regulatory uses, as the AQUATOX model is currently being used and evaluated by the Office of Pesticide Programs in assessment of atrazine re-registration.

Case Study Using AQUATOX

An Adaptive Framework for Ecological Assessment and Management. This project was a retrospective study of the effects of the pesticide dieldrin to the largemouth bass population in the Coralville Reservoir, IA. This report represents a simple use of AQUATOX, as it only involves one chemical and one species. Different pesticide reduction scenarios were examined, and the viability of the bass population under each scenario estimated. Outputs include total dieldrin concentration, the dieldrin concentration in the bass, and probability of reduced population biomass. (Mauriello and Park, 2002)

Natural Resource Damage Assessment Models (NRDAM)
The NRDAM program was developed by NOAA to support damage assessments to aquatic resources under the Oil Pollution Act (U.S. Department of Commerce, 1996). For their program, NOAA utilized computer models created by the Department of the Interior under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), which estimate average damages from discharges of oil and other hazardous substances into aquatic environments. Two NRDA models exist, one tailored to coastal and marine environments (CME) and the other to the Great Lakes Environment (GLE). The models include information on reproductive rates and food chain interactions for various species (including plants, fish, birds, and marine mammals) within these two environments. The models use input data on the amount of an oil spill, toxicity of the chemical spilled, local physical characteristics (air and water temperature, wind and current speed), and biological characteristics of affected species (reproductive rate and food chain interactions) to yield damage estimates. An environmental fate component models pollutant concentration in water and sediment, which determines if access should be restricted to the site. Based on the lethality or non-lethality of the chemical, and food chain dynamics, the biological model predicts direct and indirect mortality to relevant species. A restoration component evaluates and selects appropriate restoration activities, and there is also a value component that assesses damage in terms of the value of lost recreational resource use. The NRDA models are publicly available through NOAA, and they have use in regulatory scenarios because they help determine damages under federal regulations.

Application of NRDA Model

Loon Mortality in New England – Mortality of the Common Loon in New England, 1987 to 2000. This report for Illinois DNR demonstrates use of the biological effects submodel of NRDAM for Coastal and Marine Environments, to determine annual loon mortality. Inputs include loon census figures, and biological data collected from dead loons in New England, such as body condition, age class, contaminant levels in tissue, and physical signs of trauma. The study determined approximate causes of death and importance of various natural and human sources of contamination. (Sidor et al., 2003)

2.2.2. Population-Level Approaches for Terrestrial Resources

Program to Assist in Tracking Critical Habitat (PATCH)
EPA ORD's Western Ecology Division developed PATCH to project populations for territorial terrestrial vertebrate species (U.S. EPA, 2004c). Stressors relate specifically to territory size and population pressure. It can be used for one species at a time, and only models females of a species. Inputting GIS layers for the territory, life history parameters, initial population size, and amount of travel by the species allows this model to provide spatially explicit information on population size as a function of time. A projection matrix is used to forecast population size based on fecundity and survival. Outputs include resulting fecundity, survival, immigration to new territories, and the number of individuals by life class, all of which can be provided in GIS structure. The software and documentation is available on EPA's web site and thus is easily accessible. As it has been developed by EPA, it also has use in regulatory scenarios evaluating effects of different landscape patterns or fragmentation on a species' ability to disperse and maintain viable population size.

Application of PATCH

An Analysis of Late-Seral Forest Connectivity in Western Oregon, U.S.A. PATCH was used to determine the effects of different landscape patterns on the dispersal success of different territorial wildlife species. Input included the dispersal capabilities and home-range sizes for the modeled species, and forest condition information for 8.3 million hectares of forested landscape in western Oregon, derived from satellite imagery. Dispersal success of the species was measured for baseline conditions of land cover and two alternative patterns based on public ownership or the Northwest Forest Plan reserve system. Output indicated that dispersal success was greater for species with larger dispersal distances and smaller home ranges. For species with shorter dispersal capability, the reserve system may not maintain habitat connectivity. (Richards et al., 2002)

2.2.3 Population-Level Approaches for Other Resources

Matrix Models
Matrix models, or life history models, function by projecting future population size or growth rate based on vital rates for a species (Pastorok et al., 2002 and 2003). They can come in deterministic or stochastic forms, and with or without density dependence. Populations are divided into life-stage or age classes, and survival rates, growth, and fecundity are input specific to each age or life-stage class. Inputs include initial population size by age or stage class, and growth, survival and fecundity for each class under baseline conditions, and under stressor conditions. Thus it is necessary to know the effect of the stressor on these rates for the species of interest, or at least a closely related species. Matrix models can predict population abundance by class or growth rate over time, and can isolate populations of interest, such as commercially viable adult fish populations. Its output of decline in populations is useful because it is intuitive to decision makers and amenable to economic valuation. They are easily available for use by

any agency when data are present on life history parameters, and they have been used by regulatory agencies including EPA and DOE. They are also incorporated into other population modeling or risk assessment programs, such as PATCH, and the RAMAS models.

Examples of Matrix Models

Estimation of Potential Population Level Effects of Contaminants on Wildlife. The report summarizes a DOE-funded project to improve methods to assess risks from contaminants to wildlife populations, using life history matrix models. The project created a toxicity database, to lessen the need to extrapolate toxicity response between species; a dose-response model; and matrix-based population models coupled with the dose-response models for realistic estimation of population-level effects. Both age and stage-structured matrix models were created, for various avian species, using 100-year simulations. The framework of the model; i.e. incorporating a dose-response model into an age or stage-based matrix model that includes effects on growth and reproduction, would be useful to measure changes to populations that could be economically valued. (U.S. DOE, 2001)

Evaluation of the efficacy of extrapolation population modeling to predict the dynamics of Americamysis bahia populations in the laboratory. An age-classified project matrix model was used to predict population response of Americamysis bahia (formerly Mysidopsis bahia) to the chemical stressor para-nonylphenol (suspected endocrine disruptor). A lab assay was also conducted to test actual population growth with the chemical. (Kuhn et al., 2001)

Predicting recovery of a fish population after heavy metal impacts. The bluegill sunfish population in a reservoir contaminated with selenium was modeled with a matrix model to predict time for population recovery. The model was based on data collected from ongoing monitoring at the lake, and includes density dependence. (Crutchfield and Ferson, 2000)

Projecting population-level response of purple sea urchins to lead contamination for an estuarine ecological risk assessment. A life stage matrix model was used to predict the effect of lead contamination on population growth rate of the purple sea urchin at an estuarine site, the Portsmouth Naval Shipyard. Survival, growth, and fertility were taken from control lab studies for this species or from literature. Effects of lead on these life history values were derived from a lab bioassay study. The matrix model projected population growth rate under different concentrations of lead, and indicated the growth rate was not significantly affected at the lead concentration found at the site. (Gleason et al., 2000)

RAMAS Models

Applied Biomathematics has produced several ecological modeling packages for purchase ranging from stage-based population models to landscape and GIS-based models. The RAMAS

Ecotoxicology model uses information on survival and fecundity, as well as dose-response models for a toxic chemical, to evaluate population-level parameters such as growth rate and population size in a specified time period. It is an extension on basic life history matrix models with its use of age or stage-based projections. Density dependence parameters can be included to influence population dynamics, and Monte Carlo simulations are used to predict populations and calculate the risk of adverse events. RAMAS Metapop and RAMAS GIS have similar vital rates data requirements as RAMAS Ecotoxicology, but they do not require information on the rates as impacted by a chemical stressor, and they also use information on dispersal rates, habitat requirements, and the landscape pattern to account for the effects of spatial distribution on population dynamics. They also have additional endpoints, including risk of species extinction, expected occupancy rates of landscape patches, and abundance in different parts of the landscape. While RAMAS Ecotox explicitly incorporates toxic stressor effects, RAMAS Metapop and RAMAS GIS can be used to assess risk from a chemical stressor by running two simulations: one with control vital rates to project population parameters in the absence of the stressor, and another simulation with impacted vital rates (i.e., reproduction, growth, and survival for the species under the influence of the stressor.) No economic component is included in the RAMAS models, but population abundance predictions could be used to calculate economic impact.

Although the RAMAS software applications can predict population-level effects and they are commercially available, we did not identify uses of RAMAS Ecotox by regulatory agencies. The development company, Applied Biomathematics, also indicated that RAMAS Ecotox was not used as widely as their metapopulation models, RAMAS Metapop and RAMAS GIS. Although these latter tools have been used by regulatory agencies, including EPA and state agencies, they have been used mainly for population viability analyses or for effects of habitat fragmentation, rather than for evaluating toxicological effects. However, as mentioned above, they can be used for risk assessment if simulations are run for control conditions, versus conditions in the presence of a stressor of concern.

Case Studies of RAMAS Models

RAMAS Ecotoxicology, Version 1.0a. User's Manual. Vol 2: Ecological Risk Assessment for Structured Populations. RAMAS Ecotox was used to apply a life history model to data for the fathead minnow. Input data included the effects of Mirex (an insecticide) on fertility, hatching success, survival, and density dependence. No extrapolation from other species was needed. The model was used to predict the risk of population decline. (Spencer and Ferson, 1998)

A Stochastic Population Model Incorporating PCB Effects for Wood Frogs (Rana sylvatica) Breeding in Vernal Pools Associated with the Housatonic River - Pittsfield to Lenoxdale, Massachusetts. RAMAS Metapop was used in an ecological risk assessment for the Housatonic River. The model was used to predict the risk of wood frog population decline due to PCB's, by using initial abundances and the vital rates of exposed and unexposed populations. The model was age and sex structured, and included demographic and environmental stochasticity, density dependence, and migration. Outputs included population sizes in 10 years, and predicted time for extinction. (U.S. EPA, Region 1, 2003)

2.2.4 Environmental Quality Models

National Water Pollution Control Assessment Model (NWPCAM)
NWPCAM is a national-scale water quality model for simulating the water quality and economic benefits that can result from various water pollution control policies (U.S. EPA, 2003b). This software models fate and transport of pollutants through surface water. Input parameters include nutrient and conventional loadings to water (non-point) and toxic pollutants for point sources only. Flow regimes are modeled for different periods of time to account for temporal variation. The model operates on a regional/national scale, and has links to continental U.S. water databases to include site parameters for a large range of water bodies. The final component of the model incorporates economic benefits for the predicted change in water quality, based on a willingness to pay survey for different levels of water quality. The key to this assessment is the ability to "map" changes in water quality parameters to changes in water quality categories (fishable, swimmable). Benefits can be calculated state-by-state at the State, local, or national-scale. While the model incorporates valuation of water quality, it does not value changes in wildlife populations or ecosystems. NWPCAM has regulatory use and has been used to support water pollution control policies, including the effluent guideline rulemaking process for Animal Feed Operations/Confined Animal Feed Operations (AFO/CAFOs), Stormwater Phase II, Meat and Poultry Products, and Construction and Land Development.

Case Studies Using NWPCAM

Environmental and Economic Benefit Analysis of Final Revisions to the National Pollutant Discharge Elimination System Regulation and the Effluent Guidelines for Concentrated Animal Feeding Operations. A 2002 EPA Office of Water benefits analysis employs the NWPCAM model to evaluate the benefit of improved water quality resulting from revising NPDES regulations for animal feeding operations. A separate groundwater model (GLEAMS) was used to estimate changes to nutrient loadings from regulated farms. NWPCAM was used to derive monetary value of these changes, based on data from the GLEAMS model, and a contingent valuation survey of how people value water quality improvements. (U.S. EPA, Office of Water, 2002)

Estimation of National Surface Water Quality Benefits of Regulating Concentrated Animal Feeding Operations (CAFOs) using the National Water Pollution Control Assessment Model (NWPCAM). NWPCAM was employed in this regulatory benefits assessment to determine change to nutrient loadings as well as monetary benefits, from alternative policy scenarios. Inputs include land use data, watershed and stream discharge, animal farm types and numbers, and nutrient loadings from each operation. Changes to water quality and benefits from each scenario were evaluated at the national level. (U.S. EPA, Office of Water, 2000b)

Ecosystem Diagnosis and Treatment Model (EDT)
Mobrand Biometrics has produced this software package that has been widely used to evaluate the ecosystem health of freshwater streams in the Pacific Northwest (Mobrand Biometrics, 2004). Inputs include many attributes of a freshwater system, such as stream type, flow rate, turbidity, sediment load, and pollutant load. Altogether 46 different attributes of freshwater systems are used in the model to evaluate ecosystem health. Ecosystem health, or habitat value, is based on a diagnostic species – either chinook, coho or chum salmon, and steelhead trout. Biological performance of the stream is evaluated based on diversity of life stages for the chosen species, productivity, and capacity, and these estimates are compared with values for a control system. The model was developed to help managers establish watershed plans and has been used widely by counties and other water resource planning groups in Washington and Oregon. A publicly accessible version of EDT is available from Mobrand Biometric Inc.'s web site, for evaluating selected water basins in Washington, Oregon, Montana, Idaho, and California.

Application of Ecosystem Diagnosis and Treatment

Pierce County Watershed Analysis. This report for Pierce County, Washington shows a step-by-step application of the EDT method for a watershed analysis. Inputs for this study include watershed dimensions, land cover, land uses, water characteristic data, and salmon life history. Outputs include the watershed's capacity to sustain the population, productivity and life history diversity of the species, to determine population viability. Analyses were also conducted to determine the relative importance of different input parameters on salmon population performance. (Mobrand Biometrics, Inc., 2001)

2.2.5 General Risk Assessment Approaches

Integrating Ecological Risk Assessment and Economic Analysis
The purpose of this approach was to identify and evaluate ecological risks and benefits on a watershed basis from various environmental stressors (U.S. EPA, 2003a). The approach is based on the collaboration of stakeholders in all phases of the risk assessment and economic analysis, from problem formulation through decision-making on alternative management options. Ecological endpoints are measures of community structure (e.g., Index of Biotic Integrity), derived from data collected over multiple years on the number and type of species present in the watershed. The sites chosen as case studies to conduct this approach had large, multiple-year, watershed-wide data available, which is not always the case in risk assessments. The community indices are then related to different land-use scenarios through regression analysis. Economic endpoints are described qualitatively, for example, the decline of agricultural employment. Economic values associated with the different management options are derived through individual preference studies of nearby residents, to determine willingness to pay (WTP) for the different outcomes. Stakeholders are thus made part of the benefits assessment process.

Case Studies Using Integrated Watershed Approach

Evaluating Development Alternatives for a High-Quality Stream Threatened by Urbanization: Big Darby Creek Watershed. Stressors evaluated related to different land uses, such as high density development, low density clusters, or all agriculture. A survey was conducted to derive endpoints of willingness to pay for economic and social services and environmental quality (local income base, distance to employment, open space, proximity to police and fire services, indexed quality of stream). (U.S. EPA, 2003a)

Valuing Biodiversity in a Rural Valley: Clinch and Powell River Watershed. Various stressors, including toxic chemicals, sedimentation, exotic species, and overexploitation were evaluated for their connection to native fish and mussel species reproduction and recruitment, and measures of community structure such as the Index of Biotic Integrity, by regression analysis. Endpoints are willingness to pay for complete recovery, partial recovery, or continued decline of native species. (U.S. EPA, 2003a)

Terrestrial and Aquatic Level II Models

The Office of Pesticide Programs has refined the use of probabilistic methods in their guidance on Level II Risk Assessment models, developed for evaluating effects of pesticides to terrestrial and aquatic resources under FIFRA registrations and re-registrations. Probabilistic risk assessment refers to methods of quantifying both variability and uncertainty in risk estimates. Terrestrial and aquatic models are available which estimate risk for various modes of exposure. Data on the type of chemical, timing and amount of application, and the amount of runoff and erosion are used to estimate the levels of exposure in the environment, and data on the chemical's toxicity are used define the probable distribution of an acute mortality level. Probabilistic models (Monte Carlo) are used to examine the probable frequency with which the exposure level exceeds the mortality level, leading to death of an organism, which allows an estimation of the acute mortality rates for an exposed population. No economic component is currently included, but because population-level effects are included (i.e., number of birds that die), this model does produce risk estimates that could be economically valued. This approach has regulatory uses and acceptance because it has been developed by OPP, and is used for pesticide risk assessments.

Application of Aquatic Level II Model

Probabilistic Models and Methodologies: Advancing the Ecological Risk Assessment Process in the EPA Office of Pesticide Programs: A Probabilistic Model to Assess Risks to Aquatic Organisms. A case study was conducted to show the use of a probabilistic model for evaluating pesticide effects in a small freshwater system. Bluegill, sunfish, rainbow trout and an extrapolated generic species are evaluated for their exposure and mortality in response to a generic pesticide applied to local fields. Acute mortality is derived as the endpoint for these species. (LaPoint et al., 2001)

Superfund Risk Assessments

There is no single "Superfund" ecological risk assessment model; these assessments generally follow US EPA ecological risk assessment guidance. This approach involves the steps of problem formulation, analysis (including an evaluation of exposures and development of stressor-response profiles), and risk/benefit characterization (see EPA's 1998 *Guidelines for Ecological Risk Assessment*.) However, examining applications of ecological risk assessment at Superfund sites is important because of the sheer number of assessments carried out in this program. The goal of these risk assessments can be to establish a baseline characterization of ecological risks, to determine risk that will remain under various site-clean up options, and/or to set clean-up goals for the site. These risk assessments can take place in any ecosystem, include multiple chemical stressors, and can evaluate any species for which toxicity can be estimated. Endpoints depend on what information is available for the Superfund study site, but typically include hazard quotient, measures of species diversity or richness, and measures of species health (i.e. number of impaired individuals).

Superfund Case Studies

Weighing the evidence of ecological risk from chemical contamination in the estuarine environment adjacent to the Portsmouth Naval Shipyard, Kittery, Maine, USA. A Superfund ecological risk assessment was conducted at a shipyard with metal contamination. Multiple species in different biological communities were evaluated for their effects from lead, including pelagic, epibenthic, benthic, salt marsh, and avian communities. For animal species, endpoints included tissue concentration, species density, richness, evenness, biomass, abundance, organism body size, mortality, and dietary exposure. For plant species, endpoints included plant morphology, biomass, tissue concentration, and species cover. Exposure endpoints were also modeled, including contaminant concentration in sediment and water. Levels of risk (high, medium, and low) were assigned by a combination of hazard quotient, and the weight of evidence for each endpoint. (Johnston et al., 2002)

Final Baseline Human Health and Ecological Risk Assessment: Lower Fox River and Green Bay, Wisconsin. Multiple contaminants were involved at this freshwater stream Superfund site: several PCBs, dioxins, furans, DDT and its metabolites, and the metals arsenic, lead, and mercury. Several species of fish, birds, and mammals were assessed, and endpoints included hazard quotients and Sediment Quality Threshold (SQT). (RETEC, Inc., 2002)

2.2.6 Applicability of Risk Assessment Approaches to EPA Ecological Risk Assessment and Benefits Assessment Requirements

Population Level Approaches for Aquatic Resources

Of the aquatic population models discussed above, only AQUATOX currently can be applied to a range of ecological risk assessment scenarios; this flexibility is included in its design. NOAA's Salmon Population Modeling is highly specific to salmon populations, and is not generally applicable to wider populations within freshwater ecosystems. The Brook Trout model also assesses a particular species, as well as a specific stressor, and thus is not generally applicable for use in other risk assessments. Both stressor and resources evaluated are specified in the NRDA models, but the CME and GLE are very specific to the aquatic ecosystems that they are designed to depict (coastal/marine systems and the Great Lakes). The benefit of existing these models is that the data input requirements are low, partially because they have been developed for certain sets of data (e.g., pH and brook trout numbers, or effects of oil on Great Lakes species and their food chain interactions), and thus for the scenarios that they are designed to evaluate, they are extremely useful, have endpoints that are economically valuable, and thus could be used for ecological benefits assessment. However, these scenarios are limited. In principle, the framework for these models could be applied for other species and stressors, though much additional data collection would be involved. AQUATOX holds the most potential for generalized risk assessment use, as it can incorporate a variety of pollutants and species. The model also has potential for use in regulatory scenarios including development of water quality criteria, total maximum daily loads (TMDL's), and analysis of management alternatives. However, as it can include a large set of parameters, a large set of data may be required particularly if modeling a more complex system.

Population Level Approaches for Terrestrial Resources

The matrix models and RAMAS software both be flexible enough to incorporate any species, and contain economically-relevant endpoints (population endpoints). Similar to the salmon and brook trout models, PATCH focuses on a specific group of species, and it has mostly been applied to forest-dwelling birds (Pastorok et al., 2002). PATCH also does not include a toxicological component, and thus in its current form would not be appropriate for conducting risk assessments where the stressor is chemical rather than habitat size related. However, life history parameters could be modified to reflect the impact of toxic effects. Matrix models are highly general, and can be employed for any scenario where population size and life history parameters are known, and where effects on growth, reproduction, and survival can be estimated for a given stressor of interest. In some cases, data on the affected vital rates may not be easily attainable, particularly if laboratory studies are not feasible on the species of interest. RAMAS models can also be applied for a range of risk assessment scenarios and economically-valuable endpoints. These life history models (including matrix models, RAMAS Metapop, and RAMAS GIS) have been used for a range of species, including birds, frogs, rabbits, and fish. However, the literature review did not find an example of a practical application of RAMAS Ecotox that supported regulatory analysis. The fact that RAMAS packages are not freely available may limit their use.

Environmental Quality Models

Both NWPCAM and the Ecosystem Diagnosis and Treatment assess changes only to resource quality; however, where resource quality itself can be economically valued, these risk output measures are useful for benefits assessment. Data input requirements for NWPCAM are feasible because it has links to EPA water discharge databases and national hydrography datasets. Mobrand's EDT software takes into account more characteristics of a particular system than NWPCAM, but this likely would require more data collection. The EDT also does not evaluate effects of specific contaminants, which are often of interest to regulatory programs. Another limit to EDT's usefulness is that its output looks specifically at ecosystem health in terms of quality for salmon and trout species. It is not applicable for evaluation of water quality as it relates to other species in a freshwater ecosystem such as plants, other fish species, or insects.

General Risk Assessment Approaches

The integrated watershed approach was designed to integrate ecological and economic benefits assessment, and thus has clearly relevant outputs. However, its development was due in part to the availability of particular data for the specified watersheds (e.g., IBI), and the economic assessment was conducted at a local scale, because it relied upon interviews of affected residents to determine their preferences. These features, and the data intensive nature of the approach, make it a powerful example of integrated assessment but limit its extrapolation or widespread application to other circumstances, Level II OPP models are intended specifically for pesticide assessments, and have imbedded scenarios relevant for pesticide exposures, but the probabilistic approach could be employed for any chemical of interest, as long as relevant fate and transport and toxicity data are available. Risk assessments employed in the Superfund assessments are applications of the general ecological risk assessment method developed by EPA, and thus could be used for any contaminants or sites. However, it remains a question whether the typical outputs of these assessments, hazard quotients or population/community indices, have applicability to economic benefits assessment.

2.3. Criteria for Selecting Models and Case Studies for Benefits Assessment

Following review of selected models and approaches from regulatory agencies, a few were chosen to test a benefits transfer approach. The purpose of performing this exercise for several types of risk assessment techniques was to provide a range of examples on how ecological benefits may be assessed (or how such assessments are limited).

To determine approaches/models that may be most relevant for developing combined ecological risk-benefits assessment case studies, the following criteria were derived from Pastorok et al. (2002), modified slightly for the current purposes:

1. Is the model already available? Has it been applied successfully in a practical setting?
2. Is the model appropriate for use in regulatory programs, that is, can it be applied to a wide variety of settings to be evaluated in a national regulation, and not overly specific to a particular ecosystem/setting? Does it consider issues of interest to EPA regulators?
3. Are the input requirements reasonably feasible?
4. Does the model examine relevant endpoints (i.e., endpoints that can be valued)?
5. Does the model already incorporate economic analysis, making additional development of economic benefits assessment unnecessary?

Applying these criteria to the models discussed in Section 2, several models were eliminated from further consideration:

- Acid Deposition Impacts on Brook Trout model deals with both a narrow stressor and resource, and already includes economic analysis
- NRDAM's approach applies to specific environments (coastal/marine or Great Lakes environment), and also already includes economic evaluation
- PATCH and NOAA's Salmon population modeling, as well as Mobrand's EDT, focus on very specific species and conditions, and not on chemical risk assessment.
- RAMAS Ecotox's use of population endpoints and explicit modeling of toxicity would be relevant for many risk assessment needs, but it has not been demonstrated in a regulatory setting and is not publicly available. The other RAMAS applications (RAMAS Metapop and GIS) have been used by EPA and state resource managers, though more often for population viability analyses or evaluations of habitat structure changes rather than risk assessments related to a chemical stressor. They may increase in use, as the software company gave their first workshop to EPA program staff in 2004. Similar to RAMAS

Ecotox, they are not publicly available and must be purchased.

- NWPCAM already has an economic component
- ORD's watershed approach is locale-specific, and already includes locally-derived economic assessment
- OPP's Level II models apply to chemicals applied to agricultural fields, and evaluate acute mortality to exposed populations. These models could be used for benefits assessment, if information is available regarding the affected species. However, further extensions of these models are being developed elsewhere within EPA at this time, and thus are not appropriate for inclusion as a case study here.

The remaining models are readily available, have known regulatory uses, and can be broadly applied to different ecosystems and resources at stake. These models (AQUATOX, matrix models, and Superfund risk assessments) differ in terms of data requirements and endpoints. Data input for both AQUATOX and matrix models require abundances of the species of interest, vital rates at relevant life stages, and toxic effects. Superfund risk assessments are usually dependent on the amount of information collected/ available, and may use vital rates, abundance, and toxicity, as data are available. In terms of endpoints, AQUATOX and matrix models provide population estimates (e.g., biomass, density, or number of individuals) after a given period of time. Such estimates may be evaluated from economics benefits literature, particularly if the species is of commercial or recreational use. Superfund risk assessment endpoints are typically individual-level, or provide an index of population or community effects. Though population impact estimates are not generally made, there can be a large variety of species and resources evaluated to fully represent environmental endpoints, which could potentially be linked to economically valuable services. EPA's Superfund program is also a driver in advancing ecological risk assessments, and thus it is important to assess the potential for linking these outputs to economic benefits.

3. IDENTIFYING ECONOMIC VALUATION STUDIES

A search of economic literature was conducted to identify studies with economic valuation endpoints that could potentially be linked to the endpoints assessed by the suite of ecological risk models reviewed. Two main environmental valuation databases were used to compile relevant references: Abt Associates' in-house BenLit database and the online Environmental Valuation Resource Inventory (EVRI). A keyword search was conducted for each database. Although the scope of this project did not include a comprehensive literature search to cover all of the potentially available studies (e.g., "gray literature"), the reviewed studies provide a good illustration of the economic valuation endpoints generally available from the resource valuation literature.

The keywords selected for each ecological risk model were determined based on the types of species and ecosystems that can be evaluated by a given ecological risk assessment model. The selected keywords for each ecological model are provided below:

- ***AQUATOX***: Includes fish, freshwater, and aquatic. Does not include marine, ocean, Atlantic, saltwater, salt, or estuary. All references referring to saltwater bays were also removed.
- ***Ramas Ecotox***: Includes references for all species and ecosystems.
- ***Ramas Metapop***: Includes references for all species and ecosystems.
- ***Level II OPP Models***: Includes references that specifically refer to agricultural studies.
- ***Matrix Models***: Includes references for all species and ecosystems.
- ***Ecosystem Diagnosis and Treatment:*** Includes salmon, steelhead, Chinook, and trout. Though the model only currently evaluates salmon and steelhead it is being developed for other types of trout. Therefore, trout references have also been included.
- ***Salmon Population Modeling:*** Includes salmon and Chinook.
- ***PATCH:*** Includes references that specifically refer to the forest ecosystem.
- ***Integrating Ecological Risk Assessment and Economic Analysis in Watersheds:*** Includes fish, freshwater, and aquatic. Does not include marine, ocean, Atlantic, saltwater, salt, or estuary. All references referring to saltwater bays were also removed.
- ***NRDAM:*** Includes Great Lakes and general aquatic references that are not freshwater or wetland.
- ***NWPCAM:*** Includes water quality, freshwater, river, and lake (does not include Great Lakes); recreational fishing, recreational swimming, or recreational boating.
- ***Acid Deposition Impacts on Brook Trout Fishing:*** Includes references that refer to Brook trout in Appalachian ecosystems.
- ***Superfund Risk Assessments:*** Includes references for all species and ecosystems.

3.1. Summary of the Literature Search Results

BenLit Database:
The BenLit database comprises 911 studies, most of which value aquatic resources, particularly fish species. Keywords used to search the BenLit database were selected to serve three purposes. The first purpose of the keyword search was to separate methodology and general environmental quality studies from studies that contain species valuation; the second purpose was to separate

studies by specific species; and the third purpose was to separate studies by water body type. The results of the keyword search were then used to categorize studies according to each ecological risk model for which they may be relevant. The following keywords were used to search the BenLit database:

- Fish; Aquatic; Bird; Freshwater; Marine; Wildlife; Salmon; Lake; Great Lake; Ocean; Water; River; Bass; Game; Chinook; Atlantic; Trout; Steelhead; Saltwater; Quality; Wetland; Estuary; Salt; and Endanger.

EVRI:
From the EVRI database, 979 studies were identified as potentially relevant. We did not include aquatic resources in our search of the EVRI database, as references for aquatic resources from EVRI were included in our initial compilation of the BenLit database. Keywords were selected from a predetermined list of categories presented within EVRI. Specific keywords were chosen from the list in order to separate studies by species and ecosystem. The keywords used to search the EVRI database included:

- Bird; Endangered Species; Invertebrate; Mammal; Heather; Crops; Rainforest; Riparian; Trees; Woodland; Freshwater; Canal; Drinking Water; Estuaries; Ground Water; Saltwater; Soil; Surface Reclamation; Wetlands; Agricultural Land; Open Spaces; Landscape; Beach.

The final list of studies collected for each model can be found in Appendix D. We included all potentially relevant studies found in the search. The number of studies that are potentially relevant to each ecological risk approach can be found in Table 3-1.

Table 3-1. Number of Economic Valuation Studies Potentially Relevant to Selected Ecological Risk Assessment Models

Model/Approach	Modeled Ecological Endpoints	Total Number of Articles Found
AQUATOX	Populations of aquatic species	173
RAMAS Ecotox	Populations of any species	283
RAMAS Metapop/ RAMAS GIS	Populations of any species	283
Level II OPP models	Populations of species in agricultural areas	19
Matrix Models	Populations of any species	283
Ecosystem Diagnosis and Treatment	Habitat value for select fish species	35
Salmon Population Modeling	Populations of salmon species	25
PATCH	Populations of terrestrial territorial vertebrates	21
Integrating ecological risk assessment and economic analysis in watersheds	Aquatic community structure	168
NRDAM	Populations of aquatic species	225
NWPCAM	Water quality	42
Acid Deposition Impacts on Brook Trout Fishing	Populations of brook trout	1
Superfund Risk Assessments	Individual level endpoints, for any species	283

The table shows a range in the number of benefits references found relevant for each model. In general, models with a broader range of ecological endpoints have more references that could apply. For example, the RAMAS models, matrix models, and Superfund risk assessments have the greatest number of references, and they are the only techniques evaluated that could apply to any given species. However, this does not necessarily indicate that any study conducted with one of these models would easily yield economic benefits, as many of the economic benefits articles may actually involve a smaller set of species. More resource-specific models, such as PATCH, did not yield many references because they apply to a narrow range of species.

In addition to the studies presented on a per-model basis, the search also identified 1,223 additional studies that present a more broad based view of ecological quality. These studies may be used to supplement the studies potentially relevant to the selected ecological risk models or to develop alternative valuation approaches to ecological risk.

3.2. Case Study Selection

Using the criteria outlined in Section 2, case studies of ecological risk models were chosen that are readily available; are appropriate for use in regulatory programs; utilize feasible input parameters; and examine relevant endpoints with potential valuation links. Sections 4, 5, and 6 present results from the selected assessments, including a discussion of economic benefits.

Matrix Models
To represent the matrix model approach, the selected case study derives population-level measures of aquatic species in response to a chemical stressor. The case study utilizes a matrix model to predict bluegill sunfish population size and time to recover from freshwater contamination of the heavy metal, selenium (Crutchfield and Ferson, 2000). This case study may be useful for benefits assessment as it involves fish populations, which are likely to have associated values either from recreational use or existence values. An initial search of the economic references that apply to populations showed no valuation references that pinpointed bluegill sunfish– but there are many (54) that refer to values for recreational fishing and could be extrapolated for this fish species. Section 4 presents the benefits assessment for ecological change to bluegill population health as measured in Crutchfield and Ferson (2000).

AQUATOX
In the case study that serves as a sample application of the AQUATOX model, the authors evaluated the effects of a pesticide (dieldrin) on a population of largemouth bass in a freshwater reservoir (Mauriello and Park, 2002). Runoff from agricultural activities had led to the accumulation of pesticide residues, leading to a decline in the recreational fishery. This study has potential regulatory application as it evaluates population recovery under different scenarios of pesticide reduction, in terms of the probability of reduction of fish biomass. Species biomass represents size of the bass population, which may be assigned benefits directly in valuation literature. Of the 173 economic studies we determined could apply to general AQUATOX ecological endpoints, eight valuation references appear on bass fishing. There are additional references evaluating recreational fishing that could potentially be used to extrapolate benefits. Dieldrin concentration within fish tissue and in the reservoir was also simulated using the model, which could relate to benefits literature on water quality. Benefits assessment of AQUATOX results for the pesticide-contaminated reservoir are presented in Section 5.

Superfund Risk Assessments
In practice, ecological risk assessments conducted using Superfund risk assessment guidance often measure individual-level endpoints, such as hazard quotient, by comparison of exposure levels of a contaminant, with benchmark levels for concentration shown to cause adverse effects. Thus case studies of this approach are not likely to yield directly quantitative population-level endpoints. However, if a wide range of ecological resources is assessed, and if the quantified risk metrics (such as hazard quotient) are linked conceptually to ecologically important endpoints, there is more opportunity to derive values associated with those resources.

One of the ecological risk assessment studies identified in the literature review, the Portsmouth Navy Yard assessments (Johnston et al., 2002), exemplifies one approach to measuring ecosystem function in response to stressors. In this assessment, the assessors selected several ecosystem level assessment endpoints (in addition to population assessment endpoints) but use particular species or other indicators to evaluate these ecosystem level effects.

This report does not estimate economic values for this case study, but proposes the types of measures in this Superfund study that could potentially be linked either qualitatively or quantitatively with economic values (presented in Section 6). For example, economic literature that values the health of estuarine communities or their existence value could be used, if an overall assessment of community health can be depicted adequately from the exposure and effects measures used; e.g., this study evaluates measures for salt marsh plant species as an indicator of salt marsh health, and one economic benefits study was initially found examining the value of saltwater marsh. Also, any of the particular species that were assessed in the study could be linked to economic values assigned to those species; for example, mussels and winter flounder may have associated catch values. The ecological risk assessment also evaluates several measures of water quality, for example, contaminant concentrations in surface water, and many economic studies relate to water quality in general or estuarine water quality.

4. Economic Valuation of Benefits: Case Study Using Matrix Model-based Ecological Risk Assessment for Bluegill Sunfish

The Crutchfield and Ferson (2000) case study uses a matrix model approach to model changes to the population health of the freshwater bluegill sunfish. This study was selected for illustration of the combined ecological and economic assessment for the following reasons:

- the input data requirements are feasible for use in a regulatory context, and
- the ecological assessment endpoint—change in population of a recreationally valuable species—can be linked to economic valuation endpoints from the existing resource valuation literature.

Bluegill population density is derived as the ecological endpoint in the study, and this was linked to economic benefits by deriving willingness-to-pay (WTP) values for related endpoints that have previously been valued (i.e., changes in recreational catch of bluegill sunfish, elimination of fish consumption advisory, and changes in bluegill populations). The total value of improving fish population health was then estimated by multiplying the WTP values for each relevant ecological assessment endpoint by the corresponding changes in that endpoint, and then summing the resulting values. Both the ecological risk framework and the benefits framework can be generally applied to other risk assessment situations, where life history parameters and WTP are available for the species of concern.

4.1. Ecological Assessment Case Study Details

The chosen case study, Crutchfield and Ferson (2000), models a population of bluegill sunfish (*Lepomis macrochirus*) in North Carolina in the years following the completion of a successful selenium mitigation effort. The study site is the Hyco Reservoir, which in addition to supporting an active recreational bluegill fishery, provides cooling water and receiving waters for coal-ash pond effluents from a nearby coal electric plant owned by Carolina Power and Light. When the power plant first began operations in 1966, the wet fly ash system discharged directly to the reservoir. However, a decline became evident in the bluegill sunfish and other sport fish of the reservoir, and studies in the late 1970's indicated bioaccumulation of selenium in the lake's food chain. In 1985, North Carolina's Department of Water Quality reduced the standard for selenium and established a new selenium NPDES permit limitation to protect the reservoir's water quality. In response to the new limitation, Carolina Power and Light began changing the plant to a dry fly ash system to reduce selenium concentrations in the effluent. The conversion was completed by 1990, resulting in a large reduction in selenium input to the reservoir starting in that year. Because continued monitoring was required by the Department of Water Quality to assess the effectiveness of the new limits, Crutchfield and Ferson were able to utilize data collected by the power company to model population changes for a 10-year period.

Coal-fired electric plants have released selenium into many large reservoirs, causing significant damage to numerous fish communities. A similar ecological risk scenario was seen in Belews Lake in North Carolina, which also accumulated selenium from a coal-fired electric generator in the late 1970's and 80's. At this site, selenium caused declines in bluegill and other species, and selenium concentrations were still high enough ten years following mitigation that young bluegills were still experiencing stress. Aside from discharge from power plants, selenium can also enter freshwater bodies through irrigation drainwater, weathering of sedimentary rock and soils containing selenium, and from releases from metallic ore mining and smelting.

Bluegill sunfish, the focus of this study, are popular among anglers. They have wide distribution throughout the United States, and are found in a variety of habitat types including ponds, lakes and slow-moving streams. Bluegill feed mostly on aquatic insects and other small invertebrates. Bluegill tissue concentrates selenium, and before completion of the Hyco Reservoir mitigation, concentrations in bluegill tissues were 2.3 to 20 times greater than toxic thresholds of 8 mg/g in muscle and 12 mg/g in liver tissue. As in many other risk assessments for recreational species, the scale of the study is local. The reservoir has a total surface area of over 1,700 hectares, but for the purpose of deriving numbers of bluegill individuals (as output is given in density rather than abundance), we assumed the size of the contaminated area of the reservoir to be approximately 1,000 hectares, based on the map provided in the published study.

Crutchfield and Ferson created a Leslie matrix model using bluegill census data collected by Carolina Power and Light as part of their annual biological monitoring. Data input for the matrix model included annual abundance of bluegill by size class (representative of age class) for 1979 to 1986, and average egg production per female by age/size class for an unaffected bluegill population at a reference point in the reservoir, upstream from the source of contamination. Initial abundance and age structure in the matrix model were provided by census data on abundance by size/age class for an affected bluegill population in 1986. A 10-year simulation was run for this affected population, including Ricker density dependence, and Monte Carlo methods to include natural environmental variability.

4.2. Ecological Benefits Assessed

The ecological matrix model provides only one output measure: predicted bluegill population density in the ten years following the elimination of selenium from Hyco Reservoir. Bluegill population density was predicted to increase by about 650 individuals/hectare in the 10-year period, from approximately 50 individuals/hectare to 700 individuals/hectare. However, the increase in population size over 10 years is not predicted to be linear. Because the initial population is weighted heavily towards adult age and size classes, the model predicts a peak of 2,000 individuals/hectare by year 3. This is followed by a one-year decline, an increase to a peak of about 1,000 in year 7, another decline, and another increase to about 700 in year 10. Through the oscillations, the average population is about 500. Figure 4-1 below shows the modeled results for bluegill population density.

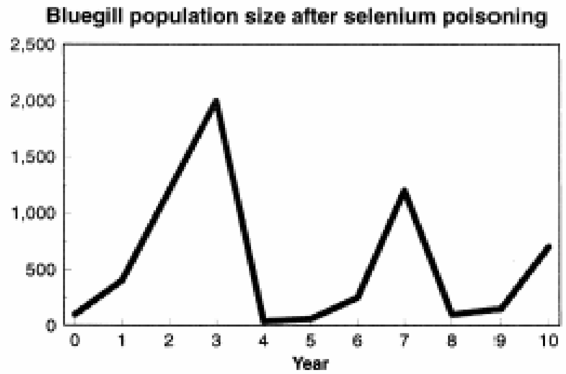

Figure 4-1. Predicted Recovery of Bluegill, from Crutchfield and Ferson, 2000.

Actual bluegill population density was also observed for the impacted population for a similar time period. The impacted population actually rose to about 6,000 individuals/hectare by 1994 (four years after mitigation effort). As the model predicted, the population declined the following year, and then began increasing again between year 5 and 6, to about 2,000 in 1996.

4.3. Economic Benefits Assessment

The study provides two measures of ecological health:

- predicted fish population density, and
- selenium concentrations in fish tissue (which is technically a model input, not an output).

The two measures represent different aspects of one assessment endpoint: bluegill population health. Changes in this ecological endpoint can be linked to a number of quantifiable economic benefits, including recreational use and non-use benefits. Recreational anglers benefit from improvements in bluegill population health because they catch more fish, and because they can eat the fish they catch. Additionally, residents of the Hyco Reservoir area may benefit from the knowledge that local bluegill populations are healthy, even if they do not catch or eat fish.

The following sections demonstrate how the measures of bluegill population health can be used as inputs to economic valuation models to quantify these welfare changes. Using data taken from Crutchfield and Ferson (2000), we estimate various economics benefits of increased bluegill population health resulting from reduction of selenium concentrations in the Hyco Reservoir.

4.3.1 Recreational Benefits of Increased Angling Success

The valuation of the recreational benefits of increases in bluegill population health attributable to the reduction of selenium concentrations took the following steps:

1) predicted change in bluegill population density to estimate the number of additional fish that would be caught by recreational anglers in the Hyco Reservoir study area;
2) estimated how much Hyco anglers would be willing to pay to catch additional bluegill sunfish, based on the results of a recent meta-analysis of recreational fishing values; and

3) estimated the recreational fishing benefits of a change in bluegill population density by multiplying the number of additional fish predicted to be caught by the calculated WTP per fish.

Step 1: Estimate the number of additional fish that would be caught by recreational anglers in the Hyco Reservoir study area.

Because the measure of fish population health provided in Crutchfield and Ferson (2000) is bluegill population density, the analysis began by linking this measure to total bluegill recreational catch. Based on creel census data from Lake Wateree (South Carolina) and Lake Hickory (North Carolina), total annual recreational catch of bluegill at the Hyco Reservoir study area was estimated at roughly 40 fish per hectare under baseline (i.e., pre-selenium poisoning) conditions. Since it is assumed that the study site is approximately 1,000 hectares in area, the total annual bluegill catch at the site is 39,789 fish. Appendix E.1 provides additional detail for these catch rate calculations.

The total bluegill catch at the reference site in each year following the elimination of selenium poisoning was estimated using the estimated total annual bluegill catch under the baseline scenario and assuming that changes in population density result in proportional changes in recreational catch. Table 4-1 shows the results of this calculation.

Table 4-1: Bluegill Population Density and Recreational Bluegill Catch

Year	Bluegill Population Density (fish/hectare)	Population Density Compared to Reference Site[a]	Recreational Bluegill Catch (fish/year)[b]	Increase in Catch Compared to Year 0 (fish/year)
0	50	3%	995	0
1	400	20%	7,960	6,965
2	1,300	65%	25,869	24,874
3	2,000	100%	39,798	38,803
4	25	1%	497	-497
5	50	3%	995	0
6	250	13%	4,975	3,980
7	1,200	60%	23,879	22,884
8	100	5%	1,990	995
9	150	8%	2,985	1,990
10	700	35%	13,929	12,934

[a] Compared to a baseline population density at the reference site of 2,000 bluegill/hectare in year 0.
[b] Assuming a baseline total catch rate of 39,789 bluegill/year (40 bluegill/hectare/year) at the reference site.

Step 2: Estimate how much Hyco anglers are willing to pay to catch additional bluegill sunfish.

The value of the increase in recreational catch was estimated using the results of a recent meta-analysis of recreational fishing values conducted to support the benefits analysis for the proposed section 316(b) rule for Phase III facilities (U.S. EPA, 2004d). The meta-analysis estimates the relationship between WTP to catch an additional fish, and resource, demographic, and study methodological characteristics. Table 4-2 presents the meta-analysis variables, the regression coefficients, and the input values assigned to each variable. The following bullets explain how these values were assigned:

- The study methodology variables were set to values that reflect a nested RUM study conducted in the year 2000 with a high resulting response rate.
- *Age* and *gender* were set to values that reflect the average values in the survey dataset. *Income* was set to $56,712, the median household income for North Carolina in 2003 (U.S. Census Bureau, 2005a).
- The species dummy variables were all set to zero, reflecting the default value (panfish).
- *Cr_nonyear* was set to 4.64 fish per day. We estimated this catch rate based on the average of the catch rates at Lake Wateree and Lake Hickory (see Appendix E.1 for more details).
- *Trips* and *shore* were set to values that reflect the average number of trips, and percentage of shore anglers, for non-Great Lakes freshwater anglers (U.S. FWS, 2002).

Based on an average catch rate of 1.16 bluegill per hour, WTP per additional bluegill is 94.9 cents. For further documentation for the meta-analysis, refer to the Regional Benefits Assessment for the Proposed Section 316(b) Rule for Phase III Facilities (U.S. EPA, 2004d).

Table 4-2: Recreational Meta-analysis Regression and Predicted WTP per Fish (2003$)

Variable	Coefficient	Input Value
Intercept	-2.9751	1
SP_conjoint	-0.2755	0
SP_dichot	0.07965	0
TC_individual	2.2848	0
TC_zonal	3.27	0
RUM_nest	2.2061	1
RUM_nonnest	2.7158	0
sp_year	0.1474	0
tc_year	-0.03301	0
RUM_year	-0.00844	24
sp_mail	-0.02076	0
high_resp_rate	-0.6542	1
inc_thou	0.02032	56.712
gender	-0.08744	89.11
spec_gender	7.4801	1
age	-0.06713	43.51
spec_age	3.2152	1
trips	-0.02307	13
spec_trips	0.7151	1
nonlocal	3.505	0
big_game_natl	1.7843	0
big_game_satl	2.7266	0
big_game_pac	2.7002	0
small_game_atl	1.6177	0
small_game_pac	2.0459	0
flatfish_atl	1.6407	0
flatfish_pac	2.2373	0
other_sw	1.0323	0
musky	3.6485	0
pike_walleye	1.379	0
bass_fw	1.6356	0
trout_rainbow	0.6093	0
trout_atlantic	1.1187	0
trout_GL	1.9356	0
trout_mountain	1.0592	0

Variable	Coefficient	Input Value
trout_pacific	0.663	0
trout_other	-0.7536	0
salmon_atlantic	5.774	0
salmon_GL	2.2719	0
salmon_pacific	2.9182	0
steelhead	3.1772	0
cr_nonyear	-0.0735	4.64
cr_year	-0.03335	0
spec_cr	0.4949	1
shore	-0.2291	0.57
WTP per Additional Fish (2003$)		**$0.949**

Step 3: Estimating the recreational fishing benefits of a change in bluegill population density

The last step in the analysis is to combine the estimates of recreational catch with the estimated WTP per additional fish. Multiplying these two values together yields the change in recreational welfare in each year following cessation of selenium poisoning, compared to year 0. Table 4-3 presents the results of this analysis. As shown in the table, there is a large degree of variability in total bluegill catch, and so the monetized benefits also show significant variations from year to year. In the third year after elimination of selenium poisoning, undiscounted recreational benefits are $36,829. However, because bluegill populations collapse in year 4, recreational benefits show a undiscounted net loss of $472 that year. In total, the cumulative undiscounted welfare gain from eliminating selenium from the reservoir is $107,183—equivalent to an average undiscounted yearly benefit of $10,718.

Table 4-3: Recreational Benefits of Elimination of Selenium Poisoning, by Year

Year	Recreational Bluegill Catch (fish/year)[b]	Increase in Catch Compared to Year 0 (fish/year)	Undiscounted Value of Increase in Catch (2003$)	Discounted Value of Recreational Catch[a] (2003$)
0	995	0	$0	$0
1	7,960	6,965	$6,610	$6,178
2	25,869	24,874	$23,609	$20,621
3	39,798	38,803	$36,829	$30,064
4	497	-497	-$472	-$360
5	995	0	$0	$0
6	4,975	3,980	$3,777	$2,517
7	23,879	22,884	$21,720	$13,526
8	1,990	995	$944	$550
9	2,985	1,990	$1,889	$1,027
10	13,929	12,934	$12,276	$6,241
Total	**n/a**	**112,928**	**$107,183**	**$80,363**
[a] Discounted to year 0 using a 7% discount rate.				

4.3.2. Recreational Benefits of Elimination of Fish Consumption Advisories

In addition to benefiting from increased catch rates, recreational anglers at Hyco Reservoir benefit from the elimination of fish consumption advisories. In their paper, Crutchfield and Ferson mention that prior to the selenium mitigation effort, selenium concentrations in bluegill muscle and liver tissue were 2.3 to 20 times greater than the threshold for detrimental toxic effects in fish. Although the authors do not discuss the effect of the mitigation on tissue concentrations, we assume that after the selenium mitigation, tissue concentrations return to their pre-contamination levels and that the fish consumption advisory on the reservoir would be lifted.

To estimate the benefits to recreational anglers of lifting the fish consumption advisory, we use a benefit transfer approach based on data from an original valuation study conducted by Jakus et al. (1997). The authors of this study used a repeated discrete choice travel cost model to examine the impacts of fish consumption advisories for PCBs, mercury, chlordane, and dioxin, in eastern and middle Tennessee. They estimate that the per-trip welfare gain from removing FCAs is:

- $1.59 ($1.82 in 2003$) to remove FCAs at one reservoir in eastern Tennessee,
- $1.85 ($2.12 in 2003$) to remove FCAs at two reservoirs in middle Tennessee, and
- $2.86 ($3.28 in 2003$) to remove FCAs at six reservoirs in eastern Tennessee.

Because this case study involves lifting a FCA at only one reservoir (Hyco Reservoir), it uses the value estimated for one reservoir in eastern Tennessee ($1.82).

The resource change valued in Jakus et al. (1997) is a relatively close match to the selenium mitigation at the Hyco Reservoir. However, unadjusted benefit transfer using a single study involves a considerable amount of uncertainty. Thus, a sensitivity analysis included estimates from another valuation study by Montgomery and Needleman (1997), even though it did not match the policy context as well.[1] Montgomery and Needleman estimated benefits from removing toxic contamination from lakes and ponds in New York State using a random utility model framework. They used a binary variable to indicate whether the New York Department of Environmental Conservation considers water quality in a given lake to be impaired by toxic pollutants. By controlling for other major causes of impairments that affect the fishing experience, they calculate the benefits of eliminating toxic contamination at all lakes to be $0.45 per day ($0.67 in 2003$).

Table 4-4 summarizes the values from these two studies.

Table 4-4: Recreational Fishing Benefits from Eliminating Fish Consumption Advisories

Source	Per Trip Welfare Gain (nominal $)	Per Trip Welfare Gain (2003$)	Description of Estimate
Jakus et al. (1997)	$1.59	$1.82	Recreational welfare gain from removing FCAs at one reservoir in eastern Tennessee, per trip
Montgomery and Needelman (1997)	$0.45	$0.67	Recreational welfare gain from eliminating toxic contamination at all lakes in NY, per trip

The values in Table 4-4 are per-trip values. To estimate the benefits of eliminating the fish consumption advisory at Hyco Reservoir using these values, we first estimated the number of

[1] We identified two other studies that estimated the welfare gains that result from eliminating fish consumption advisories: Phaneuf et al. (1998) and Breffle et al. (1999). However, because the resource characteristics of these studies are fairly different from the Hyco Reservoir situation, we did not conduct benefit transfers using values from these studies.

trips taken to the reservoir.[2] Using data from Lake Wateree and Lake Hickory, we estimate that average annual angling pressure at Hyco Reservoir is 151.8 hours per hectare. Assuming that the affected portion of the reservoir is 1,000 hectares, anglers spend 151,800 hours fishing at the study site each year. If the average fishing trip is four hours long, then anglers make 37,950 trips to the reservoir each year. We assume that all trips are one day in length.

Table 4-5 shows a range of estimates of the value of lifting the fish consumption advisory at Hyco Reservoir. Our primary estimate, based on Jakus et al. (1997), is a total welfare gain to recreational anglers of $69,149 per year, and our alternative estimate, based on Montgomery and Needelman (1997), is $25,347 per year. The difference between these two estimates reflects the uncertainty associated with unadjusted single-site benefit transfer.

Table 4-5: Recreational Fishing Benefits from Eliminating Fish Consumption Advisories

Number of Trips per Year	Welfare Gain per Trip (2003$)	Total Benefits (2003$)	Source
37,950	$1.82	$69,149	Jakus et al. (1997)
37,950	$0.67	$25,347	Montgomery and Needelman (1997)

As with any unadjusted benefit transfer, there are several limitations to this analysis. Most importantly, the changes evaluated in the two studies are not a perfect match for the Hyco Reservoir policy context. Specific demographic and resource differences between Hyco Reservoir and the study sites might lead to differences in the value of eliminating fish consumption advisories. Additionally, magnitude of the estimated welfare gain depends on the estimate of the number of trips taken to the lake, which is at best an approximation, and does not take into account the fact that angler participation at Hyco Reservoir may have been reduced during the period of selenium contamination.

4.3.3 Non-use Benefits of Improvement in Fish Population Health

As noted above, the selenium mitigation effort at the Hyco Reservoir may benefit local residents, even if they do not engage in recreational activities at the reservoir. The *non-use* benefits that individuals receive include the satisfaction of knowing that local fish populations are healthy and will be available for future generations. The evaluation of these benefits included three steps. estimating the number of households that might reasonably hold non-use values for bluegill population health at the Hyco Reservoir; using an aquatic resource meta-analysis benefits model to predict how much those households would be willing to pay to eliminate selenium contamination at the reservoir; and finally, multiplying the number of affected households by average WTP for the selenium mitigation.

This case study assumed that the population likely to hold non-use values for the Hyco Reservoir bluegill population would include all households within 25 miles of the reservoir. Using U.S. Census Bureau data, this population was estimated to include 85,624 households.[3]

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[2] This calculation is discussed in more detail in Appendix E.2.
[3] This calculation is documented in Appendix E.3.

To estimate how much each household would be willing to pay for the elimination of selenium contamination at the reservoir, a meta-analysis model developed during EPA's analysis of the benefits of the final section 316(b) rule for Phase II facilities (Johnston et al., 2005) was used. This model analyzes the relationship between resource, demographic, and methodological characteristics and total annual WTP for improvements to surface water quality and aquatic habitat. The meta-analysis expresses improvements in water and habitat quality using the Resources for the Future (RFF) water quality ladder. This water quality ladder is a ten-point scale linked to specific pollutant levels which, in turn, are linked to presence of aquatic species and recreational uses. Thus, a WQL of 10 indicates drinkable water, a WQL of 7 indicates that a water body is safe for swimming, and a WQL of 5 indicates that a water body supports game fish.

In order to use the meta-analysis model to estimate household non-use values for the case study, the changes resulting from the selenium mitigation were mapped to the RFF water quality ladder. Before the mitigation, selenium concentrations severely inhibit bluegill survival. Thus, it was assumed that the pre-mitigation WQL was 4.75—slightly below the value necessary to support game fish. Although Crutchfield and Ferson do not provide specific data that can be used to assess the level of selenium in Hyco Reservoir following the mitigation effort, they do state that the "mitigation would remove the adverse consequences of selenium essentially immediately." This case study assumed that the selenium mitigation effort reduces selenium concentrations to the baseline concentration (the level before the reservoir was contaminated), and that this change is equivalent to a one point increase on the RFF water quality ladder. Thus, the final WQL at the reservoir was set toto 5.75, a level of water quality that easily supports game fish.

Table 4-6 presents the meta-analysis variables and coefficients, as well as the values assigned to each variable. For detailed information about the variable definitions, refer to U.S. EPA (2004). The following bullets briefly explain how we assigned values to each of the variables:

- Selected survey methodological variables (*interview*, *year_indx*, *nonparam*, *discrete_ch*, *protest_bids*, *outlier_bids*) were assigned values that reflect a discrete choice survey format conducted through in-person interviews, with protest bids and outlier bids eliminated. Other survey methodological variables were set to zero.
- *Income* was set to the median household income for North Carolina in 2003 (U.S. Census Bureau, 2005a).
- The regional dummies were all set to zero, based on the judgment that the *southeast* variable is capturing study-level effects, not regional influences.
- Two of the resource description variables (*single_lake* and *num_riv_pond*) were set to values reflecting a change occurring on one reservoir. The remaining resource variables were set to zero.
- The water quality variables *WQ_fish* and *baseline* were assigned values that reflect a one point increase on the water quality ladder, from 4.75 to 5.75. The remainder of the WQ variables were set to zero, since the selenium mitigation primarily affects fish.
- The variable *non-users* was set to one so that the model would predict total WTP for non-users only—which by definition includes only non-use values. This value is assumed to be a lower bound for the non-use value for users of the resource, who, being more

familiar with the resource, might be expected to hold greater non-use values for it (independently of any use values they also hold).

Based on the variable assignments shown in the table, non-user WTP for the specified resource change is $10.90 per household. As a sensitivity analysis, this case study also calculated WTP based on a mail survey format (instead of an interview format). Under this alternative assumption, non-user WTP is $2.90—an eight dollar difference.

Table 4-6: Estimating Non-use WTP for Improvement in Water Quality at Hyco Reservoir

Variable	Coefficient	Assigned Value
Intercept	6.0158	1
year_indx	-0.1072	31
discrete_ch	0.3956	1
Voluntary	-1.633	0
Nonparam	-0.4472	1
Income	0.00000058	56712
wq_ladder	-0.3799	1
protest_bids	0.9537	1
outlier_bids	-0.8764	1
hi_response	-0.8094	1
single_river	-0.3378	0
single_lake	0.3193	1
multiple_river	-1.605	0
salt_ponds	0.7574	0
num_riv_pond	0.0791	1
regional_fresh	-0.0073	0
Southeast	1.1482	0
Plains	-0.8153	0
pacif_mount	-0.3125	0
multi_reg	0.5951	0
Nonusers	-0.5017	1
WQ_fish	0.2055	1
WQ_shell	0.2561	0
WQ_many	0.2332	0
WQ_non	0.4695	0
Fishplus	0.8052	0
Baseline	-0.1265	4.75
Interview	1.3252	1
Mail	0.5666	0
lump_sum	0.5954	0
nonfish_uses	-0.1412	0
median_WTP	0.2206	0
		WTP per Household (2003$)
Based on interview survey format (*interview*=1)		$10.90
Based on mail survey format (*mail*=1)		$2.90

The final step in the calculation of non-use value for the selenium mitigation at Hyco Reservoir was to combine the estimate of the affected population with our estimates of household WTP. Table 4-7 shows the results of these calculations. Based on a total affected population of 85,624 local households, it was estimated that the total annual non-use benefits of the selenium mitigation at the Hyco Reservoir could range from $933,075 to $247,965. Note that these estimates include the value of *all* ecosystem services that are improved by the selenium mitigation. WTP for bluegill population health is a subset of this total non-use value. Thus, this value should be considered an upper limit on the value of the improvement in bluegill populations; the actual value is likely to be much lower.

Table 4-7: Non-use Benefits of the Selenium Mitigation at Hyco Reservoir

Number of Affected Households	Non-use Value per Household	Total Non-use Benefits
85,624	$10.90	$933,075
85,624	$2.90	$247,965

There are several significant limitations and uncertainties associated with using the meta-analysis model to estimate non-use values for individuals affected by the Hyco Reservoir case study. First, as in any benefit transfer, the policy site does not match perfectly to the studies on which the meta-analysis was based, and the meta-model does not represent perfectly all characteristics of the Hyco River selenium mitigation. For example, elimination of selenium at the reservoir does not map perfectly to the water quality ladder. Additionally, important characteristics of Hyco Reservoir, such as its reputation as a good bluegill fishery, are not included in the meta-model. One additional limitation of this analysis is that it does not consider temporal fluctuations in bluegill populations. Instead, the case study assumes that the non-use benefits of the selenium mitigation are independent of the fluctuations in bluegill abundance.

4.3.4 Total Benefits of Improvement in Fish Population Health

The final step in the evaluation of the benefits of the Hyco Reservoir bluegill recovery was to calculate total social welfare gain. Comprehensive, appropriate estimates of total resource value include both use and non-use values, such that the resulting total value estimates may be compared to total social cost. The total benefits are the sum of the recreational and non-use values calculated in the previous sections. Table 4-8 presents the results of this calculation. The table shows that the total value of the bluegill population recovery at Hyco Reservoir could range from $284,030 to $1.01 million.

Table 4-8: Total Social Benefits of Restoring Bluegill Populations at Hyco Reservoir (2003$)

Type of Value	Low Estimate	Middle Estimate	High Estimate
Recreational Value from Increased Catch Rates	$10,718	$10,718	$10,718
Recreational Value from Eliminating FCAs	$25,347	$47,248	$69,149
Non-use Value of Selenium Mitigation[a]	$247,965	$590,520[b]	$933,075
Total Social Value[a]	**$284,030**	**$648,486**	**$1,012,942**

[a] The non-use value of restoring bluegill populations is a subset of the non-use value of the selenium mitigation. Thus, in this sense, all non-use estimates (and the total social value estimates) should be considered upper bounds.
[b] This value is the mid-point of the two estimates of non-use value from Section 4.3.3.

Because of the large number of assumptions necessary to estimate each of the component values presented in Table 4-8, the total social value estimates should be interpreted with caution. Availability of better data, particularly data on how changes in the ecological endpoints described in the Crutchfield and Ferson study translate into changes in angler participation and catch rates, would significantly improve the accuracy of the recreational benefits estimates. Also, the non-use benefits estimates could be significantly improved by removing the portion of the non-use estimates attributable to habitat improvements that are unrelated to fish.

4.4. Extension of Ecological and Economic Assessment to Other Case Studies

Input data requirements and the methodology for this case study can be applied to various ecological risk scenarios. The model used to predict change in population health, a Leslie matrix model, can be constructed for any number of species when there is basic life history data on survival and reproduction. In the study by Crutchfield and Ferson (2000), the power company discharging selenium collected population data on bluegill. However, since bluegill are a popular recreational species and are abundant in North American fresh water bodies, census and life history data for this species at other sites would have a good probability of being collected by wildlife managers, if not by the discharger as in this case study.

The valuation methodology for estimating the value of changes in recreational catch can by applied to a variety of situations. Output datasets from matrix models are excellent candidates for this type of valuation—as are any ecological models that provide assessment endpoint measures that can be used to estimate changes in recreational catch rates. The recreational meta-model includes variables for a variety of species and resource characteristics, making it a flexible tool for valuing a range of fish-related ecological model outputs.

The method used to value the elimination of fish consumption advisories at Hyco Reservoir is in theory broadly applicable, but in practice is heavily dependent on the presence of existing studies that value similar scenarios. In this case study, the policy change valued by Jakus et al. (1997) provided a very close match to the scenario at Hyco Reservoir. However, the literature on the value of eliminating fish consumption advisories is limited, making it hard to generalize the methodology used in this case study.

Like the recreational meta-analysis, the non-use meta-analysis can be potentially used to estimate non-use values for a wide range of resource changes. The primary requirement for applying this methodology to the output of an ecological model is that the assessment endpoint measures must be mapped to the Resources for the Future water quality ladder. Whether this is possible will depend on the kinds of ecological changes being considered. The model can actually be used to value changes that are ecologically very complex, as long as those changes result in a general improvement in aquatic resource health. It is more difficult to apply the model to situations where small changes occur to specific resources, or where the resources change in a way unlike the changes considered in the studies underlying the meta-analysis—for example, encroachment of native vegetation by an exotic plant species.

5. Economic Valuation of Benefits: Case Study Using AQUATOX-based Ecological Risk Assessment

Mauriello and Park (2002) present a retrospective study of the effects of the pesticide dieldrin to largemouth bass population health in the Coralville Reservoir, Iowa. Different pesticide reduction scenarios as well as the viability of the bass population under each scenario are examined. In addition, they used AQUATOX to evaluate population effects to other fish species present, including a commercial buffalofish population, walleye, shad, and bluegill sunfish (unpublished data, pers. comm. Park, 2005). Outputs include total dieldrin concentration, the dieldrin concentration in the bass, probability of reduced population biomass, and population density of all species evaluated between 1969 and 1986. This study was selected because the ecological model is freely available and can be applied to numerous risk scenarios. As this case study has population-level endpoints for recreationally and commercially valuable species, it also lends itself to valuation of commercial, recreational, and non-use benefits. In addition, the ecological risk framework is broadly applicable because the study measures ecological change under alternate contaminant regimes, which is useful for decision makers.

5.1. Ecological Assessment Case Study Details

In this case study, Mauriello and Park evaluate ecological change from reduced pollutant input in the Coralville Reservoir, where runoff from agricultural activities had led to the accumulation of aldrin and dieldrin. The reservoir was impounded in 1958 by the construction of the Coralville dam on the Iowa River, for both flood control and recreational use. By the 1970's a decline was seen in the population of largemouth bass and other recreational fish species, and tissue concentrations in fish of aldrin and dieldrin increased. This bioaccumulation resulted in a ban on commercial fishing in the reservoir.

The Iowa River watershed saw extensive use of similar chemicals in the 1960's and 70's. As aldrin and dieldrin were widely used on corn, cotton, and citrus crops until EPA discontinued its use in 1974, these pesticides probably accumulated in a number of water bodies in agricultural areas. Elevated levels of dieldrin have resulted in fishing bans in other lakes, such as Fosdic Lake and Lake Como in Texas (Kelly, 1999). Aldrin and dieldrin were also used to control termites, and this use of was banned later, in 1987. Aldrin is considered a persistent bioaccumulative toxin, however, it breaks down quickly into dieldrin in the environment and in the body, so they are generally considered together. They accumulate in water, sediment, aquatic organisms, and also within root crops, and create persistent contamination because dieldrin breaks down slowly in soil, water, and body tissues. In addition to ecological concerns of dieldrin, it is also of concern for human exposure, and was the second most common pesticide detected in a survey of milk in the U.S. (World Bank Group). EPA has determined that aldrin and dieldrin are possible human carcinogens.

While this risk scenario is applicable to a wide range of water bodies, Mauriello and Park evaluate ecological change on a local scale, using data for the Coralville Reservoir (surface area of about 8.4mi^2) to simulate population health and water quality under different reduction scenarios for dieldrin. The AQUATOX program was used to model ecological changes under the following alternatives: 1) there is no reduction in dieldrin input (100% dieldrin input), 2) a 50% reduction in dieldrin input, 3) a complete cessation of dieldrin input, and 4) gradual cessation of dieldrin input into Coralville Reservoir. (Note that only the fourth scenario models multiple fish species and these data were obtained directly from the authors; the first three scenarios only simulate largemouth bass and are included in the published study.) This evaluation of different amounts of contaminant reduction would be useful to risk managers and regulators, who often weigh alternate regulatory options and need to look at the resulting levels of ecological change (and the associated economic outcomes). Input data on pesticide and nutrient inputs, fish biomass at the starting point for each species, and water flow into the reservoir were taken from monitoring studies conducted for the U.S. Army Corps of Engineers, and other site data maintained by the Army Corps and by the U.S. Geological Survey. The model was run to simulate change between 1968 and 1986, which includes time before the actual reduction in dieldrin use in 1974.

5.2. Ecological Benefits Assessed

Model output data for the fourth scenario (gradual reduction in dieldrin input) include density of largemouth bass, bluegill sunfish, shad, buffalofish, and walleye, for ten-day intervals from 1969 through 1986. By 1981, dieldrin input had declined to 0. In the simulation, shad and walleye populations increased relative to 1969 levels, with shad increasing about eight times in density, and walleye doubling their density. Other species (bluegill, buffalofish, and largemouth bass) saw a decline compared to 1969, but showed varying levels of rebound during the simulation period. This case study uses the results of this simulation in the economic benefits assessment.

Published output for the first three dieldrin input scenarios include the probability of biomass reduction, percent difference of biomass from a control level, dieldrin concentration in bass tissue, and predicted largemouth bass density. Model results indicated that with 100% dieldrin input, there would be almost 100% chance of a 90% reduction in bass biomass. At a 50% pollutant reduction, there is almost 100% chance of an 80% reduction in bass biomass. With complete cessation of dieldrin input, the greatest modeled percent reduction seen is 20%, with almost 100% chance of only a 10% reduction in biomass. Another endpoint simulated by AQUATOX is the percent difference of biomass from a control scenario. This simulation indicated that both 100% and 50% levels of dieldrin input lead to 70 to 80% lower biomass below the control. When dieldrin is removed completely, the model predicts population recovery to control levels by 1977, and a 10% increase above the control by 1985. These endpoints for risk to the bass population are useful for illustrative purposes and comparison of alternative pollutant reduction options. However, it is difficult to value these changes if the control quantity for largemouth biomass is not known.

Dieldrin body burden was modeled for the 1968 to 1985 period, and only the complete cessation of dieldrin input resulted in a return to 0 ppb by 1977. At a 100% dieldrin input, concentration in bass tissue reached about 20 ppb by 1985, and at 50% dieldrin input, body burden was about 14 ppb by this same time. Dieldrin concentration in the reservoir, an indicator of water quality, was also measured for the same time period, assuming no further dieldrin input after 1975. While population recovery occurs in this situation, dieldrin remains in the reservoir at about half the peak value (at about 0.002 μg/L), until 1985.

5.3. Economic Benefits Assessment

The following sections present our estimates of the recreational, commercial, and non-use benefits of eliminating dieldrin contamination at Coralville Reservoir. Note that the benefits of eliminating dieldrin contamination at the reservoir were evaluated, not the net cost of the entire dieldrin contamination episode. Thus, the frame of reference for the following analysis is a scenario in which dieldrin is not eliminated from the reservoir. Since Mauriello and Park did not use their ecological model to predict fish populations under such a scenario, it was necessary to make a number of assumptions about how populations of each fish species might have been affected. These assumptions are documented in the following sections.

5.3.1 Recreational Benefits of Increased Angling Success

Based on the ecological data from Mauriello and Park, elimination of dieldrin contamination at Coralville Reservoir results in full or partial recovery of the populations of several recreationally valuable species, including largemouth bass, bluegill, and walleye. Estimating the effect of this increase in fish populations on the welfare of recreational anglers required the following steps:

4) predicting change in species population density to estimate the number of additional fish that would be caught by recreational anglers in the Coralville Reservoir study area;
5) estimating how much Coralville anglers would be willing to pay to catch additional fish, based on the results of a meta-analysis of recreational fishing values; and
6) estimating the recreational fishing benefits of the predicted change in species population density by multiplying the number of additional fish predicted to be caught by the calculated WTP per fish.

In addition to the three species mentioned above, Mauriello and Park provided information on the effect of eliminating dieldrin on shad and buffalofish populations. Gizzard shad, the type of shad typically found in Iowa, are a small forage fish with little recreational value.[4] A commercial fishery does exist for buffalofish, but this species is seldom targeted by recreational anglers (Iowa DNR, 1987). Thus, we excluded these species from our analysis of recreational angling benefits.

Step 1: Estimate the number of additional fish that would be caught by recreational anglers in the Coralville Reservoir study area.

Because the measure of fish population provided by Mauriello and Park is species population density, we began our analysis by linking this measure to total recreational catch. The baseline level of recreational catch for each species in the year before the dieldrin poisoning (1969) were estimated, using creel census data from similar water bodies in Indiana (West Boggs Creek Reservoir) and Illinois (Shabonna Lake). Based on these data, baseline total annual recreational

[4] Because gizzard shad sometimes crowd out more important recreational species such as bluegill, growth of gizzard shad populations may actually result in negative benefits to recreational anglers. (Iowa DNR. 1987. "Gizzard Shad." http://www.iowadnr.com/fish/iafish/gizzshad.html.)

catch at Coralville Reservoir was estimated to be roughly 8 largemouth bass per acre, 54 bluegill per acre, and 5.3 walleye per acre. Since the study site is 5,376 acres in area, baseline annual recreational catch for the entire reservoir is 43,239 largemouth bass, 290,304 bluegill, and 28,493 walleye. Appendix F.1 provides additional detail for these catch rate calculations and discusses limitations and uncertainties associated with these estimates.

The total annual catch at Coralville Reservoir during the period from 1970 to 1986 was calculated by assuming that changes in population density result in proportional changes in recreational catch. For example, the population density of largemouth bass in 1970 is only 44 percent of the population density in 1969, so it was assumed that recreational catch in 1970 was also 44 percent of the recreational catch in 1969.

To calculate the recreational fishing benefits of eliminating dieldrin contamination at Coralville Reservoir, this case study assumed that if dieldrin had not been eliminated, populations of bass, walleye, and bluegill would have been reduced to zero by 1978 and would have remained zero through 1986.[5] Thus, the improvement in recreational catch resulting from eliminating dieldrin is equal to the difference in recreational catch between the two scenarios. Tables 5-1, 5-2, and 5-3 show the results of this calculation for largemouth bass, bluegill, and walleye.

Table 5-1: Largemouth Bass Population Density and Recreational Catch

Year	Population Density Compared to Baseline[a]	Recreational Largemouth Bass Catch with Remediation (fish/year)[b]	Recreational Largemouth Bass Catch without Remediation (fish/year)[b]	Increase in Recreational Catch (fish/year)
1970	44%	19,175	19,175	0
1971	8%	3,398	3,398	0
1972	1%	463	463	0
1973	0%	45	45	0
1974	0%	39	39	0
1975	0%	42	42	0
1976	0%	36	36	0
1977	0%	23	23	0
1978	0%	50	0	50
1979	0%	177	0	177
1980	1%	537	0	537
1981	3%	1,485	0	1,485
1982	8%	3,338	0	3,338
1983	15%	6,577	0	6,577
1984	26%	11,046	0	11,046
1985	35%	15,045	0	15,045
1986	40%	17,462	0	17,462

[a] The baseline population density in 1969 was 18,157 fish per acre.
[b] Assuming a baseline total annual catch rate of 43,239 bass/year (8 bass/acre/year) in 1969.

[5] As discussed above, Mauriello and Park did not estimate fish populations under a scenario in which dieldrin is not eliminated from the reservoir. However, during the first few years of the dieldrin contamination when dieldrin concentrations were high, fish populations fell rapidly. We assumed that if dieldrin concentrations had remained at those levels, populations of bass, bluegill, and walleye would have eventually been reduced to zero. We somewhat arbitrarily assume that this would have occurred in 1978.

Table 5-2: Bluegill Population Density and Recreational Catch

Year	Population Density Compared to Baseline[a]	Recreational Bluegill Catch with Remediation (fish/year)[b]	Recreational Bluegill Catch without Remediation (fish/year)[b]	Increase in Recreational Catch (fish/year)
1970	0%	1,067	1,067	0
1971	0%	166	166	0
1972	0%	119	119	0
1973	0%	235	235	0
1974	0%	268	268	0
1975	0%	222	222	0
1976	0%	133	133	0
1977	0%	52	52	0
1978	0%	227	0	227
1979	0%	422	0	422
1980	0%	1,150	0	1,150
1981	1%	3,555	0	3,555
1982	3%	7,512	0	7,512
1983	5%	13,463	0	13,463
1984	7%	20,539	0	20,539
1985	8%	24,096	0	24,096
1986	10%	28,058	0	28,058

[a] The baseline population density in 1969 was 14,524 fish per acre.
[b] Assuming a baseline total annual catch rate of 290,304 bluegill/year (54 bluegill/acre/year) in 1969.

Table 5-3: Walleye Population Density and Recreational Catch

Year	Population Density Compared to Baseline[a]	Recreational Walleye Catch with Remediation (fish/year)[b]	Recreational Walleye Catch without Remediation (fish/year)[b]	Increase in Recreational Catch (fish/year)
1970	7%	1,888	1,888	0
1971	0%	18	18	0
1972	0%	13	13	0
1973	0%	21	21	0
1974	0%	26	26	0
1975	0%	23	23	0
1976	0%	15	15	0
1977	0%	7	7	0
1978	0%	95	0	95
1979	0%	132	0	132
1980	2%	496	0	496
1981	18%	5,170	0	5,170
1982	129%	36,726	0	36,726
1983	474%	135,163	0	135,163
1984	531%	151,408	0	151,408
1985	320%	91,042	0	91,042
1986	209%	59,626	0	59,626

[a] The baseline population density in 1969 was 1,199 fish per acre.
[b] Assuming a baseline total annual catch rate of 28,493 largemouth/year (5.3 bluegill/acre/year) in 1969.

Step 2: Estimate how much Coralville anglers are willing to pay to catch additional fish.

To estimate the value of the predicted increase in recreational catch, the analysis used the results of a recent meta-analysis of recreational fishing values conducted to support the benefits analysis for the proposed section 316(b) rule for Phase III facilities (U.S. EPA, 2004d). The meta-analysis estimates the relationship between WTP to catch an additional fish, and resource, demographic, and study methodological characteristics. Table 5-4 presents the meta-analysis variables, the regression coefficients, and the input values assigned to each variable. The following bullets explain how these values were assigned:

- The study methodology variables were set to values that reflect a nested RUM study conducted in the year 2000 with a high resulting response rate.
- *Age* and *gender* were set to values that reflect the average values in the survey dataset. *Income* was set to $41,827, the median household income for Iowa in 2002 (U.S. Census Bureau, 2005b).
- Selected species dummy variables (*bass_fw* and *pike_walleye*) were set equal to 1, depending on the species being considered. For bluegill, all the species dummy variables were set to 0, reflecting the default value, *panfish*.
- *Cr_nonyear* was set to .21 fish per day for bass, 4.7 fish per day for bluegill, and .82 fish per day for walleye, based on average U.S. catch rates for these species. Use of catch rates specific to Coralville Reservoir or Iowa would substantially improve the accuracy of the analysis (U.S. EPA, 2004d).
- *Trips* and *shore* were set to values that reflect the average number of trips, and percentage of shore anglers, for non-Great Lakes freshwater anglers (U.S. FWS, 2002).

Based on the variable assignments shown in Table 4, estimated WTP per additional fish caught is $4.99 for largemouth bass, $0.70 for bluegill, and $3.69 for walleye. For further documentation of the meta-analysis, refer to the Regional Benefits Assessment for the Proposed Section 316(b) Rule for Phase III Facilities (U.S. EPA, 2004d).

Table 5-4: Recreational Meta-analysis Regression and Predicted WTP per Fish (2003$)

Variable	Coefficient	Input Value		
		Largemouth Bass	**Bluegill**	**Walleye**
Intercept	-2.9751	1	1	1
SP_conjoint	-0.2755	0	0	0
SP_dichot	0.07965	0	0	0
TC_individual	2.2848	0	0	0
TC_zonal	3.27	0	0	0
RUM_nest	2.2061	1	1	1
RUM_nonnest	2.7158	0	0	0
sp_year	0.1474	0	0	0
tc_year	-0.03301	0	0	0
RUM_year	-0.00844	24	24	24
sp_mail	-0.02076	0	0	0
high_resp_rate	-0.6542	1	1	1
inc_thou	0.02032	41.827	41.827	41.827
gender	-0.08744	89.11	89.11	89.11
spec_gender	7.4801	1	1	1
age	-0.06713	43.51	43.51	43.51
spec_age	3.2152	1	1	1
trips	-0.02307	13	13	13
spec_trips	0.7151	1	1	1
nonlocal	3.505	0	0	0
big_game_natl	1.7843	0	0	0
big_game_satl	2.7266	0	0	0
big_game_pac	2.7002	0	0	0
small_game_atl	1.6177	0	0	0
small_game_pac	2.0459	0	0	0
flatfish_atl	1.6407	0	0	0
flatfish_pac	2.2373	0	0	0
other_sw	1.0323	0	0	0
musky	3.6485	0	0	0
pike_walleye	1.379	0	0	1
bass_fw	1.6356	1	0	0
trout_rainbow	0.6093	0	0	0
trout_atlantic	1.1187	0	0	0
trout_GL	1.9356	0	0	0
trout_mountain	1.0592	0	0	0
trout_pacific	0.663	0	0	0
trout_other	-0.7536	0	0	0
salmon_atlantic	5.774	0	0	0
salmon_GL	2.2719	0	0	0
salmon_pacific	2.9182	0	0	0
steelhead	3.1772	0	0	0
cr_nonyear	-0.0735	0.21	4.7	0.82
cr_year	-0.03335	0	0	0
spec_cr	0.4949	1	1	1
shore	-0.2291	0.57	0.57	0.57
WTP per Additional Fish (2003$)		**$4.99**	**$0.70**	**$3.69**

Step 3: Estimate the recreational fishing benefits of a change in population density

The last step in our analysis was to combine the estimates of the increases in recreational catch with the estimated WTP per additional fish. By multiplying these values together, we calculated the change in recreational welfare in each year following cessation of dieldrin poisoning, compared to a scenario in which there was no cessation of dieldrin inputs. Tables 5-5, 5-6, and 5-7 present the results of this analysis. The tables show that in 1986, the final year of the analysis, the undiscounted value of the annual increase in recreational catch from eliminating dieldrin contamination is $87,191 for largemouth bass, $19,593 for bluegill, and $219,902 for walleye. Including all three species, the cumulative undiscounted recreational welfare gain from eliminating dieldrin contamination is $2.12 million over the period from 1970 to 1986—equivalent to an average undiscounted yearly benefit of $124,535.

Table 5-5: Recreational Benefits to Largemouth Bass Anglers of Elimination of Dieldrin Poisoning, by Year

Year	Recreational Largemouth Bass Catch with Remediation (fish/year)	Increase in Catch Compared to Scenario without Remediation (fish/year)	Value of Increase in Catch (undiscounted; 2003$)
1970	19,175	0	$0.00
1971	3,398	0	$0.00
1972	463	0	$0.00
1973	45	0	$0.00
1974	39	0	$0.00
1975	42	0	$0.00
1976	36	0	$0.00
1977	23	0	$0.00
1978	50	50	$249.83
1979	177	177	$885.70
1980	537	537	$2,683.44
1981	1,485	1,485	$7,413.84
1982	3,338	3,338	$16,669.15
1983	6,577	6,577	$32,837.52
1984	11,046	11,046	$55,155.17
1985	15,045	15,045	$75,122.11
1986	17,462	17,462	$87,191.28
Total	**n/a**	**55,718**	**$278,208.03**

Table 5-6: Recreational Benefits to Bluegill Anglers of Elimination of Dieldrin Poisoning, by Year

Year	Recreational Bluegill Catch with Remediation (fish/year)	Increase in Catch Compared to Scenario without Remediation (fish/year)	Value of Increase in Catch (undiscounted; 2003$)
1970	1,067	0	$0.00
1971	166	0	$0.00
1972	119	0	$0.00
1973	235	0	$0.00
1974	268	0	$0.00
1975	222	0	$0.00
1976	133	0	$0.00
1977	52	0	$0.00
1978	227	227	$158.38
1979	422	422	$294.91
1980	1,150	1,150	$802.78
1981	3,555	3,555	$2,482.51
1982	7,512	7,512	$5,245.52
1983	13,463	13,463	$9,401.26
1984	20,539	20,539	$14,342.70
1985	24,096	24,096	$16,826.80
1986	28,058	28,058	$19,592.93
Total	**n/a**	**99,021**	**$69,147.79**

Table 5-7: Recreational Benefits to Walleye Anglers of Elimination of Dieldrin Poisoning, by Year

Year	Recreational Walleye Catch with Remediation (fish/year)	Increase in Catch Compared to Scenario without Remediation (fish/year)	Value of Increase in Catch (undiscounted; 2003$)
1970	1,888	0	$0.00
1971	18	0	$0.00
1972	13	0	$0.00
1973	21	0	$0.00
1974	26	0	$0.00
1975	23	0	$0.00
1976	15	0	$0.00
1977	7	0	$0.00
1978	95	95	$349.43
1979	132	132	$486.00
1980	496	496	$1,830.83
1981	5,170	5,170	$19,068.92
1982	36,726	36,726	$135,447.43
1983	135,163	135,163	$498,484.20
1984	151,408	151,408	$558,397.43
1985	91,042	91,042	$335,764.82
1986	59,626	59,626	$219,902.48
Total	**n/a**	**479,858**	**$1,769,731.55**

There are a number of limitations and uncertainties implicit in this analysis:

- As noted above, estimating catch rates at Coralville Reservoir based on catch rates for other water bodies introduces a significant degree of error into our calculations.
- The assumption that fish populations would fall to zero after 1977 in the uncontrolled scenario is not based on results from an ecological model, and thus may not accurately represent how populations would have actually behaved.
- As with any benefit transfer, use of the meta-analysis model introduces some error. The policy site does not match perfectly to the studies on which the meta-analysis was based, and the meta-model does not capture all characteristics of the recreational angling experience at Coralville Reservoir.

5.3.2 Commercial Benefits of Improvement in Fish Population Health

According to Mauriello and Park, Coralville Reservoir supports a commercial fishery for buffalofish, a large fish in the sucker family. Their ecological model predicts that contamination of the reservoir with dieldrin results in an increase in the population of buffalofish, which are fairly tolerant of contamination compared to other species. However, the commercial fishery does not immediately benefit from the population increase because elevated levels of toxin in buffalofish tissues result in closure of the fishery from 1970 to 1979. Furthermore, once dieldrin contamination is eliminated and the commercial fishery is reopened, the ecological model predicts that the population of buffalofish falls below its pre-contamination level, possible due to competition with other recovering species.

The evaluation of the commercial benefits of elimination of dieldrin included four steps:

1) estimating the commercial harvest of buffalofish in each year of the simulation by combining population density data with an estimate of the quantity of buffalofish harvested before the dieldrin contamination. To calculate the increase in harvest attributable to eliminating dieldrin from the reservoir, we also estimated the commercial harvest in a scenario in which dieldrin is not eliminated from the reservoir.
2) calculating the market value of the increase in commercial catch by multiplying the additional quantity of buffalofish harvested by the commercial price per pound.
3) estimating the change in consumer and producer surplus from eliminating dieldrin, based on the market value of the increase in commercial buffalofish catch.
4) calculating the total commercial benefits of eliminating dieldrin from the reservoir by adding the changes in consumer and producer surplus.

Step 1: Estimate the change in commercial harvest attributable to eliminating dieldrin from Coralville Reservoir.

The ecological model predicts the population density of buffalofish in Coralville Reservoir in each year of the simulation. This population density was multiplied by the total surface area of the reservoir (5,376 acres) to calculate the total buffalofish population in the reservoir. Then, based on the assumption that one fifth of the fish population is harvested by commercial anglers each year, the analysis estimated total commercial catch in each year. However, based on the aquatic contamination data from the ecological model, it was assumed that fish consumption advisories prevent commercial harvest of buffalofish from 1970 through 1979.

Mauriello and Park did not estimate buffalofish populations under a scenario in which dieldrin is not eliminated from Coralville reservoir. Thus, to estimate the incremental commercial benefits of eliminating dieldrin from the reservoir, it was assumed that if dieldrin had not been eliminated, the reservoir would have remained closed to commercial fishing through the end of the period covered by this analysis. Thus, the increase in commercial catch resulting from eliminating dieldrin is equal to the total catch in the remediation scenario in the years following the lifting of the fishing ban. Table 5-8 presents the results of this calculation. The table shows that eliminating dieldrin does not result in an increase in commercial harvest until 1980.

Table 5-8: Increase in Commercial Harvest of Buffalofish at Coralville Reservoir Attributable to Elimination of Dieldrin

Year	Buffalofish Population Density (lb/acre)	Commercial Harvest with Remediation (entire lake)[a]	Commercial Harvest without Remediation (entire lake)[b]	Increase in Commercial Harvest (entire lake)
1969	117	126,315	126,315	0
1970	114	0	0	0
1971	126	0	0	0
1972	129	0	0	0
1973	124	0	0	0
1974	130	0	0	0
1975	144	0	0	0
1976	169	0	0	0
1977	195	0	0	0
1978	234	0	0	0
1979	238	0	0	0
1980	202	217,617	0	217,617
1981	169	181,567	0	181,567
1982	143	153,737	0	153,737
1983	111	119,547	0	119,547
1984	78	84,139	0	84,139
1985	61	65,699	0	65,699
1986	51	54,345	0	54,345

[a] In the scenario with remediation, we assume that a fish consumption advisory prevented commercial fishing from 1970 through 1979.
[b] In the scenario with no remediation, we assume that a fish consumption advisory prevented commercial fishing from 1970 through 1986.

Step 2: Calculate the market value of the increase in buffalofish harvest.

Two sources of information on the price of buffalofish were identified. According to the Iowa Department of Natural Resources (IDNR), in 1987, 1.5 million pounds of buffalofish with a total value of $350,000 were harvested in Iowa, with an average price of $0.23 per pound ($0.38 per pound in 2003$) (Iowa DNR, 2005). Another source, Kentucky Fish and Wildlife, indicated that the price was $0.25 per pound in Kentucky in 1997 ($0.29 per pound in 2003$) (Kentucky FW, 1997). For this analysis, we used the average of these two prices: $0.33 per pound (2003$).

To calculate the market value of the increase in buffalofish harvest, we multiplied our estimate of the quantity of fish caught by the average price presented above.

Step 3: Estimate the change in consumer and producer surplus from eliminating dieldrin, based on the market value of the increase in commercial buffalofish catch.

To estimate the change in consumer surplus (i.e., the benefits to buffalofish consumers) from market changes resulting from eliminating dieldrin, the analysis assumed that consumer demand for buffalofish is relatively elastic (i.e., small changes in price result in large changes in the amount of fish purchased by consumers). Since there are a large number of substitutes available for buffalofish meat, this may be a reasonable assumption. However, in this situation, a change in the supply of buffalofish (such as the changes in supply that result from changes in harvest at Coralville Reservoir) would have little effect its price, and the resulting change in consumer surplus would be very small. This case study made the simplifying assumption that the change in consumer surplus would be zero.

The change in producer surplus (i.e., benefits to commercial anglers) was estimated by first calculating the change in commercial revenues from an increase in buffalofish harvest, which is equal to the price of buffalofish times the increase in the amount harvested. Since there was no information on the elasticity of supply for buffalofish, two scenarios were evaluated, one in which the supply curve is elastic with respect to price (i.e., buffalofish harvest is fairly dependent on the price of buffalofish), and one in which the supply curve is relatively inelastic (i.e., buffalofish harvest varies only a little with changes in price). Based on these two scenarios, a range of possible changes in producer surplus was estimated using the results of previous econometric studies of producer surplus in fish markets (Hupert, 1990, and Retting and McCarl, 1985). These value range from 40% of the change in revenues to 90% of the change in revenues. For example, in 1980, the first year the FCA is lifted, commercial revenues increase by $72,280, with an associated increase in producer surplus of $28,912 to $65,052.

Step 4: Calculate the commercial benefits of eliminating dieldrin from Coralville Reservoir by adding the changes in consumer and producer surplus.

Since it was assumed that there is no change in consumer surplus from eliminating dieldrin from the reservoir, the commercial benefits of this action are equal to the change in the producer surplus. Table 5-9 presents a summary of the commercial benefits of dieldrin remediation at Coralville Reservoir. The table shows that although the concentration of dieldrin in Coralville Reservoir declines steadily and buffalofish populations rise steadily after 1970, the fish consumption advisory on buffalofish prevents commercial harvests until 1980. In that year, commercial benefits are estimated to be between $28,912 and $65,052. However, by 1986, commercial benefits fall to a value between $7,220 and $16,245.

Table 5-9: Commercial Harvest of Buffalofish at Coralville Reservoir and Resulting Commercial Benefits (2003$)

Year	Increase in Commercial Harvest for Entire Lake	Commercial Benefits (Change in Producer Surplus)[a]	
		Low	High
1969	0	$0	$0
1970	0	$0	$0
1971	0	$0	$0
1972	0	$0	$0
1973	0	$0	$0
1974	0	$0	$0
1975	0	$0	$0
1976	0	$0	$0
1977	0	$0	$0
1978	0	$0	$0
1979	0	$0	$0
1980	217,617	$28,912	$65,052
1981	181,567	$24,123	$54,276
1982	153,737	$20,425	$45,957
1983	119,547	$15,883	$35,736
1984	84,139	$11,178	$25,152
1985	65,699	$8,729	$19,639
1986	54,345	$7,220	$16,245

[a] The change in consumer surplus from these changes in harvest is assumed to be zero. Thus, commercial benefits include only changes in producer surplus.

The primary limitation of this analysis is that the input values for a number of important variables are based on estimates or assumptions. For example, the results of the analysis are heavily dependent on the assumptions made about the percentage of the buffalofish population that is harvested each year, the price elasticity of supply and demand for buffalofish, the percentage of revenue that is associated with producer surplus, and the years in which the FCA is in effect.

5.3.3 Non-use Benefits of Improvement in Fish Population Health

The methodology used to assess the non-use benefits of the elimination of dieldrin from the Coralville Reservoir is similar to the methodology used for the selenium mitigation effort at Hyco Reservoir. The evaluation of these benefits included three steps: estimating the number of households that might reasonably hold non-use values for bluegill population health at the Coralville Reservoir; using the aquatic resource meta-analysis benefits model to predict how much those households would be willing to pay to eliminate dieldrin contamination at the reservoir; and finally, multiplying the number of affected households by average WTP for eliminating dieldrin.

The case study assumed that the population likely to hold non-use values for ecosystem health of the Coralville Reservoir would include all households within 25 miles of the reservoir. Using U.S. Census Bureau data, this population was estimated to include 133,456 households.[6]

To estimate how much each household would be willing to pay for the elimination of dieldrin contamination at the reservoir, the analysis used a meta-analysis model developed during EPA's analysis of the benefits of the final section 316(b) rule for Phase II facilities (Johnston et al., 2005). An overview of this model is provided in Section 4.3.3.

In order to use the meta-analysis model to estimate household non-use values for this case study, the changes resulting from eliminating dieldrin at the reservoir had to be mapped to the RFF water quality ladder. This required evaluating water quality at the reservoir under two scenarios: the remediation scenario described by Mauriello and Park, and an alternative scenario in which dieldrin is not eliminated from the reservoir. Since Coralville Reservoir is described as being shallow and eutrophic, but nonetheless supporting a healthy population of game fish, it was assumed that the WQL under both scenarios is 6.25 in 1969 (the year before the reservoir is contaminated with dieldrin). During the period from 1970 to 1980, under both scenarios, dieldrin contamination in the reservoir essentially eliminates all species except buffalofish, a large fish in the sucker family. Thus, it was assumed that the WQL during this period is 4.75—slightly below the value necessary to support game fish. Under the first scenario, the dieldrin concentration drops below EPA's water quality criterion for protection of freshwater aquatic life (0.0019 μg/L) in 1980, and walleye populations exceed their pre-contamination levels by 1982. Although the recovery of fish populations under the first scenario takes place over several years, the analysis made the simplifying assumption that the WQL remains at 4.75 until 1980, at which point it increases to 6.0 and stays at that level, which is slightly below the pre-contamination WQL. In the second scenario, there is no recovery, so the WQL remains at 4.75 from 1980 to the end of the analysis.

Table 5-10 summarizes the water quality under both scenarios.

[6] This calculation is documented in Appendix F.2.

Table 5-10: Water Quality Change Resulting from Elimination of Dieldrin at Coralville Reservoir[a]

Year	Water Quality (with remediation)	Water Quality (without remediation)	Change in Water Quality
1969	6.25	6.25	0
1970	4.75	4.75	0
1971	4.75	4.75	0
1972	4.75	4.75	0
1973	4.75	4.75	0
1974	4.75	4.75	0
1975	4.75	4.75	0
1976	4.75	4.75	0
1977	4.75	4.75	0
1978	4.75	4.75	0
1979	4.75	4.75	0
1980	4.75	4.75	0
1981	6.00	4.75	1.25
1982	6.00	4.75	1.25
1983	6.00	4.75	1.25
1984	6.00	4.75	1.25
1985	6.00	4.75	1.25
1986	6.00	4.75	1.25

[a] This table expresses water quality using the Resources for the Future water quality ladder.

Table 5-11 presents the meta-analysis variables and coefficients, as well as the values assigned to each variable. For detailed information about the variable definitions, refer to U.S. EPA (2004d). The following bullets briefly explain how we assigned values to each of the variables:

- Selected survey methodological variables (*interview*, *year_indx*, *nonparam*, *discrete_ch*, *protest_bids*, *outlier_bids*) were assigned values that reflect a discrete choice survey format conducted through in-person interviews, with protest bids and outlier bids eliminated. Other survey methodological variables were set to zero.
- *Income* was set to the median household income for Iowa in 2003 (U.S. Census Bureau, 2005a).
- The regional dummies were all set to zero, based on the judgment that these variables may be capturing study or author-level effects, not regional influences.
- Two of the resource description variables (*single_lake* and *num_riv_pond*) were set to values reflecting a change occurring on one reservoir. The remaining resource variables were set to zero.
- The water quality variables *WQ_fish* and *baseline* were assigned values that reflect a 1.25 point increase on the water quality ladder, from 4.75 to 6. The remainder of the WQ variables were set to zero, since the ecological model only models fish.
- The variable *non-users* was set to one so that the model would predict total WTP for non-users only—which by definition includes only non-use values. This value is assumed to be a lower bound for the non-use value for users of the resource, who, being more familiar with the resource, might be expected to hold greater non-use values for it (independently of any use values they also hold).

Based on the variable assignments shown in the table, non-user WTP for the specified resource change is $11.52 per household per year. A sensitivity analysis also calculated WTP based on a mail survey format (instead of an interview format). Under this alternative assumption, non-user WTP is $5.40—a six dollar difference.

Table 5-11: Estimating Non-use WTP for Improvement in Water Quality at Coralville Reservoir

Variable	Coefficient	Assigned Value: Based on Interview Survey Format	Assigned Value: Based on Mail Survey Format
Intercept	6.0158	1	1
year_indx	-0.1072	31	31
discrete_ch	0.3956	1	1
voluntary	-1.633	0	0
nonparam	-0.4472	1	1
income	0.00000058	64341	64341
wq_ladder	-0.3799	1	1
protest_bids	0.9537	1	1
outlier_bids	-0.8764	1	1
hi_response	-0.8094	1	1
single_river	-0.3378	0	0
single_lake	0.3193	1	1
multiple_river	-1.605	0	0
salt_ponds	0.7574	0	0
num_riv_pond	0.0791	1	1
regional_fresh	-0.0073	0	0
southeast	1.1482	0	0
plains	-0.8153	0	0
pacif_mount	-0.3125	0	0
multi_reg	0.5951	0	0
nonusers	-0.5017	1	1
WQ_fish	0.2055	1	1
WQ_shell	0.2561	0	0
WQ_many	0.2332	0	0
WQ_non	0.4695	0	0
fishplus	0.8052	0	0
baseline	-0.1265	4.75	4.75
interview	1.3252	1	0
mail	0.5666	0	1
lump_sum	0.5954	0	0
nonfish_uses	-0.1412	0	0
median_WTP	0.2206	0	0
WTP per Household, per Year (2003$)		**$11.52**	**$5.40**

The final step in our calculation of the non-use benefits of eliminating dieldrin contamination at Coralville Reservoir was to combine our estimate of the affected population with our estimates of household WTP. Table 5-12 presents the results of this calculation. The table shows that based on a total affected population of 133,456 local households, the cumulative non-use benefits of the selenium mitigation at the Hyco Reservoir could range from $4.3 million to $9.2

million. Note that these estimates include more than just the value of the changes in population for the five fish species discussed in this case study. Although using the variable *WQ_fish* limits the model to predicting non-use WTP for changes in water quality that primarily affect fish and fish habitat, the WTP value still includes the non-use value of *all* changes at the waterbody. The predicted non-use value includes WTP to improve the health of other fish species and to improve general water quality, in addition to WTP to improve the health of the five species discussed in this case study. Thus, the values in Table 5-12 should be considered an upper limit on the value of the improvement in populations of these five species. The actual value is likely to be lower.

Table 5-12: Non-use Benefits of Eliminating Dieldrin Contamination at Coralville Reservoir (undiscounted, 2003$)

Year	Water Quality Change	Non-use Value Per Household (2003$)		Total Non-use Benefits (2003$)[b]	
		Low[a]	High[a]	Low	High
1969	0	$0.00	$0.00	$0	$0
1970	0	$0.00	$0.00	$0	$0
1971	0	$0.00	$0.00	$0	$0
1972	0	$0.00	$0.00	$0	$0
1973	0	$0.00	$0.00	$0	$0
1974	0	$0.00	$0.00	$0	$0
1975	0	$0.00	$0.00	$0	$0
1976	0	$0.00	$0.00	$0	$0
1977	0	$0.00	$0.00	$0	$0
1978	0	$0.00	$0.00	$0	$0
1979	0	$0.00	$0.00	$0	$0
1980	0	$0.00	$0.00	$0	$0
1981	1.25	$5.40	$11.52	$720,183	$1,537,796
1982	1.25	$5.40	$11.52	$720,183	$1,537,796
1983	1.25	$5.40	$11.52	$720,183	$1,537,796
1984	1.25	$5.40	$11.52	$720,183	$1,537,796
1985	1.25	$5.40	$11.52	$720,183	$1,537,796
1986	1.25	$5.40	$11.52	$720,183	$1,537,796
Total, undiscounted				**$4,321,098**	**$9,226,775**

[a] Low and high per-household non-use values are estimated from the meta-analysis regression equation based on mail and interview survey formats, respectively.
[b] Total undiscounted non-use benefits were calculated by multiplying the per-household non-use values by the population within 25 miles of Coralville Reservoir, which is 133,456 households.

There are several significant limitations and uncertainties associated with using the meta-analysis model to estimate non-use values for individuals affected by the Coralville Reservoir case study. First, as in any benefit transfer, the policy site does not match perfectly to the studies on which the meta-analysis was based, and the meta-model does not factor in all characteristics of the Coralville River dieldrin mitigation. For example, elimination of dieldrin at the reservoir does not map perfectly to the water quality ladder. Additionally, important characteristics of Coralville Reservoir, such as the presence of a commercial fishery for buffalofish, are not included in the meta-model. One additional limitation of this analysis is that it does not consider short-term temporal fluctuations in fish populations. Populations of different species recover at different rates, and some populations of some species, such as walleye and shad, decline somewhat in the later years of the recovery.

5.3.4 Total Benefits of Improvement in Fish Population Health

The final step in our evaluation of the benefits of eliminating dieldrin from Coralville Reservoir was to calculate total social welfare gain. Comprehensive, appropriate estimates of total resource value include both use and non-use values, such that the resulting total value estimates may be compared to total social cost. The total benefits are the sum of the recreational, commercial, and non-use values calculated in the previous sections. Table 5-13 presents the results of this calculation. The table shows that the total value of the recovery of fish populations after the elimination of dieldrin from Coralville Reservoir could range from $6.55 million to $11.61 million.

Table 5-13: Total Social Benefits of Fish Population Recovery at Coralville Reservoir (undiscounted, 2003$)

Year	Recreational Benefits	Commercial Benefits		Non-use Benefits[a]		Total Benefits[a]	
		Low	High	Low	High	Low	High
1969	$0	$0	$0	$0	$0	$0	$0
1970	$0	$0	$0	$0	$0	$0	$0
1971	$0	$0	$0	$0	$0	$0	$0
1972	$0	$0	$0	$0	$0	$0	$0
1973	$0	$0	$0	$0	$0	$0	$0
1974	$0	$0	$0	$0	$0	$0	$0
1975	$0	$0	$0	$0	$0	$0	$0
1976	$0	$0	$0	$0	$0	$0	$0
1977	$0	$0	$0	$0	$0	$0	$0
1978	$758	$0	$0	$0	$0	$758	$758
1979	$1,667	$0	$0	$0	$0	$1,667	$1,667
1980	$5,317	$28,912	$65,052	$0	$0	$34,229	$70,369
1981	$28,965	$24,123	$54,276	$720,183	$1,537,796	$773,271	$1,621,037
1982	$157,362	$20,425	$45,957	$720,183	$1,537,796	$897,970	$1,741,115
1983	$540,723	$15,883	$35,736	$720,183	$1,537,796	$1,276,789	$2,114,255
1984	$627,895	$11,178	$25,152	$720,183	$1,537,796	$1,359,257	$2,190,843
1985	$427,714	$8,729	$19,639	$720,183	$1,537,796	$1,156,625	$1,985,149
1986	$326,687	$7,220	$16,245	$720,183	$1,537,796	$1,054,090	$1,880,728
Total, undiscounted	**$2,117,087**	**$116,470**	**$262,057**	**$4,321,098**	**$9,226,775**	**$6,554,656**	**$11,605,920**

[a] The non-use value of restoring fish populations is a subset of the non-use value of the dieldrin mitigation. Thus, all non-use estimates (and the total social value estimates) should be considered upper bounds.

Because of the large number of assumptions necessary to estimate each of the component values presented in Table 5-13, the total social value estimates should be interpreted with caution. Availability of better data, particularly data on how changes in the ecological endpoints described in the Mauriello and Park study translate into changes in angler participation and catch rates, would significantly improve the accuracy of the recreational benefits estimates. Also, the non-use benefits estimates could be significantly improved by removing the portion of the non-use estimates attributable to habitat improvements that are unrelated to fish.

5.4 Extension of Ecological and Economic Assessment to Other Case Studies

Input data requirements and the methodology for this case study can be applied to various ecological risk scenarios. The model utilized in the ecological risk assessment is readily available to all agencies or individuals conducting such assessments, and is offered on EPA's web site. Although in this case study, there is only one contaminant and five species of concern, AQUATOX can be applied to a more complex scenario of multiple contaminants and multiple species at various trophic levels. It also has contaminant libraries and biological data libraries, reducing the need to collect extensive data. Another input to the model requires characteristics of the water body, and AQUATOX has a link to a database (BASINS) with national watershed information.

The U.S. Army Corps of Engineers required close monitoring of water quality in the reservoir because they had overseen its construction, and thus they maintained monitoring data for the reservoir that was input in the model. Annual studies provided data on initial conditions before the ban on aldrin and dieldrin, and site information on volume and flow was maintained by the Army Corps and by the U.S. Geological Survey. Additionally, data are more readily available on aldrin and dieldrin because they are persistent in the environment and have been tracked by agencies including EPA and the Fish and Wildlife Service. Similar data would be available for other organochlorine insecticides such as DDT and chlordane, which were also banned for agricultural use in the 1970's.

The valuation methodology for estimating the value of changes in recreational catch can by applied to a variety of situations. Output datasets from the AQUATOX model are excellent candidates for this type of valuation—as are any ecological models that provide assessment endpoint measures that can be used to estimate changes in recreational catch rates. The recreational meta-model includes variables for a variety of species and resource characteristics, making it a flexible tool for valuing a range of fish-related ecological model outputs.

The methodology used to value the effect of ecological changes on commercial fishing can also be generalized to a variety of situations. This methodology requires data on market prices for fish, the expected change in commercial fish harvest, and the sensitivity of fish markets to changes in price. Fish prices are always available; commercial fish harvest can often be estimated by combining data on commercial catch rates with fish population data from an ecological model; and reasonable assumptions can be made about the sensitivity of fish markets to changes in price. Thus, in most situations, this methodology can be used to the commercial benefits of ecological changes predicted by models such as AQUATOX.

Like the recreational meta-analysis, the non-use meta-analysis can be potentially used to estimate non-use values for a wide range of resource changes. The primary requirement for applying this methodology to the output of an ecological model is that the assessment endpoint measures must be mapped to the Resources for the Future water quality ladder. Whether this is possible will depend on the kinds of ecological changes being considered. The model can actually be used to value changes that are ecologically very complex, as long as those changes result in a general improvement in aquatic resource health. It is more difficult to apply the model to situations where small changes occur to specific resources, or where the resources change in a way unlike the changes considered in the studies underlying the meta-analysis—for example, encroachment of native vegetation by an exotic plant species.

6. Economic Valuation of Benefits: Case Study Using Superfund Ecological Risk Assessment

6.1. Ecological Assessment Case Study

The Superfund risk assessment framework used by Johnston et al. (2002) combines a variety of ecological indicators to evaluate the risk to six different ecological communities within Clark Cove, Maine. For each community, the authors present indicators of the evidence of ecological effect and exposure. This study contains a conceptual model for linking the specific exposure and effects measures to broader ecological endpoints: the health of pelagic species, epibenthic species, the benthic community, eelgrass plants, the salt marsh community, waterfowl, and birds of prey. The authors then combine the measures of effect and exposure to generate an indicator of the evidence of risk to each community.

The approach taken assesses a large number of different resources, and uses a weight of evidence approach, which is recommended in ecological risk. The weight of evidence approach takes into account the quality and strength of available data on each endpoint, such as the quality of data obtained on winter flounder abundance, or salt marsh plant species cover. A wide range of data was collected because of the Superfund site status, which requires monitoring and assessment of ecological conditions. These data would not necessarily be available at all sites requiring risk assessment under the Superfund program. However, the methods used are standard approaches, and did not require intensive population or ecosystem modeling efforts, which not all risk assessors can readily conduct.

6.2. Economic Benefits Assessment

The various indicators produced by the risk assessment framework are challenging to value using economic techniques, for the following reasons:

- Although some of the ecological endpoints used as inputs for the risk assessment model can be linked directly to economic values, other endpoints have only indirect relationships with economically valuable ecological service flows. For example, although winter flounder size and abundance has a direct impact on the welfare of recreational anglers, marsh grass morphology has at best a very indirect relationship with economic use values.
- The community level endpoints generated by the model are very general and thus difficult to use to evaluate effects on specific species with economic use values. For example, it would be very difficult to use an overall indicator of epibenthic community health to calculate changes to shellfish harvest.
- The effect, exposure, and risk indicators are all qualitative, making it difficult to use them to estimate specific economic values.
- The way that people value reductions in ecological risk, as opposed to ecological damages, is not well understood.

For all of these reasons, this report does not attempt to generate quantitative estimates of the value of potential ecological changes at Clark Cove. Instead, the types of economic values associated with each ecological assessment endpoint are summarized. Then, for each economic value, potentially relevant valuation studies are cited.

Table 6-1 lists the ecological measures of effect and assessment endpoints presented in Johnston et al. (2002), describes the economically valuable ecological services associated with each of those assessment endpoints, and lists the types of economic values associated with those services. Because of the complex and interrelated nature of the components of the Clark Cove ecosystem, it is impossible to list all possible linkages between ecological services and economic values. The ecological services and economic values presented in Table 6-1 should be viewed as a preliminary listing.

Table 6-1: Relationship Between Ecological Measures of Effect, Ecological Services, and Economic Values at Clark Cove

Ecological Measures Used to Assess Potential Changes in Community Health[a]	Ecological Assessment Endpoint[a]	Related Ecological Services with Economic Values[b]	Economic Values
• Phytoplankton biomass • Mussel growth after 28 days • Sea urchin fertilization after exposure to water • Winter flounder abundance and size • Winter flounder spleen histopathology	Pelagic community health	• Habitat for recreationally and commercially valuable fish species	• Welfare gain from recreational fishing • Welfare gain from commercial fishing
		• Pelagic community health	• Non-use values for pelagic community health (as part of overall ecosystem health)
• Lobster density • Indigenous mussel density • Indigenous mussel shell length • Indigenous mussel condition index • Fucoid algae biomass	Epibenthic community health	• Habitat for recreationally and commercially valuable species (lobsters, mussels, etc)	• Welfare gain from recreational harvest of crustaceans and shellfish • Welfare gain from commercial harvest of crustaceans and shellfish
		• Epibenthic community health	• Non-use values for epibenthic community health (as part of overall ecosystem health)
• Amphipod mortality after exposure to sediment • Benthic community richness • Benthic community density • Benthic community evenness	Benthic community health	• Benthic community health	• Non-use values for benthic community health (as part of overall ecosystem health)
• Eelgrass leaf morphology • Eelgrass root and rhizoe morphology • Eelgrass vegetative shoot density • Eelgrass reproductive shoot density; • Eelgrass ratio of leaves to shoots • Eelgrass spatial distribution	Eelgrass community health	• Habitat and/or forage area for recreationally and commercially valuable fish and invertebrates (Atlantic cod, tomcod, winter flounder, cunner, rock crab, American lobster, etc) [1]	• Welfare gain from recreational fishing and harvest of crustaceans and shellfish • Welfare gain from commercial fishing and harvest of crustaceans and shellfish
		• Forage area for waterfowl[1]	• Welfare gain from bird watching and hunting • Non-use values for avian health

Ecological Measures Used to Assess Potential Changes in Community Health[a]	Ecological Assessment Endpoint[a]	Related Ecological Services with Economic Values[b]	Economic Values
		• Sediment stabilization and prevention of shore and beach erosion[1]	• Reduction of damages and remediation costs associated with beach and shore erosion
		• Eelgrass community health	• Non-use values for eelgrass community health (as part of overall ecosystem health)
• Marsh grass cover • Marsh grass morphology • Amphipod abundance	Salt marsh community health	• Habitat for invertebrates[2] • Spawning and nursery area for fish[2]	• Welfare gain from recreational fishing and harvest of crustaceans and shellfish • Welfare gain from commercial fishing and harvest of crustaceans and shellfish
		• Breeding and feeding habitat for wildlife[2]	• Welfare gain from bird and wildlife watching and hunting
		• Salt marsh community health[2]	• Non-use values for salt marsh community health (as part of overall ecosystem health)
No measures of effect were presented in Johnston et al. (2002)	Avian health	• Avian health	• Welfare gain from bird watching and hunting • Non-use values for avian health
[a] Ecological measures and ecological assessment endpoints taken from Johnston et al. (2002). [b] Many of the ecological services listed in this column have little or no direct economic value, but may indirectly affect other ecological service flows that do have significant economic values. For example, although there is no direct human welfare gain from fish spawning in salt marsh habitat, there is a significant welfare gain from the effect of growth of fish populations on commercial and recreational angling.			
References: [1] *Wilbur, Anthony. 2004. "Spotlight on Eelgrass: A Species and Habitat at Risk." http://www.mass.gov/czm/coastlines/2004-2005/habitat/e_grass.htm.* [2] *Stratus Consulting. "Ecological Maturation in Restored Salt Marshes." http://www.stratusconsulting.com/Staff/PDFs/Maturity.PDF.*			

Table 6-2 summarizes the economic values presented in Table 6-1. For each type of economic value, the table presents specific examples of the ecological services being valued, and presents citations to relevant valuation literature for those examples.

Table 6-2: Economic Values Associated with Ecological Services Provided by Clark Cove

Type of Economic Value	Examples of Economic Value (at Clark Cove)	Selected Citations to Relevant Valuation Literature, Market Data, and Cost Studies
Value of recreational fishing and harvest of crustaceans and shellfish	Recreational angling for winter flounder	• Hicks, Robert. 2002. *Stated Preference Methods for Environmental Management: Recreational Summer Flounder Angling in the Northeastern United States.* Final report prepared for Fisheries Statistics and Economics Division, Office of Science and Technology, National Marine Fisheries Service. Requisition Request # NFFKS-18. March. • U.S. EPA. 2004. Regional Benefits Assessment for the Proposed Section 316(b) Rule for Phase III Facilities. http://www.epa.gov/waterscience/316b/ph3docs/p3_rba_fullreport.pdf.
	Recreational angling for Atlantic cod	• Rowe, R.D., E.R. Morey, A.D. Ross, and W.D. Shaw. 1985. *Valuing Marine Recreational Fishing on the Pacific Coast.* Energy and Resource Consultants Inc. Report prepared for the National Marine Fisheries Service, National Oceanic and Atmospheric Administration. Report LJ-85-18C. March. • Schuhmann, Peter William. 1997. "Deriving Species-Specific Benefits Measures for Expected Catch Improvements in a Random Utility Framework." *Marine Resource Economics*. 13:1-21. • U.S. EPA. 2004. Regional Benefits Assessment for the Proposed Section 316(b) Rule for Phase III Facilities. http://www.epa.gov/waterscience/316b/ph3docs/p3_rba_fullreport.pdf.
	Recreational collection of lobsters	• Anderson, Eric. 1989. Economic Benefits of Habitat Restoration: Seagrass and the Virginia Hard-Shell Blue Crab Fishery. *North American Journal of Fisheries Management*. Vol. 9, pp. 140-149.
	Recreational collection of mussels	• *No relevant studies identified*
Value of commercial fishing and harvest of crustaceans and shellfish	Commercial fishing for winter flounder	• NOAA Fisheries. 2005. "Fishery Market News." http://www.st.nmfs.gov/st1/market_news/
	Commercial fishing for Atlantic cod	• NOAA Fisheries. 2005. "Fishery Market News." http://www.st.nmfs.gov/st1/market_news/
	Commercial collection of lobsters	• NOAA Fisheries. 2005. "Fishery Market News." http://www.st.nmfs.gov/st1/market_news/ • Milton, J.W., S. L. Larkin, and N. M. Ehrhardt. 1999. *Bioeconomic Models of the Florida Commercial Spiny Lobster Fishery*. Sea Grant Report Number 117. Florida Sea Grant College Program, Florida. • Wang, Stanley D. H. and Christopher B. Kellogg. 1988. "An Econometric Model for American Lobster." *Marine Resource Economics*. 5(1): 61-70.
	Commercial collection of mussels	• NOAA Fisheries. 2005. "Fishery Market News." http://www.st.nmfs.gov/st1/market_news/

Type of Economic Value	Examples of Economic Value (at Clark Cove)	Selected Citations to Relevant Valuation Literature, Market Data, and Cost Studies
Value of bird watching	Watching waterfowl and seabirds	• Boyle, Kevin J., Richard C. Bishop, Daniel Hellerstein, Mike Welch, Mary C. Ahearn, Andrew Laughland, John Charbonneau, and Raymond O'Conner. 1998. "Test of Scope in Contingent Valuation Studies: Are the Numbers for the Birds?" Paper presented at the World Congress of Environmental and Resource Economists. Venice, Italy. June 25-27. • Loomis, John. 1989. *Valuing Nonconsumptive Use and Preservation Values of Game and Nongame Wildlife in California: Results of Surveys on Deer, Birds, and Mono Lake*. Division of Environmental Studies, Department of Agricultural Economics, University of California, Davis.
Value of bird hunting	Hunting for waterfowl	• Sorg, C.F. and L.J. Nelson. 1987. *Net Economic Value of Waterfowl Hunting in Idaho*. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado 80526. • Cooper, J. and J. Loomis. 1993. "Testing Whether Waterfowl Hunting Benefits Increase with Greater Water Deliveries to Wetlands." *Environmental and Resource Economics*. Vol 3: 545-561. • Duffield, J. and C. Neher. 1991. *Montana Waterfowl Hunting, A Contingent Valuation Assessment of Economic Benefits to Hunters*. Montana Department of Fish, Wildlife, and Parks. • Hay, M.. 1988. *Net Economic Recreation Values for Deer, Elk, and Waterfowl Hunting and Bass Fishing*. U.S. Department of the Interior, Fish and Wildlife Service.
Non-use values associated with maintaining healthy bird populations	Non-use values for maintaining healthy waterfowl and seabird populations	• Loomis, John. 1989. *Valuing Nonconsumptive Use and Preservation Values of Game and Nongame Wildlife in California: Results of Surveys on Deer, Birds, and Mono Lake*. Division of Environmental Studies, Department of Agricultural Economics, University of California, Davis.

Type of Economic Value	Examples of Economic Value (at Clark Cove)	Selected Citations to Relevant Valuation Literature, Market Data, and Cost Studies
Non-use values associated with general aquatic ecosystem health	Non-use values for marine ecosystem health	• Huang, Ju-Chin, T. Haab, J.C. Whitehead. 1997. "Willingness to Pay for Quality Improvements: Should Revealed and Stated Preference Data Be Combined?" *Journal of Environmental Economics and Management*. Vol. 34, No. 3, pp. 240-255. • Hayes, K.M., T.J.Tyrell and G. Anderson. 1992. "Estimating the Benefits of Water Quality Improvements in the Upper Narragansett Bay." *Marine Resource Economics*. Vol. 7, pp. 75-85. • Bockstael, N.E., K.E. McConnell and I.E. Strand. 1989. "Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay." *Marine Resource Economics*. Vol. 6: 1-18. • Whitehead, John C., G.C. Blomquist, T.J. Hoban, W.B. Clifford. 1995. "Assessing the Validity and Reliability of Contingent Values: A Comparison of On-Site Users, Off-Site Users, and Non-users." *Journal of Environmental Economics and Management*. Vol. 29. • Carson, Richard T., W. M. Hanemann, R. J. Kopp, J. A. Krosnick, R. C. Mitchell, S. Presser, P. A. Ruud, and V. K. Smith. 1994. *Prospective Interim Lost Use Value due to DDT and PCB Contamination in the Southern California Bight. Volume 2.* Report to the National Oceanic and Atmospheric Administration, September. • Whittington, Dale, G. Cassidy, D. Amaral, E. McClelland, H. Wang and C. Poulos. 1994. *The Economic Value of Improving the Environmental Quality of Galveston Bay*. Department of Environmental Sciences and Engineering, University of North Carolina at Chapel Hill. • Johnston, R.J., E.Y. Besedin , R. Iovanna, C. Miller, R. Wardwell, M. Ranson. 2005. "Systematic Variation in Willingness to Pay for Aquatic Resource Improvements and Implications for Benefit Transfer: A Meta-Analysis." *Canadian Journal of Agricultural Economics* (forthcoming).
Reduction of damages and remediation costs associated with beach and shore erosion	Reduction of damages and remediation costs associated with shore erosion	• U.S. Army Engineer Institute for Water Resources. 1994. *Shoreline protection and beach erosion control study, Phase I: Cost comparison of shoreline protection projects of the U.S. Army Corps of Engineers*. Alexandria, Virginia. • US Army Corps of Engineers, New England Division. *Prospect Beach Shore Protection and Erosion Control Project, West Haven, Connecticut: Detailed Project Report and Environmental Assessment*. Waltham, Massachusetts.

7. FUTURE DIRECTIONS

The objective of this report was to evaluate how outputs available from existing ecological risk assessment methods and techniques could be used to economic benefits assessments. The case studies in this report focused on population modeling risk assessment approaches, but did not include other studies of other types of ecosystem processes and services. The Ecological Benefits Strategy (US EPA, 2004) lists as action items the need to "*develop a catalogue of existing* ***relevant ecosystem process models*** *at different geographic scales to support benefits assessment*" and to "*expand the portfolio of* ***models to address the ecosystem processes*** *important to benefits assessment at multiple geographic scales*." The literature review and research conducted for this report revealed a number of existing local level case studies that characterize ecosystem functions using a wide variety of models and attempt to apply economic value to these functions. For example, the study on drinking water filtration function of a watershed cited in a recent NAS report, recent ORD case studies on integrated ecological risk assessment and economic analysis for watersheds in US EPA Region V, and recent efforts in Portland, Ore., (the Portland Lents case study) have looked at valuing watersheds/ecosystems based on their functions and have assigned economic values using a variety of techniques. These applications have been tailored to site conditions. These models and inputs used, and the results, could be uses as the basis for recommendations for generalizing these approaches and results for use in broader EPA regulatory settings.

The research conducted for this study also revealed the need to synthesize information from economic valuation studies that can be linked to most common ecological assessment endpoints generated by the identified risk assessment models. The purpose of this analysis/synthesis would be to create an adjustable valuation function that would account for resource and population characteristics on willingness-to-pay (WTP) for changes in the relevant changes in ecosystem functions. Depending on the number and quality of available data, this may include statistical (i.e., regression-based) analysis of results and the relevant data from the original studies. If only a few studies are available for the relevant endpoints, ranges and central tendency values for the relevant WTP estimates could be developed. In the past, regression-based meta-analyses of WTP for water quality improvements and for changes in recreational catch rates have been developed; there is a need to explore the development of similar meta-models for other resources/ecosystem services, including shellfish, birds, and other wildlife.

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Appendices

Appendix A: Models and Case Study Details

Provided as separate Excel file – "Appendix A Model Details.xls"

Appendix B: Ecological Risk Assessment Guidance Sources

Provided as separate Excel file – "Appendix B Model Sources.xls"

Appendix C: Contacts for Ecological Risk Models

Appendix D: Economic Literature References

Provided as separate Excel file – "Appendix D Economic Lit Search.xls"

Appendix E: Estimating Catch Rates, Angling Participation, and Affected Population at Hyco Reservoir

Appendix F: Estimating Catch Rates and Affected Population at Coralville Reservoir

Appendix B. Guidance Documents and General Reports

Title	Author	Year	Online Location	Description
Ecological Risk Assessment in the Federal Government	Committee on Environment and Natural Resources of the National Science and Technology Council	1999	http://yosemite.epa.gov/SAB/sabcvpess.nsf/0/b882baf473df807185256de4006a39a5/$FILE/ecorisk.pdf	Committee assessment of eco-risk assessment methods used in various agencies (with case examples)
Incorporating Ecological Risk Assessment into Remedial Investigation/ Feasibility Study Work Plans	DOE	1994	http://homer.ornl.gov/oepa/guidance/listsubdocs.cfm?ID=255&Home=Homer	Describes the steps in an ecological risk assessment, including types of data needed..
Risk Characterization for Ecological Risk Assessment of Contaminated Sites	DOE	1996	http://www.esd.ornl.gov/programs/ecorisk/documents/tm200.pdf	Describes DOE approach for assessing risk in different cases, and using multiple lines of evidence.
ECOFRAM Aquatic Draft Report	Ecological Committee on FIFRA Risk Assessment Methods (ECOFRAM) (includes EPA, OPP)	1999	http://www.epa.gov/oppefed1/ecorisk/aquareport.pdf	Draft report on recommendations for aquatic risk assessments, tools available, and the steps in a tiered risk assessment
ECOFRAM Terrestrial Draft Report	Ecological Committee on FIFRA Risk Assessment Methods (ECOFRAM) (includes EPA, OPP)	1999	http://www.epa.gov/oppefed1/ecorisk/terrreport.pdf	Draft report on recommendations for terrestrial risk assessments, tools available, and the steps in a tiered risk assessment, probabilistic risk assessment.
Ecological Risk Assessment Bulletins--Supplement to RAGS	EPA	2001	http://www.epa.gov/region4/waste/ots/ecolbul.htm	Region 4 guidance for CERCLA ecological risk assessments.
Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment	EPA	2003	http://cfpub2.epa.gov/ncea/cfm/recordisplay.cfm?deid=55131	Provides guidance to risk assessors involved in conducting an ecological assessment. The document describes a set of generic assessment endpoints, that can be adapted for specific assessments.
OPP's Initiative to Revise the Ecological Assessment Process for Pesticides	EPA, OPP	2004	http://www.epa.gov/oppefed1/ecorisk/	Links to EPA and SETAC documents relating to aquatic and terrestrial risk assessments
Implementation Plan for Probabilistic Ecological Assessment	EPA, OPP	2000	http://www.epa.gov/scipoly/sap/2000/index.htm	Includes questions and responses resulting from a meeting on the use of probabilistic ecological assessments

Appendix B. Guidance Documents and General Reports

Title	Author	Year	Online Location	Description
Overview of the Ecological Risk Assessment Process in OPP	EPA, OPP	2004	http://www.epa.gov/oppfead1/endanger/consultation/ecorisk-overview.pdf	Contains an overview of both the screening level and species-specific ecological risk assessment process at OPP
Guidelines for Ecological Risk Assessment	EPA, ORD	1998	http://cfpub2.epa.gov/ncea/cfm/recordisplay.cfm?deid=12460	EPA's guidelines for the ERA process. Covers all steps involved, from problem formulation through risk characterization.
Wildlife Exposure Factors Handbook	EPA, ORD	1993	http://cfpub2.epa.gov/ncea/cfm/wefh.cfm?ActType=default	Provides data, references, and guidance for conducting exposure assessments for wildlife species exposed to toxic chemicals
NCEA Risk Assessment Models	EPA, ORD	2003	http://cfpub2.epa.gov/ncea/cfm/nceariskmodels.cfm?ActType=DatabaseAndTools&detype=model&excCol=archive	Lists all of NCEA's risk models available on the web. Models include Benchmark Dose Software, Exposure Model for Soil-Organic Fate and Transport, Wildlife Contaminants Exposure Model,etc.
Examination of EPA's Risk Assessment Principles and Practices	EPA, OSA	2004	http://www.epa.gov/osa/ratf.htm	EPA staff review of how risk assessment is conducted at EPA
Ecological Risk Assessment Guidance for Superfund	EPA, Superfund	1997	http://www.epa.gov/superfund/programs/risk/ecorisk/ecorisk.htm	Reviews types of sources and data available to conduct an ERA. Also has statistical considerations
Tools for Ecological Risk Assessment	EPA, Superfund	2004	http://www.epa.gov/superfund/programs/risk/tooleco.htm	Provides links to ERA guidance
Linking population-level risk assessment with landscape and habitat models	H. Resit Akcakaya	2001	http://www.ramas.com/STOTEN.pdf	Describes an approach to linking a landscape dynamics program (LANDIS) to a metapopulation modeling program (RAMAS), to incorporate transitional dynamics of the landscape into assessment of viability and threat.
Navy guidance for conducting ecological risk assessments	Navy	1999	http://web.ead.anl.gov/ecorisk/index.cfm	Has links to the Navy's new guidance on conducting ecological risk assessments

Appendix B. Guidance Documents and General Reports

Title	Author	Year	Online Location	Description
Population-Level Ecological Risk Assessment - Proposal for a SETAC Pellston Workshop	SETAC	2002	http://www.setac.org/eraag/Pellston_Workshop_12-021.pdf	Describes issues with population level risk assessment, asks questions to be discussed during the workshop
Framework for Integrated Risk Assessments	WHO International Programme on Chemical Safety	2003	Human and Ecological Risk Assessment, Special Section, Volume 9, Number 1, February 2003.	Several journal articles on the methods and case examples of integrated human health and ecological risk assessments

APPENDIX C: CONTACTS FOR ECOLOGICAL RISK MODELS

Name	Affiliation	Email	Phone	Notes
Steve Newbold	EPA, NCEE	Newbold.Steve@epa.gov	(202) 566-2293	Identified by NCEE project team.
Tim Barry	EPA, NCEE	Barry.Timothy@epa.gov	(202) 566-2370	Identified by NCEE project team.
Susan Norton	EPA, ORD	Norton.susan@epa.gov	(202) 564-3246	Identified by NCEE project team. Suggested the volumes of ecological assessment case studies compiled by EPA in 1993 and 1994, in the process of creating EPA's 1998 Ecological Risk Assessment Guidelines.
Randy Bruins	EPA, ORD	Bruins.randy@epa.gov	(513) 569-7481	Identified by NCEE project team. Recommended a recent report by the Science Advisor office, "Examination of EPA's Risk Assessment Principles and Practices;" and internal drafts of the strategic plan for improving benefits assessment in the Agency and "Ecological Benefits Assessment: Problem Formulation and Research Needs." Recommended NWPCAM model used by OW. Recommended speaking with the economists Charles Griffiths (NCEE) and John Powers or Joel Corona (OW).

Name	Affiliation	Email	Phone	Notes
Wayne Munns	EPA, ORD	Munns.Wayne@epa.gov	(401) 782-3017	Identified by NCEE project team. He believed there might be overlap with ORD's work with OPP risk assessment staff to help them develop population effects models. Recommended types of population models: matrix models; demographic models; and spatially explicit individually based models (more development on this in Europe). Recommended contact with fisheries groups or NOAA that may have more standardized methods and economic focus than EPA. Also recommended speaking with Steve Newbold (NCEE) and Randy Bruins (ORD). Regarding Superfund risk assessments, he recommended work done at the Portsmouth Naval Yards in Kittery, Maine.
Glenn Suter	EPA, ORD	Suter.glenn@epa.gov	(513) 569-7808	Identified by NCEE project team. Has authored books on eco risk techniques, including "Species Sensitivity Distributions in EcoToxicology." Recommended the water criteria model Aquatox used by OW, and the aquatic and terrestrial methods outlined by ECOFRAM. Recommended speaking with Ed Fite at OPP.
Keith Sappington	EPA, ORD	Sappington.Keith@epa.gov	(202) 564-1538	Identified by NCEE project team. Recommended EPA's Risk Assessment Guidelines as a general framework. For population-level assessments, recommended papers by Larry Barnthouse. Recommended contacting staff in 316b program regarding benefits assessment.

Name	Affiliation	Email	Phone	Notes
TJ Wyatt	EPA, OPP	Wyatt.TJ@epa.gov	(703) 308-7228	Identified by NCEE project team. Discussed economic aspects which should be components of model such as market and non-market use and non-use value. Recommended speaking with Jim Wylan of UC Davis, Jim Sanchirico of Resources for the Future, and Marty Smith at Duke University for information on bioeconomic modeling.
Ed Fite	EPA, OPP	Fite.Edward@epa.gov	(703) 305-5368	Identified by NCEE project team. Recommended viewing OPP's website for recent research.
Ed Odenkirchen	EPA, OPP	Odenkirchen.Edward@epa.gov	(703) 305-6449	Identified by NCEE project team. Provided "Overview of Ecological Risk Assessment Process in OPP" and recommended viewing "Implementing Probabilistic Ecological Assessments" and "Advancing Ecological Risk Assessments within the EPA.
Ingrid Sunzenauer	EPA, OPP	Sunzenauer.Ingrid@epa.gov	(703) 305-5196	Identified by NCEE project team. Supported Ed Odenkirchen's recommendations, but did not provide additional references.
Michael Fogarty	Woods Hole	Mfogarty@whsun1.wh.whoi.edu		Identified by through Elena Besedin's contacts. Could not provide information on models that are both population based and contain an economic component. Suggested that population modeling is possible if information about lethal and non-lethal effects of toxicity is known for all vital rates (such as reproduction, etc), but did not provide information about a particular model.
Alyce Fritz	NOAA		(206) 526-6305	Identified via SETAC conference proceedings. Works on identifying assessment endpoints, but recommended speaking to Wayne Munns at EPA for population model details.

Name	Affiliation	Email	Phone	Notes
Lisa Pelstring	NOAA	Lisa.Pelstring@noaa.gov		Identified as damage assessment and restoration program (DARP) contact. Recommended materials on website.
Barry Berejikian	NOAA	Barry.Berejikian@noaa.gov		Identified by Sabrina Ise-Lovell because of work on salmon population models. Recommended speaking with Jeff Hard.
Jeff Hard	NOAA	Jeff.Hard@noaa.gov		Recommended speaking with Paul McElhany
Paul McElhany	NOAA	Paul.McElhany@noaa.gov	(206) 860-5608	Spoke at length about NOAA salmon population modeling (see text for more) and recommended Mobrand models used throughout the Northwest region.
Kenneth A. Rose	Louisiana State University	karose@lsu.edu	(225) 578-6346	Listed as a collaborator on DOE's population modeling report. He suggested looking at Applied Biomathematics' matrix models, and books published by SETAC (Society for Environmental Toxicology and Chemistry). He is also working on matrix models for 30 species, to see if life history traits can indicate vulnerability to stressors.
Lev Ginzburg and Resit Akcakaya	Applied Biomath-ematics		(631) 751-4350	Applied Biomathematics provides RAMAS ecological modeling software. For more information on how RAMAS can be used, they conduct workshops on using the software for risk assessments and population modeling (workshop costs are between $3,000 and $8,000.) There are examples in the User's Manual (comes with software purchase.) Their RAMAS GIS software has been used much more widely than RAMAS Ecotox.

Name	Affiliation	Email	Phone	Notes
Marjorie Coombs Wellman	EPA, OST (Office of Science and Technology)		(202) 566-0407	Listed as a contact for the AQUATOX model. She discussed the AQUATOX model and its capabilities, that it can be used for a complex ecosystem, and model species within different trophic guilds. It also has associated chemical libraries, including some pesticides, and toxicity for various species. Suggested the validation studies included in the AQUATOX User Manual for case studies. Suggested talking with Mary Frankenberry who might know about versions of AQUATOX tailored for OPP, or John Powers who has worked on models with economic benefit components.
John Powers	EPA, OW		(202) 564-5776	An economist, listed as a contact for NWPCAM. Also recommended by Randy Bruins and Marjorie Wellman as having looked at models for economic benefits. He explained basis of the NWPCAM model as a water quality simulation model, that takes one further step of assigning monetary value to the water quality values based on one valuation study. He suggested that any valuation dataset could be used to process the water quality endpoints modeled by NWPCAM.
Staci Gatica	EPA, ORD	Gatica.staci@epa.gov	(202) 564-2321	Identified by Applied Biomathematics, as the EPA contact coordinating a workshop on population models for EPA's Region 5 Superfund staff. She said the Superfund program had identified a need for population modeling. She recommended contacting Stace Cuje and Dale Matey for information on case studies (they are also part of the Ecological Risk Assessment Forum).

Name	Affiliation	Email	Phone	Notes
Ken Finkelstein (NOAA), and Fred Evans (Navy)	NOAA, U.S. Navy		Ken Finkelstein (617) 918-1499 Fred Evans (610) 595-0567 x159	Ken Finkelstein is a NOAA contact for the Portsmouth Naval Shipyard risk assessment. He suggested contacting Fred Evans (Navy) to obtain the ecological risk assessment (ERA) report for the Naval Shipyard site. We submitted a request to Fred to obtain the ERA.
David Brauner	EPA Region 5 Superfund	Brauner.David@epamail.epa.gov	(312) 886-1526	Contacted David for more recent case studies of Superfund risk assessments, that included endpoints beyond only hazard quotient. He suggested the assessment conducted at the St. Regis Paper Company site in Cass Lake, MN. He also sent the associated site's assessment endpoints.

Appendix E: Estimating Catch Rates, Angling Participation, and Affected Population at Hyco Reservoir

E.1 Estimating Bluegill Catch Rates

To estimate the baseline total recreational catch of bluegill at Hyco Reservoir, North Carolina, we relied on data from a report by Duke Power.[7] The report presents creel census data for several lakes and rivers in North Carolina and South Carolina. For each water body, the report provides (1) total recreational catch of all species, (2) the percentage composition of catch for a few major species, (3) total hours of angler effort per hectare, and (4) total hours of angler effort. From this data, we were able to estimate total catch per hectare for the main species targeted in each water body. Sufficient data were available to calculate total bluegill catch per hectare for two water bodies: Lake Wateree in South Carolina, and Lake Hickory in North Carolina. Total catch at these lakes was 21.8 sunfish/hectare (1990 data) and 57.7 bluegill/hectare (1997 data), respectively. We assume that total annual bluegill catch at Hyco Reservoir is equal to the average of these two estimates, or 39.8 fish/hectare.

In addition to estimating total annual bluegill catch, we estimate the average hourly catch rate for anglers targeting bluegill. Since we have no information specifically for anglers targeting bluegill, we assume that these anglers catch bluegill at the same rate as anglers targeting other species. The average catch rate for all species is 0.86 fish/hour at Lake Wateree, and 1.46 fish/hour at Lake Hickory, so we estimate that the average bluegill catch rate for bluegill anglers at Hyco Reservoir is the average of these values, or 1.16 bluegill/hour.

E.2 Estimating Participation for Anglers Targeting Bluegill

We estimated the number of trips taken to Hyco Reservoir based on additional data from the Duke Power report. According to this report, fishing pressure at Lake Wateree and Lake Hickory is 124 hours/hectare and 179.6 hours/hectare, respectively. Based on the average of these two values, we estimate that annual angling pressure at Hyco Reservoir is 151.8 hours per hectare. Assuming that the affected portion of the reservoir is 1,000 hectares, then anglers spend 151,800 hours fishing at the study site each year. By assuming that the average fishing trip is four hours long, we estimate that anglers make 37,950 trips to the reservoir each year.

E.3 Estimating Affected Population Near Hyco Reservoir

To calculate the number of households within a 25-mile radius of Hyco Reservoir, we used a map of census tracts from the U.S. Census Bureau[8] in combination with the U.S. EPA's Reach File[9] of waterways. These data were loaded into a geographical information system (GIS), which was used to plot the center of each census tract as well as calculate the 25-mile radius surrounding Hyco Reservoir. The total number of households was then calculated by summing the number of households in each census tract whose center fell within the 25-mile radius

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[7] Duke Power. 2003. Catabwa Wateree Project: First Stage Consultation Document. http://www.catawbahydrolicensing.com/pdfs/catwat_fscd_part5c.pdf.
[8] U.S. Census Bureau. 2000. United States Census 2000.
[9] U.S. Environmental Protection Agency (EPA). 1996. "USEPA Reach File Version 1.0 (RF1) for the Conterminous United States (CONUS)."

surrounding Hyco Reservoir. The result of this analysis showed that 85,624 households are located with 25 miles of Hyco Reservoir.

The decision to include all households within 25 miles of the study site was somewhat arbitrary. In all likelihood, many households located further than 25 miles from Hyco Reservoir may hold non-use values for changes in fish population health at the reservoir. Empirical studies in economic literature suggest that individuals are likely to hold non-use values for both local and regional resources.[10,11] Thus, this assumption may provide a conservative estimate of the total affected population near Hyco Reservoir.

[10] Pate, J. and J. Loomis. 1997. The effect of distance on willingness-to-pay values: A case study of wetlands and salmon in California. *Ecological Economics* 20:199-207.
[11] Schulze, W.D., R.D. Rowe, W.S. Breffle, R.R. Boyce, and G.H. McClelland. 1995. Contingent Valuation of Natural Resource Damages due to Injuries to the Upper Clark Fork River Basin. State of Montana Natural Resource Damage Litigation Program. Prepared by RCG/Hagler Bailly, Boulder, CO.

Appendix F: Estimating Catch Rates and Affected Population at Coralville Reservoir

F.1 Estimating Catch Rates at Coralville Reservoir

To estimate the baseline recreational catch of largemouth bass, bluegill, and walleye in 1969 at Coralville Reservoir, Iowa, we relied on data from a reports from the Hoosier Times[12] in Indiana and the Illinois Department of Natural Resources[13]. These reports present creel census data for West Boggs Reservoir and Shabbona Lake, two similar lakes located in Indiana and Illinois, respectively. For each water body, these reports provide number of fish caught per acre per year. In 1999, total catch at West Boggs Reservoir was 8.0 bass/acre and 54 bluegill/acre. In 1997, total catch at Shabbona Lake was 5.3 walleye/acre. Using this data, we estimated catch rates at Coralville Reservoir by assuming that catch rates at Coralville Reservoir are identical to catch rates at these lakes.

F.2 Estimating Affected Population Near Coralville Reservoir

To calculate the number of households within a 25-mile radius of Coralville Reservoir, we used a map of census tracts from the U.S. Census Bureau[14] in combination with the U.S. EPA's Reach File[15] of waterways. These data were loaded into a geographical information system (GIS), which was used to plot the center of each census tract as well as calculate the 25-mile radius surrounding Coralville Reservoir. The total number of households was then calculated by summing the number of households in each census tract whose center fell within the 25-mile radius surrounding Coralville Reservoir. The result of this analysis showed that 133,456 households are located with 25 miles of Coralville Reservoir. Note that because of way water bodies are defined in the EPA Reach File, Coralville Reservoir was assumed to include Lake McBride. Because these two water bodies are located very close to each other, this assumption had little effect on the calculated total number of households.

The decision to include all households within 25 miles of the study site was somewhat arbitrary. In all likelihood, many households located further than 25 miles from Coralville Reservoir may hold non-use values for changes in fish population health at the reservoir. Empirical studies in economic literature suggest that individuals are likely to hold non-use values for both local and regional resources.[16,17] Thus, this assumption may provide a conservative estimate of the total affected population near Coralville Reservoir.

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[12] Hoosier Times. 2001. "West Boggs Creek: Boom cycle continues at 'tremendous' reservoir. "http://realindy.com/westboggs.htm

[13] Illinois Department of Natural Resources. 1999. "Status of Walleye in Illinois." http://dnr.state.il.us/conservation/fisheries/WALL.htm

[14] U.S. Census Bureau. 2000. United States Census 2000.

[15] U.S. Environmental Protection Agency (EPA). 1996. "USEPA Reach File Version 1.0 (RF1) for the Conterminous United States (CONUS)."

[16] Pate, J. and J. Loomis. 1997. The effect of distance on willingness-to-pay values: A case study of wetlands and salmon in California. *Ecological Economics* 20:199-207.

[17] Schulze, W.D., R.D. Rowe, W.S. Breffle, R.R. Boyce, and G.H. McClelland. 1995. Contingent Valuation of Natural Resource Damages due to Injuries to the Upper Clark Fork River Basin. State of Montana Natural Resource Damage Litigation Program. Prepared by RCG/Hagler Bailly, Boulder, CO.