ABSTRACT
Habitat Equivalency Analysis (HEA) and Resource Equivalency Analysis (REA) each require the Natural Resource Damage Assessment (NRDA) practitioner to assess both the likely behavior of impacted systems or populations given the pollution event, response actions, and restoration. In the past several years, a general consensus regarding the approach and biological assumptions underlying HEA has arisen. While the details vary with the particulars of the spill, cooperative NRDA for Chalk Point, MV Stuyvesant, Lake Barre, and other spills reveal an emerging consensus surrounding the application of HEA to specific components of oil spill assessment.

The same progression and consensus has not been observed with respect to the biological assumptions underlying REA. Assessments conducted for the Chalk Point (NOAA et al. 2002a), the Lake Barre (Penn and Tomasi 2002), North Cape (Sperduto et al. 1999, 2003) the Stuyvesant (CDFG et al. 2004), the Anitra (NJDEP et al. 2004) and the New Carissa (MV New Carissa 2006) oil spills have used REA frameworks that differ from one another. Further, the variation in REA frameworks alone has resulted in NRD assessments that would differ by millions of dollars, even if all assumptions related to initial injury levels, underlying biological parameters, and the functioning of restoration projects were identical.

The purpose of this paper is to fuse the work of Penn and Tomasi (2002) Sperduto et al. (1999, 2003), NOAA et al. (2002a), CDFG et al. (2004) Zafonti and Hampton (2005) and others with basic ecological theory to refine and extend current practices resulting in a more standardized approach to REA.

For spills in which chronic, long term exposure is not an issue, we apply basic ecological approaches that consider a species ability to respond to environmental perturbation based on (1) changes to the proportion of adults attempting to breed, (2) changes in productivity (defined as fledglings per nesting pair), and (3) changes to annual survival rates. We show that, these factors, when integrated with a simple set of population dynamic equations, provide NRDA practitioners a common, easily understood framework upon which they can assess damages and the necessary scale of potential restoration options.

INTRODUCTION
The Oil Pollution act of 1990 (OPA), the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) and a host of state regulations allow Trustee agencies to seek compensation for injuries to natural resources that occur as the result of oil spills and other pollution events. Under OPA and CERCLA, compensation may take the form of a payment sufficient to offset the public’s lost use values and passive use values which may include option and existence values (56 Federal Register 19760 (1991)). Alternatively, compensation may take the form of an environmental restoration project that is of sufficient magnitude such that the public experiences no net loss in the provision of environmental services (NOAA 1997). Economic theory suggests that the socially efficient compensation package will rely on the least costly approach.

The field of Natural Resource Damage Assessment (NRDA) and literature describing the process of resource injury quantification and compensation are direct outgrowths of those regulations. That literature reveals that, in the early and mid-1990’s, agencies typically sought monetary compensation based on lost natural resource values. This required market observations to estimate lost use values and relied on Contingent Valuation (CV) surveys to estimate potential lost non-use values (Zafonti and Hampton 2005). By the late 1990’s, the guidance in NOAA’s damage assessment rule for oil spills combined with the relatively high cost of the CV tool, its practical limitations, and some theoretically troubling results, caused some agencies to pursue ecological restoration claims based on restoration cost as the primary form of NRD compensation (Flores and Thatcher 2002 and Thompson 2002).

Unsworth and Bishop (1994) published the seminal work describing methods that facilitate a comparison of the injury caused by a pollution event relative to the benefit generated by an environmental restoration project. Using their basic methods, NRDA practitioners have sought to determine the size of environmental restoration projects that, when implemented, would compensate the public for specific pollution events (i.e., Penn and Tomasi 2002, Strange et al. 2002, Sperduto et al. 2003, and Donlan et al. 2003). The process of identifying and sizing a restoration project such that the public is fully compensated is often referred to as resource to resource scaling².

Using the basic construct outlined by Unsworth and Bishop (1994), identifying the proper size of a restoration project is a three step project. First, the level of ecological services that would have flowed had there not been a pollution event (baseline) is identified. Second, the level of ecological services flowing from the

2 In addition to providing a cost-based restoration claim, resource to resource scaling has the added benefit of providing a restoration plan along with the claim amount. In contrast, value-based methods require the trustees to decide what to do with any money recovered as a separate step.
environment given the pollution event and a restoration project is identified. Finally, the NRDA practitioner compares the two service flows (Figure 1). While service levels are below the baseline, a debit, measured in service years, accumulates. When ecological service provision exceeds the baseline, a credit, also measured in service years, accumulates. The public is fully compensated (i.e., the proper size of the restoration project has been identified) when the discounted service years, accumulates.

When ecological service provision exceeds the baseline, a credit, also measured in service years, accumulates. The public is fully compensated (i.e., the proper size of the restoration project has been identified) when the discounted service years, accumulates. The public is fully compensated (i.e., the proper size of the restoration project has been identified) when the discounted service years, accumulates.

This service to service based approach can take two distinct forms: Habitat Equivalency Analysis (HEA) and Resource Equivalency Analysis (REA). The choice of assessment tool (HEA or REA), is dependant upon the effects of the pollution event, the selected restoration project, the preferences of the NRDA practitioners, and other factors. However, the following observations generally apply.

HEA often focuses on the ecological services provided by an acre of an ecosystem. As such, HEA debits and credits are measured in Discounted Service Acre Years (DSAYs) where one DSAY represents a composite measure of all of the services flowing from one acre of the system under baseline conditions. That is, the DSAY is a composite measure of the water that acre would purify, the plants that would grow there, the bird watching opportunities that would be provided, etc.

HEA can be an appropriate tool if all components of the injured system are believed to be of similar value or are provided (and injured) in fixed ratios. For example, if one percent (%) of a dune system were completely destroyed along with all inhabitants of that area, HEA is readily used to determine the amount of nearby dune acreage that must be created to compensate the public for the environmental injury (Dunford et al. 2004).

Alternatively, REA assesses pollution impacts one service at a time. That is, rather than treating all services flowing from a habitat at a composite level, REA assesses potential impacts to each service separately. For example, HEA may assess impacts to the wetland, where as REA would focus on a species residing in the wetland. As such, REA debits and credits are generally measured in species-specific metrics such as Discounted Common Loon Years or Discounted Terrapin Years where a discounted Common Loon or Terrapin Year represents the services provided by one individual living for one year (NOAA et al. 2002a and CDFG et al. 2004).

REA is more appropriately employed if specific species are disproportionately impacted by a release relative to the impacts to the ecosystem. REA may also be more appropriate if a restoration project disproportionately benefits a specific species or if individual species are believed to be of significantly different values. Under these circumstances, a HEA approach is technically sound and the additional effort associated with conducting multiple services specific REAs and then integrating results into a theoretically sound restoration package may be warranted. For example, an oil spill may contact only 5 % of a wetland but could result in 30 % of the wetland's avian population being exposed to oil. If basic wetland services and avian services are not interchangeable, REA may be preferred.

As described, both HEA and REA require the NRDA practitioner to assess (1) the likely behavior of impacted system/populations had there not been a pollution event (baseline) and (2) the behavior of impacted system/populations given the pollution event and restoration.

While the details vary, in the past several years, work done by NRDA practitioners has facilitated a general consensus regarding the biological assumptions and methods underlying HEA. NOAA (2006) provides an overview and illustrates the method with examples, including applications and conditions appropriate for use. Strange et al. (2002) compares results of several different scenarios in hypothetical HEA calculations for salt marsh restoration.

The same progression has not been observed with respect to the biological assumptions underlying REA and refinement is in development. Assessments conducted for the Chalk Point (NOAA et al. 2002a), the Lake Barre (Penn and Tomasi 2002), North Cape (Sperduto et al. 1999, 2003) the Stuyvesant (CDFG et al. 2004), the Anitra (NIDEP et al. 2004) and the New Carissa (M/V New Carissa 2006) oil spills have used REA frameworks that differ. Further, the variation in REA frameworks alone has resulted in NRD liability estimates that differ even when all assumptions related to initial injury levels, underlying biological parameters, and the functioning of restoration projects are similar.

For example, CDFG et al. (2004) specifies nest site limitation as the mechanism limiting the common and red throated loons (Gavia spp.) along the Pacific Coast. Using a simple dynamic modeling approach, they estimate 7.6 Discounted Loon Years of debit per loon mortality associated with a 1999 M/V Stuyvesant spill in Northern California. M/V New Carissa (2006) used a static approach with an unspecified recovery mechanism to estimate 10.1 Discounted Loon Years of debit per mortality for the 1999 New Carissa oil spill in Coos Bay Oregon (approximately 150 miles away). If the static approach had been applied to the Stuyvesant spill, all else equal, loon liability would have increased by hundreds of thousands of dollars.

Similarly, CDFG et al. (2004) used dynamic modeling to estimate that compensation for Stuyvesant related marbled murrelet (Brachyramphus marmoratus) mortality required the protection of approximately 0.1 marbled murrelet nest sites per mortality. M/V New Carissa (2006) used static assumptions to estimate that each marbled murrelet mortality related to the New Carissa spill required the protection of about 0.2 marbled murrelet nest site per mortality. Had the CDFG dynamic model been applied to the New Carissa spill, liability would have been reduced by approximately millions of dollars all else being equal.

However, we note that it is difficult to compare REAs and the resulting restoration requirements across incidents. It is possible that other considerations not described in the assessments and restoration plans may offset some of the apparent differences. The purpose of our comparisons is strictly to point out that different approaches to REA are employed in apparently similar cases and that the REA approach can significantly influence outcomes.

The larger purpose of this paper then is to fuse the work of Penn and Tomasi (2002), Sperduto (1999, 2003), NOAA et al. (2002a), CDFG et al. (2004), Zafonti and Hampton (2005) and others with basic ecological theory to refine and extend current practices resulting in a more standardized approach to REA.

BACKGROUND

Magnitude of initial impacts

The first step in REA is estimating the magnitude of initial injury. We define the magnitude of initial injury as the number of adults, juveniles, and fledglings that were killed or not created as a result of exposure to pollution or during the response. Several methods have been used to estimate the magnitude of initial impacts. These include the use of data describing carcass collections and search effort (Wakefield and Tomasi 2003, Flint et al. 1999, Ford et al. 1996, Ford 1991a, Ford 1991b, and Page et al. 1990), exposure modeling (Fichera 2003, French McCay and Rowe 2004), and simple application of an assumed multiplier applied to the number of organisms observed to have been exposed (RIDEM et al. 1998).
While each method is unique and each contains considerable uncertainty, they are generally consistent with one another both in theory and, when carefully implemented using all available site specific data, prediction.

Approaches for Estimating Time Paths
The next REA requirement is to estimate the time path along which the populations recover to and eventually exceed baseline. As was the case with initial injury estimation, several methods have been employed. These methods include simple assumption, a static modeling approach (Sperduto et al. 1999, MV New Carissa 2006), and simple dynamic modeling (CDFG et al. 2004). Zafonti and Hampton (2005) have noted that these approaches are often at odds with one another and with basic ecological theory. Further, they can generate significantly different results that, through the NRDA process, frequently translate into liability estimates that can diverge by hundreds of thousands if not millions of dollars for an individual spill.

Each method is briefly described below. The reader is directed to the original reference for a more detailed description.

Simple Assumption
The practitioners in several NRDA’s assumed that avian populations naturally recover to baseline levels within about one year. Under certain circumstances this may be a reasonable assumption. However, the observations reportedly justifying one year recoveries, which include relatively minor magnitude of initial impact (Sperduto et al. 1999, NOAA et al. 2002a, Penn and Tomasi 2002), stochastic (variable or random) environments (Wiens 1995), growing populations (Sperduto et al. 1999), and high intrinsic growth rates (Sperduto et al. 1999), are not sufficient conditions to indicate relatively rapid recovery. In fact, demographic modeling reveals that the rapidly growing populations noted by Sperduto et al. (2003) may preclude rapid recovery.4

Static Arithmetic
Other NRDA’s 5 utilized static arithmetic outlined by Sperduto et al. (1999) to estimate debits. This arithmetic suggests calculating debit as two discrete components. First, the number of mortalities for any one species is multiplied by the average life expectancy of a population member. This calculation is meant to represent direct impacts measured in Bird Years (BYs) to the birds (i.e., birds that experienced direct mortality). Indirect effects are estimated, in BYs, by assuming that none of the offspring from the birds that were directly impacted will ever exist. Debit associated with indirect impacts is a function of the number of fledglings assumed not to exist and their average life expectancy.

Logical extrapolation of the static interpretation implies that offspring of the birds indirectly lost due to the spill should also not exist. The same would be true of their offspring’s offspring and so on of all subsequent generations. This logic implies that (1) populations can never recover from environmental perturbation and (2) each mortality represents a debit that will continue to accumulate for an infinite time period, and accordingly, static interpretation must limit debit calculations to one or two generations.

Unfortunately limiting the static interpretation to one or two generations requires the existence of an unusual mechanism of population regulation. That is, the static interpretation implies some mechanism that prohibits replacement of the relatively small number of individuals that are components of the direct impact calculation. Similarly, the mechanism prohibits replacement of individuals that are a component of the indirect impact. At the same time the mechanism requires complete and instantaneous replacement of all subsequent generations.

In addition, the static interpretation does not provide a means of assessing demographic assumptions. That is, within the static framework, one can not determine if, when combined, assumed survival rates and reproductive parameters are roughly consistent with observations.

Dynamic Modeling and Density Dependence
The NRDA practitioners in CDFG et al. (2004) combined simple dynamic equations with density dependant mechanisms to conduct several REAs for the Stuyvesant NRDA. The dynamic modeling approach has the advantage of providing a framework that facilitates the assessment of compatibility among demographic parameters that are often estimated independently of one another. It is also easily calibrated to be consistent with current observations and intrinsic growth rates. Finally, it provides a framework in which the NRDA practitioner can easily assess the actual effect of the assumed injury and relevant recovery/restoration mechanisms.

Zafonti and Hampton (2005) and CDFG et al. (2004) outline the basic approach in more detail. Essentially, population dynamics are determined by annual survival rates, annual productivity (measured as fledgling produced per breeding female), and the proportion of females that attempt to breed.

For a model that recognizes 3 age classes (fledglings, adults 1 year old, and adults > 1 year old) and that is based on an annual post-breeding census, the number of fledglings present at the time of census is calculated as:

\[ F(t+1) = \frac{F(t)}{S} \times SF \times CP \times PB + \frac{A(t)}{S} \times SA \times CP \times PBA, \]

and the number of adults present as:

\[ A(t+1) = F(t) \times SF + A(t) \times SA, \]

where:

- \( F = \) number of fledglings,
- \( S = \) the inverse of the proportion of birds that are female (typically assumed to be 2),
- \( SF = \) annual survival rate of fledglings,
- \( CP = \) chicks fledged per pair per year,
- \( PB = \) proportion of 1-year old adults breeding,
- \( PBA = \) proportion of birds greater than 1 year old that attempt to breed,
- \( A = \) number of adults,
- \( T = \) time measured in years, and
- \( SA = \) annual survival rate of adults.

Equation (1) represents the production of fledglings in the current year. The first half of the equation represents the production of fledglings by one year-old birds (i.e., surviving fledglings produced the previous year). Note that the previous year’s fledglings, \( F(t) \), survive their first winter (i.e., \( SF \)) before they breed (i.e., \( CP \)), and that only a portion of these one year-olds breed (i.e., \( *PB \)). Similarly, the second half of the equation represents adults alive the previous year that survive the winter (i.e., \( *SA \)) and then breed (i.e., \( *CP \)). Note that PBA represents the proportion of adults greater than one year-olds that attempt to breed.

Equation (2) represents survival of fledglings through their first winter to adulthood, i.e., \( F(t) \times SF \), and survival of adults from one year to the next, i.e., \( A(t) \times SA \), and calculates the total number of individuals that are components of the direct impact calculation.

---

4 This is because an impacted population can not recover to baseline population if the baseline population is growing at the maximum rate that can be sustained by the species.

5 Such as North Cape (RIDEM et al. 1998), Chalk Point (NOAA et al. 2002a), North Pass (NOAA et al. 2002b), Lake Barre (LOSCO et al. 1999), Westchester Spill (LOSCO et al. 2001), 10 oil spills in Southeast Florida (FDEP 2003a), and 7 oil spills in Northeast Florida (FDEP 2003b).
of adults expected to be present at a post-breeding census of the population.

In this simple specification survivorship, the proportion of the population that breed, and fecundity, are independent of population levels. That is, there is no density dependence and therefore no recovery mechanism. As such, the injured population will remain proportionally reduced relative to the baseline population regardless of the magnitude of the initial injury, the stochasticity of the environment, the current population trend, or the populations' intrinsic growth rate.

Hence, while the dynamic modeling approach represents a significant theoretical refinement and a large step toward consistency with the ecological literature, the approach itself does not obviate the need to understand the underlying biology and to make assumptions regarding species specific density dependent mechanisms. Instead, the approach may more accurately be described as a means of investigating the effects of various combinations of demographic parameters and density dependent mechanisms as they relate to the species for which REA is being conducted.

That is, the dynamic modeling approach, combined with an ecological framework that carefully considers a population’s inherent ability to respond to perturbation as well as the biological mechanisms that regulate populations, represent the set of necessary REA inputs.

THE ECOLOGICAL FRAMEWORK

The following framework focuses on two considerations of any REA: the inherent ability of a species to recover from a perturbation, and the biological mechanisms that regulate populations.

The inherent ability of a species to recover from perturbation

Organisms have traditionally been characterized by the relative importance of intrinsic growth rate (r) and carrying capacity (K) in their life cycles (Krebs 1994). Species described as r-selected tend to live in stochastic environments. Survival among r species is generally not density dependent or predictable. These r-selected species generally reach adulthood rapidly, produce many offspring per reproductive cycle, and invest little energy in parental care. This life strategy facilitates significant population growth within one reproductive cycle if environmental conditions are favorable (i.e., these species are characterized by high maximum intrinsic population growth rates). K-selected species tend to live in more stable environments. Survival is density dependent and relatively predictable. These K-selected species generally reach adulthood slowly, produce few offspring per reproductive cycle, and invest considerable energy in parental care. This strategy limits maximum intrinsic population growth rates, but facilitates survival in a resource-limited equilibrium setting (Steams 1992, Reznick et al. 2002, MacArthur and Wilson 1967, and Planka 1970 in Reznick et al. 2002).

The current ecological literature includes a debate regarding the relevance of the r-K concept (Steams 1992, Mueller 1997, and Reznick et al. 2002). It has been suggested that the idea over-simplifies the process of natural selection and does not allow for important exceptions to the general observation (Steams 1992). It has also been noted that the framework is not rich enough to incorporate forces acting on the individual, and therefore, it provides little insight into the mechanisms that regulate populations.

This theoretical debate is directly applicable to NRDA. NRDA practitioners often utilize the r-K concept to predict general outcomes at a population level. Those species that are r-like (reach adulthood rapidly, produce many offspring per reproductive cycle, and invest little energy in parental care) and are thought to be members of populations not growing at their maximum intrinsic growth rate, are often assumed to respond to non-chronic anthropogenic incidents through rapid population increases within one reproductive cycle. K-like species are often assumed to take longer to recover.

The above appears to be a reasonable construct for segregating bird species into two categories: those for which recovery to baseline will likely occur within one year may be a reasonable assumption and those which require additional analysis. However, because the r-K concept does not provide insight into the mechanisms that regulate populations, it may be necessary to look beyond the r-K concept to gain insight into recovery time paths and limitations among r-like species that are increasing at their maximum intrinsic growth rate, as well as K-like species.

The biological mechanisms that regulate populations

Lieske et al. (2000) summarized our understanding of population regulating mechanisms in noting that “The use of self-regulating mechanisms to explain long-term stability has a long history in ecology (Cappuccino 1995) but has been notoriously difficult to demonstrate in practice. Even in taxa that exhibit well-developed territorial behavior, such as birds, detection of self regulating mechanisms is difficult (Murdoch 1994).” However, “the question is not “is it regulated?” but “how is it regulated?” (Cappuccino 1995 quoted in Lieske et al. 2000).

Regardless of the species characteristics (r or K-like species), there are three primary, density dependent mechanisms that regulate a population’s growth rate, response to non-chronic anthropogenic incidents, and response to restoration (i.e. time path to recovery). The mechanisms include: changes in the proportion of females breeding, changes in mortality rates, and changes to the number of fledglings produced per breeding female (fecundity). Each mechanism is described below and accompanied by several examples.

Changes to the proportion of females breeding

The proportion of adult birds attempting to breed in the years following a pollution event may change. The sign and magnitude of this change is based on three factors: (1) the number of birds experiencing direct mortality; (2) sub-lethal impacts may affect pair bonding or fitness such that, among previously exposed adults or juveniles the proportion attempting to breed is reduced; (3) the proportion of non-breeding adults, if any, that are recruited into the breeding population in response to increased per capita nesting/ resource availability.

For example, two years following Exxon Valdez oil spill, surveys of murres (Uria spp.) were conducted to measure the impacts of the spill. Mortality estimates suggested that the spill should decrease attendance at some colonies by 60 to 70%. However, Fry (1993), Boersma et al. (1995) and Erikson (1995) found that the number of murres attending colonies in the years following the spill was similar to pre-spill levels. The authors indicated that, while data to specify the mechanism behind this observation was lacking, the recruitment of non-breeding individuals was a likely explanation.

Recruitment of non-breeding individuals has also been documented in herring gulls (Larus argentatus). Duncan (1978) found that, following attempts to cull gull populations, young adults were recruited into the breeding population earlier than they otherwise would have been. This observation is supported by Coulson et al. (1982), who examined the same herring gull population and showed that the mean age of recruitment into the breeding population had decreased following culling efforts.

Anecdotal observations and data collected in 2003 also suggest that a similar mechanism manifested itself in the common loon (Gavia immer) population following the Bouchard-120 spill. U.S. Fish and Wildlife Service (USFWS) (Personal Communication, V. Varela) indicated that a breeding female was observed on her previous breeding territory in 2003. The female was reported to be oiled and unhealthy. A younger healthier female loon apparently
displaced the reportedly oiled loon and acquired the breeding territory.

Changes to annual survivorship

Changes to the rate of annual survivorship within the population may also occur following spills. If individuals are chronically exposed to low levels of oiling, annual survivorship among exposed individuals may decrease. For example, due to chronic exposure, it is believed that the overwinter female survival rate decreased in harlequin ducks (Histrionicus histrionicus) following the Exxon Valdez Oil spill (Exxon Valdez Oil Spill Trustee Council 2006).

At the same time, annual survivorship among individuals not exposed may increase. This may happen if members of the baseline population would have experienced mortality associated with resource limitation. Following a mortality event, competition for resources is reduced. This reduced competition may result in healthier individuals and an increase in annual survivorship.

Lieske et al. (2000) found a direct relationship between adult survival and breeding density among urban nesting merlins (Falco columbarius). Lieske reports that a halving of density resulted in an approximate doubling of survival rates. Territorial breeding pairs of a passerine species, (great tit [Parus major]) were removed each year for three years from their nesting area, hence, decreasing the population density (Both and Visser 2000). The reduction then caused an increase in territorial sizes of the surviving birds, which was correlated to adult survival.

Duncan (1978) noted that, following the herring gull culling project, survival rates among remaining adults increased as the population decreased. Similarly, Coulson et al. (1982) found that body weight and wing length of breeding herring gulls increased as the population size decreased. These factors are both correlated with increased annual adult survival.

This can also happen if there is some density dependant stressor, such as a change in the population’s food source. For example, Carter and Golightly (2003) describe the El Nino affects to a murre colony and refer to a study that found post fledging survival is often low due to poor feeding conditions or inclement weather (Manuwal and Carter 2001 cited in Carter and Golightly 2003). Alternatively, due to improved nutrition from agricultural food resources, it is thought that snow goose (Chen caerulescens) populations have increased, which can be attributed to the increased survival rate and reproductive rates (and safety of refuges) (MDNR 2006 cited in Erickson and Meegan 2007).

Changes to the number of fledglings produced per breeding pair

A change in the number of offspring produced per breeding pair (fecundity) may occur following spills. Following a spill, if inexperienced adults or adults of reduced fitness are recruited into the breeding population, then fecundity may decrease.

At the same time, if resource competition among breeding adults decreases following a mortality event\(^6\), individuals may be healthier and annual fecundity (defined as fledgling per breeding adult) higher than it would have been under baseline conditions.

Several studies discuss evidence of fecundity increases associated with reduced resource competition. Ferrer and Donazar (1996) discussed a 32-year study of an eagle species. This study revealed an inverse relationship between population size and fecundity. A similar trend was observed by Coulson et al. (1982), who found that mean egg size of the breeding herring gulls (assumed to be a correlate of fecundity) increased as the population size decreased. The same result was replicated experimentally by Rodenhouse et al. (2003) and Both and Visser (2000). In the Rodenhouse study, fecundity among breeding females doubled as researchers removed warblers from a nesting habitat. Both and Visser (2000) manipulated the population of a passerine species by removing and relocating territorial individuals in the population. This manipulation resulted in an increase in territorial size and nesting survival.

Finally, and the largest body of evidence indicating a relationship between population size/density and fecundity, is associated with waterfowl. The USFWS has, for at least 50 years, been analyzing the effect of non-chronic, anthropogenic mortality events (i.e. hunting) on waterfowl. Since 1995, Johnson et al. (2002) have investigated the dynamics of the mallard population using flyway level harvest data, population estimates, and an adaptive modeling approach. They have found that an annual harvest of approximately 1% does not reduce the population to levels below where they would exist if there were no hunting. Instead, each spring breeder produce offspring such that the population exists at the largest size the environment can support.

The specific conclusion of the USFWS adaptive modeling approach is that the existence of, or reproductive success associated with any one individual, has very little to do with the reproductive success of the population. That is because reproductive success is strongly density dependent. In fact, the USFWS now assumes strongly density dependant reproduction when establishing waterfowl bag limits (USFWS 2004).

The Ecological Framework applied to REA

The California Office of Spill Prevention and Response (OSPR) has reported criteria upon which assumptions regarding recovery mechanisms could be assessed. In CDFG et al. (2004), OSPR states that the selected assumptions should reflect (1) the magnitude of the spills initial impact; (2) potential ongoing exposure; and (3) a biologically informed opinion regarding a species’ population dynamics. To that list, we add an understanding of the population’s current growth rate relative to its maximum intrinsic growth rate and, if the population is not increasing at its maximum rate, an informed opinion regarding the ecological constraints currently limiting the population’s growth.

Table 1 reports bird year debits that would be estimated using the dynamic modeling approach under varying assumptions. We use 3 combinations of demographic parameters associated with an r-like avian species, a K-like avian species, and an intermediate species. For illustrative purposes, we evaluate these demographic parameters under 5 assumptions. For consistency, we assume in all cases that restoration would not be initiated until after populations recover to baseline levels\(^7\). We also assume, in all cases, that 25% of the local population has experienced spill related mortality.

1. Population growth rates are not limited by any density dependant mechanism. They are currently growing at their maximum intrinsic growth rate and will continue to do so for the foreseeable future. Under this scenario, debit per mortality is a function of the ratio between the maximum intrinsic growth rate and the economic rate of discount. Assuming a 3% economic rate of discount, debit per mortality will be no less than 33.3 Discounted Bird Years (DBY’s).

2. Populations are in equilibrium and limited by fecundity. When spill related mortality occurs, the ratio of resources to adults increases resulting in an increase in productivity. For discussion purposes only, we rely on data from a recent spill and assume a 50% reduction in birds at a breeding colony can increase productivity by as much as 25%. We also assume this relationship is linear over the relevant range. Specifically, when the population is X% below baseline,

---

\(^6\) Resource competition among breeding adults is likely to decrease following a non-chronic anthropogenic event unless the quantity or quality of prey items is reduced by a greater proportion or non-breeding individuals are recruited into the population.

\(^7\) If restoration is initiated before populations recover to baseline, DBY's of debit will be reduced. The reduction is a function of the effectiveness and timing of the restoration project.
productivity = annual productivity at baseline + (0.5X*annual productivity at baseline).

3. Populations are in equilibrium and limited by adult survival rates. When spill related mortality occurs, the ratio of resources to adults increases resulting in an increase in annual survival rates. It is necessary to specify the relationship between density and annual survival. For discussion purposes only, we rely loosely on Lieske et al. (2000) and assume a reduction of local populations by half would result in a 50% reduction in adult mortality and that this relationship is linear over the relevant range. Specifically, when the population is X% below baseline, adult annual survival = annual adult survival at baseline + (X*Annual mortality at baseline).

4. Populations are in equilibrium and potentially nest site limited. When spill related mortality occurs nest sites become available and are immediately filled by non-breeding adults of average fecundity. This results in a change in the proportion of females that attempt to reproduce and the same number of fledglings that would have been produced when the population is at baseline.

5. Population in equilibrium (and potentially limited by both the proportion of breeding and adult survival). When spill related mortality occurs nest sites become available resulting in an increase in the proportion of females that attempt to reproduce and the ratio of resources to adults increases resulting in an increase in annual survival rates as specified in assumption 3 (above).

DISCUSSION

The dynamic modeling approach (CDFG et al. 2004 and Zafonti and Hampton 2005) represents an extension of methods used in past REAs and a step toward standardization across REAs. However, the approach itself does not obviate the need to understand the underlying biology and to make assumptions regarding species specific density dependant mechanisms. Instead, the approach may more accurately be described as a means of investigating the effects of various combinations of demographic parameters and density dependant mechanisms as they relate to the species for which the REA is being conducted.

For example, if a spill results in the mortality of a species (regardless whether it is an r-like, K-like, or intermediate species), the estimated debit per mortality is dependant upon the factor limiting the population. For example, the modeling presented in Table 1 suggests that for a K-like species, if mortality results in an increase in annual adult survival, debit per mortality would be 8.5 whereas, if the population responds by recruiting adults into the breeding population, the debit per mortality would be 6.3. As such, the NRDA practitioner must focus not only on the population maximum potential growth rate (r-like, K-like, or intermediate), but also on the mechanism that facilitates recovery.

This being the case, the NRDA practitioners might benefit if the thought process illustrated in Figure 2 were conducted early in the NRDA process. First, determine if, before the spill, the population was at its maximum intrinsic growth rate or if it was limited by some non-density dependent mechanism. If so, then population recovery may not begin until restoration is initiated. In contrast, if a density dependant mechanism is believed to have been limiting the population at the time of the spill, then practitioners should consider what density-dependent factor(s) were limiting the population and how the population would respond to a mortality event.

The results in Table 1 could be used as a starting point for NRDA practitioners to approximate bird year debits for a variety of situations via benefits transfer. The dynamic modeling approach was applied to 3 species types (r-like, K-like and interme-

BIography

Jeff Wakefield is a Ph.D. Economist who also holds an advanced degree in Marine Biology with a focus on population dynamics. Since joining ENTRIX he participated in several cooperative oil spill NRDA assessments for which avian impacts were key resources.

Julie Anna McNutt holds an advanced degree in Environmental Biology with a focus on population trends in waterbirds. During her tenure at ENTRIX, she conducted NRDA assessments focusing on avian impacts and restoration planning. This presentation is reflective of her work while she was at ENTRIX; she is currently a contractor (I.M. Systems Group) at NOAA Restoration Center (this work does not represent NOAA views).

REFERENCES


sant%20DARP/darp%20index.htm


FIGURE 1. COMPARISON OF SERVICE FLOWS

Ecological Service Level

Baseline Service Flow

Spill and Restoration

Impact of Pollution event

Time

Service Flow with Spill and Restoration

FIGURE 2. DECISION SCHEMATIC FOR NRDA PRACTITIONER TO DETERMINE SPECIES' POPULATION RESPONSE DUE TO THE SPILL INJURIES


TABLE 1. BIRD YEAR DEBIT PER MORTALITY

<table>
<thead>
<tr>
<th>Species</th>
<th>Demographic Parameters</th>
<th>DBYs of Debit per Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>(1) Unconstrained Growth</td>
</tr>
<tr>
<td></td>
<td></td>
<td>1/2</td>
</tr>
<tr>
<td>r-like</td>
<td>Age at first breeding = 2</td>
<td>&gt;&gt;33</td>
</tr>
<tr>
<td></td>
<td>Fecundity = 0.91</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Survival fledged to age 1 = 60%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Annual Survival Thereafter = 64.7%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Maximum Age = 24</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>2.7</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3.4</td>
</tr>
<tr>
<td></td>
<td>Intermediate</td>
<td>&gt;33</td>
</tr>
<tr>
<td></td>
<td>Age at first breeding = 3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Fecundity = 0.52</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Survival fledged to age 1 = 68%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Annual Survival Thereafter = 76.5%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Maximum Age = 24</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>12.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>5.1</td>
</tr>
<tr>
<td>K-like</td>
<td>Age at first breeding = 5</td>
<td>≤33</td>
</tr>
<tr>
<td></td>
<td>Fecundity = 0.27</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Survival fledged to age 1 = 76%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Annual Survival Thereafter = 88.5%</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Maximum Age = 24</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>16.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>8.5</td>
</tr>
</tbody>
</table>

1. In all cases fecundity has been calibrated to imply a stable population at baseline. Rounding to the second decimal place can place demographic parameters out of equilibrium and affect results.

2. Unconstrained growth implies either survival or fecundity are above the baseline levels reported in the column labeled "demographic parameters".