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Technical Comparison of EPA, BLM and Forest Service Pesticide Risk Assessments, Final Report

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1. Introduction

Pesticide risk assessments tend to cover similar types of information; however, the structure, scope, and methods used in risk assessments may vary substantially, depending both on the intent of the risk assessment as well as the preferences of the government agencies or even the offices within those agencies conducting the risk assessments. For example, pesticide risk assessments prepared for compliance with the National Environmental Policy Act (NEPA) may differ substantially from those prepared under purely regulatory activities such as the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) or the Safe Drinking Water Act (SDWA).

The current white paper reviews and compares pesticide risk assessment methods used in Forest Service risk assessments, which are conducted under NEPA, with pesticides risk assessments conducted by other Federal Agencies under NEPA as well as those conducted by the U.S. EPA under FIFRA. The motivation for this comparison is two-fold.

First, this document is intended to highlight differences between the pesticide risk assessments conducted for the Forest Service and those conducted by U.S. EPA/OPP. The purpose of this comparison is expository rather then judgmental. A comparison of different risk assessment methods is intended to help the Forest Service determine the extent to which they should maintain the current differences in risk assessment methods (because of legitimate differences in the intent of the risk assessments) or attempt to harmonize their methods with those of the U.S. EPA.

The second goal of this document is to note differences between Forest Service risk assessments and risk assessments prepared by or for other government agencies and services under NEPA. Under NEPA, there is substantial latitude in terms of how risk assessment activities are implemented and how risk assessment documents are organized. Consequently, different government agencies have evolved different types of NEPA risk assessments. A comparison of these differences is intended to allow the Forest Service to better assess the feasibility of pooling their resources with other government entities to develop a single risk assessment document that can be used by several different agencies in support of their pesticide program activities. Conveniently, for the sake of comparison, the U.S. Department of Interior, Bureau of Land Management (BLM) recently completed a series of risk assessments in support of the Final Vegetation Treatments Using Herbicides Programmatic Environmental Impact Statement (http://www.blm.gov/wo/st/en/prog/more/veg_eis.html). Since several of the recent risk assessments developed for BLM (i.e., ENSR 2005a) relied on Forest Service risk assessments, and since the (ENSR 2005b) ecological risk assessment on fluridone was used in the recent Forest Service risk assessment on fluridone (SERA 2008a), it makes sense to focus the comparison of Forest Service risk assessments with other NEPA risk assessments on the recent series of risk assessments prepared for BLM.

2. Conceptual Approach

2.1. Conceptual Overview

Risk assessments conducted by the Forest Service differ somewhat from those conducted by the U.S. EPA's Office of Pesticides in terms of the conceptual framework on which the risk assessments are based. Each Forest Service risk assessment contains a human health risk assessment as well as an ecological risk assessment, and each of these risk assessments is divided into four sections: hazard identification, exposure assessment, dose-response assessment, and risk characterization (SERA 2007a). Although the U.S. EPA takes the same general approach as the Forest Service to human health risk assessment, the EPA approach to ecological risk assessments is somewhat different, involving an analysis phase consisting of exposure characterization, effects characterization, and risk characterization, known as a *problem formulation*.

Generally, Forest Service risk assessments are organized according to the recommendations made by the National Research Council of the National Academy of Sciences (NRC 1983). In contrast, the conceptual framework for the ecological risk assessments prepared by the U.S. EPA is based on a general approach first recommended by the Ecological Committee on FIFRA Risk Assessment Methods (ECOFRAM). Since the release of the initial ECOFRAM reports (ECOFRAM 1999a,b), the U.S. EPA/OPP (1998b, 2003a) has refined and modified the recommendations from ECOFRAM. A comparison of the NAS and ECOFRAM approaches is illustrated in Figure 1.

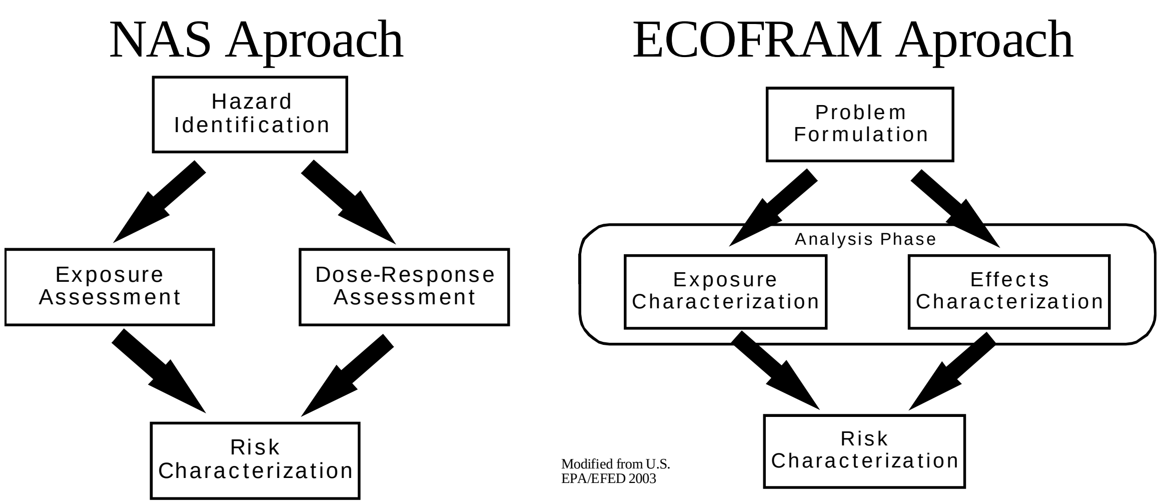

Figure 1: Overview of Risk Assessment Approaches

The most substantial conceptual difference between the NAS and ECOFRAM approaches to risk assessment involves the first step in the risk assessment process. The NAS approach is based on the *hazard identification* and the ECOFRAM approach is based on a *problem formulation*.

In the context of the NAS approach, the hazard identification is the process of identifying what, if any, effects a compound is likely to have on an exposed population. Unless some plausible biological effect can be demonstrated, there may be no need for an exposure assessment, and the nature of any subsequent dose-response assessment and risk characterization is likely to be extremely limited. In Forest Service risk assessments, the hazard identification is used as the first step in both human health and ecological risk assessments. Most guidelines for human risk assessment prepared by the U.S. EPA (e.g., U.S. EPA 1991), also start with a hazard identification.

In the context of the ECOFRAM approach to risk assessment, the problem formulation is quite different from the hazard identification in that the problem formulation is a tool used to define the scope and detail of the risk assessment. The use of the problem formulation and screening level risk assessments is closely related to the concept of *tiered risk assessments*, as discussed further in Section 2.2.

Essentially, the problem formulation functions as a preliminary or “screening level” risk assessment used by risk assessors and risk managers to determine the extent, if any, to which the preliminary risk assessment needs to be expanded or refined to meet the needs of the risk manager. As specified in U.S. EPA/OPP (2004a),

> *The characteristics of an ecological risk assessment are directly determined by agreements reached by risk managers and risk assessors during early planning meetings. … the problem formulation will document, when necessary, any aspects of the analysis that extend beyond the initial screening level risk assessment efforts. The problem formulation will allow for an analysis of any changes in risk estimates based on different assessment assumptions, including those that may be related to proposed mitigation options, and data used for risk analyses.*
>
> – U.S. EPA/OPP 2004a, p. 28

As illustrated in Figure 1, the other differences in organization between Forest Service and EPA risk assessments are, to a large extent, semantic, reflecting little more than differences in terminology. The exposure assessment (NAS approach) and exposure characterizations (ECOFRAM) are conceptually identical. Specific differences in methods for assessing exposure between Forest Service and EPA risk assessments are discussed in further detail in Section 4 of the current report. Similarly, there is very little conceptual difference between the dose-response assessment (NAS approach) and effects characterization (ECOFRAM approach). Again, however, different methods are used in Forest Service and EPA risk assessments to conduct dose-response assessments. These differences are discussed further in Section 5 of the current report. Both the NAS and ECOFRAM approaches use the term *risk characterization* to designate the interpretation or conclusions of the risk assessment, which basically involve a combination of the exposure and dose-response data. Nonetheless, differences do exist between Forest Service and EPA methods in how the exposure and dose-response data are combined. These differences are discussed in more detail in Section 6 of the current report.

2.2. Tiered Risk Assessments

The ecological risk assessments prepared by the U.S. EPA are closely linked to the concept of *tiered* risk assessments. The formal development of tiered risk assessments was first proposed in ECOFRAM (1999a,b) as illustrated in Figure 2.

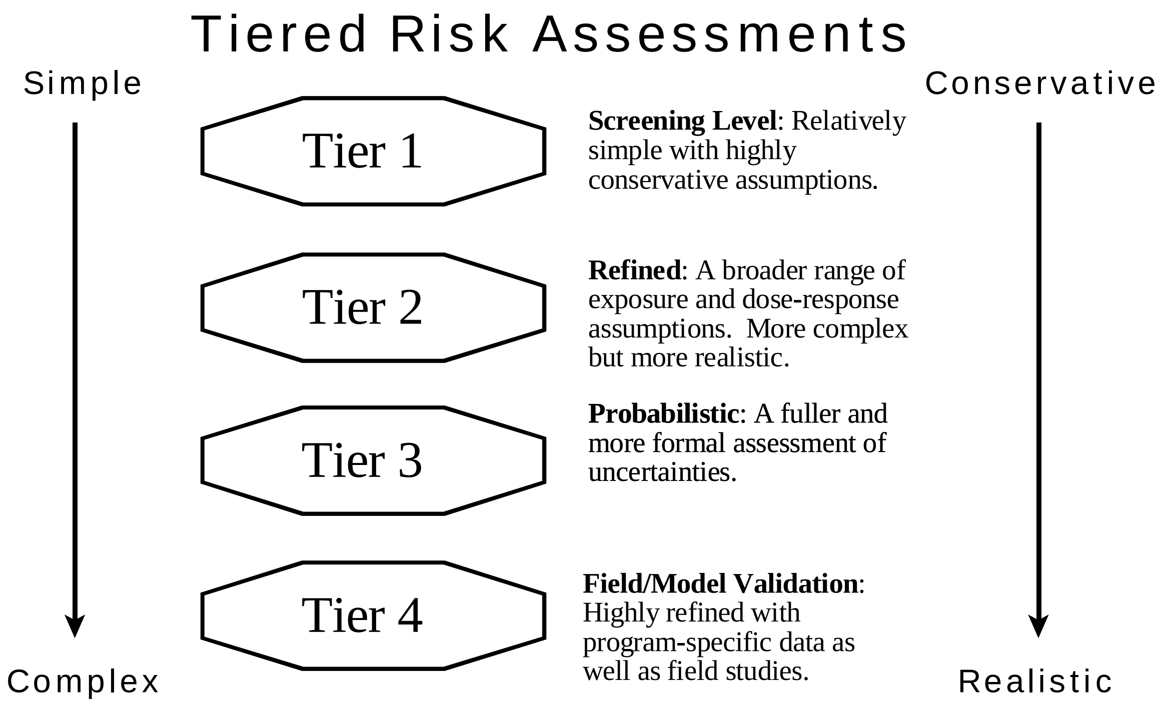

Figure 2: Tiered ecological risk assessments under ECOFRAM

Conceptually, tiered risk assessments are extremely logical and efficient. The first step in the process is a Tier 1 or *screening level* risk assessment. These risk assessments are based on very conservative or protective assumptions analogous to *worst case* scenarios and assumptions used in Forest Service risk assessments.

The U.S. EPA (U.S. EPA 1998b; U.S. EPA/OPP 2004a) has adopted a somewhat less structured approach to tiered ecological risk assessments; nevertheless, as noted in Section 2.1, the basic concept of a tiered risk assessment approach is central to the development of a problem formulation. As discussed in Section 4 (Exposure Assessments) and Section 5 (Dose-Response Assessments), the Tier 1 or screening level risk assessments conducted by U.S. EPA/OPP typically use very conservative exposure assessment models, like GENEEC or SCIGROW, along with very conservative toxicity values (i.e., the most sensitive species). On the whole, screening level risk assessments are relatively simple and require less effort than more refined risk assessments. If the screening level assessment results in a risk characterization that does not suggest a cause for concern, no further work is required. If, on the other hand, the screening level assessment indicates a cause for concern in one or more subgroups of organisms (e.g., mammals, birds, fish, etc.), the exposure assessments and/or dose-response assessments for the subgroup(s) can be expanded or refined to develop an alternative and more realistic risk characterization (i.e., a Tier 2 risk assessment). If the Tier 2 risk assessment also leads to the conclusion that some risks are unacceptable, the risk assessment may be further refined through the use of probabilistic models (Tier 3) or site-specific modeling with field validation studies (Tier 4).

Forest Service risk assessments as well as human health risk assessments prepared by the U.S. EPA do not use a formal tiered risk assessment process. Traditionally, Forest Service risk assessments use exposure scenarios termed *worst-case*, *extreme*, or *accidental* to define the upper bounds of risk, and may also use exposure scenarios termed *expected* or *typical* to define risks that are more likely to occur. As discussed in detail in SERA (2007a), the evolution of this process led to the development of *Extreme Value Risk Assessments*. In these more recent Forest Service risk assessments, most of the values used to estimate risk are not presented as a single number, but are, instead, expressed as a central estimate and a range that is sometimes quite large. The central estimate generally corresponds to the *typical* value, while the upper value in the range corresponds to what used to be called the *"worst-case"* value. The lower bound of the range might be termed the *"best case"* value, suggesting that an unacceptable level of risk from a *best case* scenario is likely to cause adverse effects from exposure to the pesticide under any circumstances. While Forest Service risk assessments do not routinely use probabilistic tools, these methods may be employed in either the dose-response assessment (Section 5) or exposure assessment (Section 6), depending on the available data.

The human health risk assessments prepared by ENSR (2005b) for BLM use the standard NAS approach as do Forest Service risk assessments. For ecological effects, the risk assessment methodology prepared for BLM (ENSR 2004) uses the ECOFRAM approach. Nevertheless, in the ecological effects risk assessment for fluridone, ENSR (2005a) modified the ECOFRAM approach by including a summary of toxicity data, along with a summary of the chemical and physical properties, in an initial section preceding the ecological risk assessment. The *Effects Characterization* included in the ENSR (2005a) risk assessment (i.e., Section 4.2.2) includes a reference to a table of toxicity values used in the risk assessment but consists largely of a discussion of the risk characterization criteria used by the U.S. EPA—i.e., the variable levels of concern (LOCs) for RQs. This approach is discussed further in Section 6 (Risk Characterization) of the current report.

3. Date Coverage

In general, risk assessments prepared under NEPA are required to cover a wide body of published and unpublished literature (i.e., the best available science requirement). As discussed further below, the unpublished literature consists primarily of studies submitted to the U.S. EPA to support the registration requirements of the U.S. EPA's Offices of Pesticide Programs (U.S. EPA/OPP). Unlike the approach taken by the U.S. EPA, Forest Service risk assessments may give preference to open literature studies, when the studies provide useful information not included in the studies submitted by the registrant.

- There are many commercial databases that can be used to search the published literature. Initially, Forest Service risk assessments are typically based on searches of TOXLINE (including PubMed) and AGRICOLA. These two data bases usually identify most of the relevant published literature. Other supplemental searches may be conducted using other commercial data bases as detailed below. The literature searches are almost always conducted in stages. The initial searches are quite general and unselective, based on the pesticide name and CAS number; however, as the risk assessment proceeds, other more specialized searches are conducted, depending on the topics of greatest concern. In addition to the databases, the bibliographies of publications identified through the literature search are screened for relevant citations. In more recent Forest Service risk assessments, this process is documented in the bibliography of the risk assessment. The start of the bibliography lists the specific stages of each phase of the literature search, and the references associated with each stage are identified by a *set* number—e.g., SET01, SET02, and so on.

The coverage of literature in pesticide risk assessments under FIFRA differs substantially from those prepared under NEPA. Under FIFRA, the U.S. EPA/OPP is required to specify the types of studies required for the registration of pesticides. Similarly, the U.S. EPA's Office of Pollution Prevention and Toxics (OPPT) is mandated to develop test guidelines for industrial chemicals under the Toxic Substances Control Act (TSCA). These two offices have developed harmonized guidelines for testing, and a complete list of the specific types of studies and the protocols for these studies is available at http://www.epa.gov/opptsfrs/home/guidelin.htm.

The harmonized guidelines are organized in the following 10 series:

810 - Product Performance Test Guidelines
830 - Product Properties Test Guidelines
835 - Fate, Transport and Transformation Test Guidelines
840 - Spray Drift Test Guidelines
850 - Ecological Effects Test Guidelines
860 - Residue Chemistry Test Guidelines
870 - Health Effects Test Guidelines
875 - Occupational and Residential Exposure Test Guidelines
880 - Biochemicals Test Guidelines
885 - Microbial Pesticide Test Guidelines

In risk assessments conducted by U.S. EPA/OPP, registrant submitted studies that follow the above guidelines almost always take preference over studies from the open literature. The rationale for this approach appears to be 3-fold: the registrant submitted studies meet the specific guidelines developed by the U.S. EPA, the studies are conducted following Good Laboratory Practices (GLPs), and full copies of the studies, including all raw data, are available to the U.S. EPA for review.

Forest Service risk assessments consider studies submitted to the U.S. EPA/OPP under FIFRA. The studies of greatest significance are those performed under the Health Effects Test Guidelines (Series 870), Ecological Effects Test Guidelines (Series 850) and Fate, Transport and Transformation Test Guidelines (Series 835).

Full Studies	Contain the most detail but include interpretation only by the individuals conducting the study. Typically not available and not subject to FOIA.
DERs	Less detail than full studies but include evaluations of the studies by the U.S. EPA and often include statistical reanalyses by the U.S. EPA.
EPA Summaries	Taken from REDs, Science Chapters, and related documents. Less detail than DERs but include EPA evaluations as well as the uses of the studies by EPA.
One-liners	Very brief summaries of the key information. One-liners are not used in Forest Service or EPA risk assessments.

Figure 3: Consideration of FIFRA Studies

As illustrated in Figure 3, there are various ways in which registrant submitted studies are considered in the process of conducting pesticide risk assessments. Prior to 2000, the U.S. EPA released full copies of FIFRA CBI studies to the Forest Service contractor, and these studies were used in the preparation of Forest Service risk assessments; however, the U.S. EPA discontinued this practice. Sometimes, however, , full studies are available from the pesticide manufactures. Although full studies are sometimes the preferred source of information because they provide the most experimental detail, Forest Service risk assessment are typically based on data evaluation records (DERs) requested either from the pesticide manufacturer or from the U.S. EPA under the Freedom of Information Act (FOIA). DERs are prepared by the U.S. EPA. Each DER involves an independent assessment of the study by the U.S. EPA to ensure that the EPA Guidelines are followed. DERs often include detailed reanalysis of the data provided in the full study. In addition, each DER undergoes internal review (and sometimes several layers of review) by the U.S. EPA. Particularly for DERs that are relatively recent, the DER is often a better source of critical information than the full study. Summaries of registrant submitted studies are also given in documents released by the U.S. EPA. Depending on the pesticide, many different types of documents may be available. The most common types of EPA documents used in Forest Service risk assessments are Registration Eligibility Decision Documents (REDs) and Science Chapters prepared by HED and EFED. Older Forest Service risk assessments as well as some NEPA risk assessments cite *one-liners*.

As the name implies, one-liners are particularly brief summaries of registrant submitted studies. One-liners are no longer used in Forest Service risk assessments.

The U.S. EPA sometimes considers studies from the open literature; however the coverage is usually not as inclusive as it is in Forest Service risk assessments. For human health risk assessments conducted by the U.S. EPA, the approach to the open literature is highly variable and does not appear to follow a consistent pattern. For ecological risk assessments, the U.S. EPA has developed ECOTOX, a database of studies from the open literature that catalogues the open literature on ecological effects that the EPA considers relevant. ECOTOX, which is routinely searched in the preparation of Forest Service risk assessments, is available to the general public at http://cfpub.epa.gov/ecotox/.

4. EXPOSURE ASSESSMENT

4.1. Worker Exposure

A major and substantial difference in risk assessments conducted by the U.S. EPA/OPP and risk assessments conducted for the Forest Service involves the methods used to estimate worker exposure. Two general types of methods can be considered for worker exposure modeling, deposition-based and absorption-based. The U.S. EPA/OPP uses a deposition-based approach, as do NEPA risk assessments prepared for BLM (ENSR 2005a). The Forest Service uses an absorption-based approach. An overview of the two approaches is given in Table 1.

Table 1: Overview of Worker Exposure Methods

Factor	Forest Service	EPA and BLM
General Approach	Absorption based	Deposition based
Database	Worker exposure studies measuring absorbed dose covering all routes of exposure (SERA 1998).	Worker exposure studies on deposition from Pesticide Handlers Exposure Database (PHED Task Force 1995).
Worker groups	Aerial, boom spray, and backpack applicators.	37 different groups are defined in database – e.g., mixing, loading, application, flaggers) for different application methods.
Absorption rates	Not explicitly used – i.e., incorporated into studies in database.	Uses daily absorption rates if available, otherwise a default of 10% for dermal and 100% for inhalation.
Personal Protective Equipment	Chemical specific study or taken from PHED	Taken from PHED
Accidental Exposures	Wearing contaminate gloves and spilling pesticide onto skin	None in EPA assessments. One scenario in BLM assessments.

The U.S. EPA's Office of Pesticide Programs employs a deposition-based approach using data from the Pesticide Handler's Exposure Database (PHED Task Force 1995). In this type of model, the exposure dose is estimated from air concentrations and skin deposition monitoring data. PHED Version 1.1, the version currently used in U.S. EPA/OPP risk assessments, is implemented as both a database and a DOS computer program. U.S. EPA/OPP (1998c) has summarized surrogate exposures from PHED for 37 types of exposures, involving mixer-loaders, flaggers, and applicators, for several different types of formulations (e.g., liquid, granular, and wettable powders) applied with ground or aerial equipment. Using the estimates of deposited dose and concentration of the pesticide in air, the absorbed dose for workers can be calculated if estimates are available on absorption rates for inhalation and dermal exposure. As summarized in U.S. EPA/OPP (1998c), dermal exposures are much greater than exposures associated with inhalation.

The USDA Forest Service generally uses absorption-based models in which the amount of chemical absorbed is estimated from the amount of chemical handled (e.g., USDA/FS 1989a,b,c). The use of absorption-based models rather than deposition-based models is based on two common observations from field studies. First, as discussed in the review by van Hemmen (1992), most studies that attempt to differentiate occupational exposure by route of exposure indicate that dermal exposure is much greater than inhalation exposure for pesticide workers. As noted above, this is consistent with the exposure estimates in PHED. Second, most pesticide exposure studies that monitored both dermal deposition and chemical absorption or some other method of biomonitoring noted a very poor correlation between the two values (e.g., Cowell et al. 1991; Franklin et al. 1981; Lavy et al. 1982).

In USDA Forest Service exposure assessments for workers, the primary goal is to estimate the absorbed dose so that it can be compared with the available information on the dose-response relationships for the chemical of concern, as detailed further in Section 6 (Risk Characterization). Thus, if dermal deposition does not correlate well with the absorbed dose and if the inhalation is not a substantial route of exposure for pesticide workers, the absorption-based approach may be advantageous, compared with the deposition-based approach.

Regardless of whether a deposition- or absorption-based model is used to estimate worker exposure, the general algorithms for estimating worker exposure (*Exp*) are similar and are calculated generally, as the product of the exposure rate (*ExpRate*) and the amount of the pesticide that is handled by the worker (*Amnt*):

Equation 1

$$Exp = Amnt \times ExpRate$$

Typically, the amount of pesticide handled is calculated as the product of the application rate (*ApRt* in lbs/acre) and the number of acres treated per day:

Equation 2

$$Amnt = ApRt_{\frac{lbs}{acre}} \times \frac{Acres}{Day}$$

While this basic algorithm is used in Forest Service, BLM, and EPA risk assessments, the number of acres treated per day for a particular application method differs among Forest Service, BLM, and EPA risk assessments. The values used in Forest Service risk assessments are generally based on estimates from field crews performing typical Forest Service applications (e.g., USDA/FS 1989a,b,c). The values used in BLM risk assessments appear to be program-specific values developed by BLM. The values used in EPA risk assessments vary according to the risk assessment. Generally, these values reflect information from registrants as well as judgments made by the EPA. The EPA's Science Advisory Council for Exposure Policy (ExpoSAC 2001) has proposed standard values for daily acres treated in agriculture. These guidelines are cited in some EPA risk assessments; however, it is not clear that these values are used widely or consistently in U.S. EPA/OPP risk assessments.

The U.S. EPA routinely presents different exposure scenarios for workers using different levels of personal protective equipment (PPE) based on studies in PHED. In EPA risk assessments such as those used to support reregistration of a pesticide, the risks associated with different levels of PPE may be used to set regulatory requirements for the use of PPE during pesticide applications. Forest Service risk assessments will consider PPE only when PPE is required by EPA. This is most often the case with some insecticides. When PPE is considered in Forest Service risk assessments, estimates of the effectiveness of PPE are based on chemical-specific studies, if available. Otherwise, estimates of the effectiveness of PPE are taken from EPA assessments of the specific chemical or are developed from the PHED database.

EPA assessments of occupational exposure to pesticides typically do not consider accidental scenarios. Forest Service risk assessments, on the other hand, always consider four accidental scenarios, two involving wearing contaminated gloves and two involving pesticide spills onto clothing. Similarly, BLM considers one complex accidental exposure scenario involving a spill onto both the clothing and the exposed skin of a worker (i.e., ENSR 2005b, p. 4-9).

4.2. General Public

4.2.1. Receptors

Within the context of risk assessments, the term *receptor* refers generically to an organism that may be exposed to a compound. While this term was first used in ecological risk assessments, it is currently used in human health risk assessments to refer to a subgroup in the human population.

An overview of the general receptor groups used in Forest Service, BLM, and EPA risk assessments is presented in Figure 4.

	FS	BLM	EPA
Adult Male:	Standard and *Subsistence*	Standard and *Native American*	Standard
Adult Female:	Young woman		Young woman
Child:	13.3 kg Child	15 kg Child	Several categories (infant to adolescent) for dietary.

Figure 4: General Receptors Used in FS, BLM, and EPA Risk Assessments

4.2.1.1. Adult Male

The adult male is used as a receptor in Forest Service, BLM, and EPA risk assessments and the characteristics of the adult male are based on the *reference man* defined by ICRP (1992) (i.e., an adult male weighting 70 kg or about 150 pounds). Both Forest Service and BLM risk assessments use two groups of adult males referred to as standard and subsistence in Figure 4. Here the term *standard* refers to a typical member of the general public who might be exposed to a pesticide in the various scenarios discussed below. BLM risk assessments designate subgroups of adult males such as the hiker/hunter, berry picker, and swimmer. In terms of input parameters, however, each of these subgroups is treated as a standard adult male, and differences among the groups relate to the different exposure scenarios, as discussed in Section 4.2.2.

Both Forest Service and BLM risk assessments include a second group, referred to as *subsistence populations* in Forest Service risk assessments and *Native Americans* in BLM risk assessments. For the most part, this second group is used to represent individuals who may have greater exposures to pesticides than in members of the general public (i.e., the *standard* adult male) because they consume greater amounts of certain commodities, such as wild caught fish, or because they are more likely to engage in certain types of activities, such as basket weaving.

4.2.1.2. Child

Forest Service, BLM, and EPA risk assessments also typically include a young child as a receptor for at least some exposure scenarios. In Forest Service and BLM risk assessments, the young child is involved in several different acute exposure scenarios, and only a single set of input values is used for the child. Unlike the *reference man*, there is no single *reference child*, and different groups use somewhat different input values. As noted in Figure 4, the Forest Service risk assessments use a 13.3 kg child (a toddler aged 2 to 3), which is taken from the U.S. EPA's Exposure Factors Handbook (U.S. EPA/ORD

1996). BLM, on the other hand, uses a 15 kg child taken from a U.S. EPA methods document developed under Superfund (U.S. EPA/OSWER 1991). Other body weights for children have been used in other series of risk assessments: the U.S. EPA Office of Drinking Water typically uses a 10 kg child (Ohanian and Cotruvo 1986) and the U.S. EPA (2008) has recently changed the reference body weight for a toddler from 13.3 kg to 13.8 kg.

These body weight differences for children are relatively modest, and the values are reasonably well-documented for children of different ages. No single value is necessarily any better or more reasonable than any other. Nonetheless, using different body weights leads to different values for other attendant input values, like the surface area of the body and the consumption of different commodities. Thus, although using slightly different body weights for children is not likely to alter a risk conclusion, it will lead to somewhat different estimates of exposure. In terms of the cross-use of risk assessments (e.g., the Forest Service risk assessments by BLM and vice versa), the use of different reference values for children (and other groups) will be a source of inconsistency between the numerical estimates of risk.

The U.S. EPA risk assessments use a much more elaborate set of receptors for children. Rather than using a single, rather generic child, the U.S. EPA often presents risk values for a relatively large number of age groups, including several groups of children. Typically, the following general age classifications are used by the U.S. EPA:

- Infants
- Children 1-2 years
- Children 3-5 years
- Children 6-12 years
- Youth 13-19 years
- Adults 20-49 years
- Adults 50+ years
- Females 13-19 years

As discussed further in Section 4.2.3.2 (Contaminated Food), the use of these age categories is closely related to the nature of the dietary assessments used by the U.S. EPA, which are substantially different from those used in Forest Service or BLM risk assessments.

4.2.1.3. Young Woman

Both Forest Service and EPA risk assessments explicitly consider exposures to a young woman. Forest Service risk assessments use a 64 kg woman, with input parameters for the woman taken from various EPA reports (U.S. EPA/ORD 1985, 1992, 1997). As discussed in further detail in Section 4.2.2, the young woman rather than the adult male is used in several acute exposure scenarios in Forest Service risk assessment. The U.S. EPA does not typically use a single young woman. Nonetheless, as summarized above, the EPA typically provides dietary exposure assessments specifically for 13- to 19-year-old females. The concern for the young woman is related to concerns for potential adverse reproductive effects in women of childbearing age. As discussed further in Section 5, many acute RfDs are based on NOAELs for reproductive toxicity, and these RfDs are applied in Forest Service and EPA risk assessments to a receptor that is intended to encompass women of childbearing age.

Table 2: Exposure Scenarios for Members of the General Public

Factor or Scenario Type	Forest Service	BLM	EPA	Section with Detailed Discussion
Number of scenarios	Accident= 5 Acute = 7 Chronic = 5	Accident= 40 Acute = 30 Chronic, Variable	Highly variable depending on uses.	Section 4.2.2.
Food	Treated vegetation	Treated vegetation	Market basket residues	Section 4.2.3.1.
Water	Gleams-Driver for EECs and spill scenario.	GLEAMS for EECs and spill scenario.	GENEEC (Tier 1) PRZM/EXAMS (higher tiers). No spill scenario.	Section 4.2.3.2.
Fish	All assessments based on concentrations in water and bioconcentration factor in fish.			Section 4.2.3.3.
Swimming	Dermal only	Dermal and oral	Several components	Section 4.2.3.4.
Direct Spray, Dermal	Woman and child	Many	None	Section 4.2.3.5.
Contaminated Vegetation, Dermal	Young woman	All receptors	Occasional	Section 4.2.3.6.
Other	Various			

BLM risk assessments do not explicitly consider women. Women are mentioned in the exposure methodology for the recent BLM risk assessment (ENSR 2005b, pp. 4-13 to 4-14); however, the input values are identical to those used for men (e.g., a body weight of 70 kg). In other words, it appears the BLM risk assessments use the standard 70 kg man to generically represent an adult of either sex. As with the young child discussed in Section 4.2.1.2, the small differences in body weights (70 kg vs 64 kg) and associated values will lead to small differences in exposure estimates and risk values. Because of the nonlinear relationships of body weight to surface area and consumption values, a smaller value for body weight will generally lead to somewhat greater estimates of dose per unit of body weight.

4.2.2. Exposure Scenarios

The specific exposure scenarios included in Forest Service, BLM, and EPA risk assessments are highly varied. As summarized in Table 2, all three types of risk assessments commonly include scenarios for the consumption of contaminated food, the consumption of contaminated water and fish taken from contaminated water, as well exposures associated with swimming in contaminated water. In addition to these common general sets of exposure scenarios, Forest Service and BLM risk assessments typically include various scenarios in which a member of the general public is directly sprayed with a pesticide as well as exposure scenarios for dermal absorption of the pesticide based on dermal contact with contaminated vegetation. Neither of these types of scenarios is typically included in EPA risk assessments.

The comparison of exposure scenarios focuses on the scenarios that are most often used. As noted in Table 2, risk assessments on some pesticides may include other types of exposure scenarios unique to a particular pesticide. For example, the Forest Service risk assessment on DDVP includes a scenario involving a child mishandling a test strip that contains DDVP. These types of atypical exposure scenarios are not further considered in this risk comparison.

Each of the general groups of scenarios summarized in Table 2 differ, and sometimes differ substantially, among the risk assessments conducted by Forest Service, BLM, and the EPA, depending on how and what specific exposure models are used as well as differences in the receptor groups to which the models are applied. As noted in the last column in Table 2, these specific differences are discussed in Sections 4.2.3.1 through 4.2.3.6. The remainder of this section discusses the more general differences between Forest Service, BLM, and EPA risk assessments.

As noted in Table 2, a striking difference between Forest Service and BLM exposure assessments involves the number of standard exposure assessments used in each risk assessment. As summarized in Table 3, Forest Service risk assessments typically present 17 exposure scenarios consisting of five accidental, seven routine, and five chronic scenarios. The list of exposure scenarios presented in Table 3 is adapted from Worksheet E02 of the worksheets developed for Forest Service risk assessments.

Table 3: Standard exposure scenarios used in Forest Service HHRAs

Receptor	Scenario	Source
Accidental Scenarios		
Child	Dermal deposition (whole body)	Direct spray
Adult female	Dermal deposition (lower legs)	Direct spray
Child	Water consumption	Pond Spill
Adult male	Fish consumption	Pond spill
Subsistence male	Fish consumption	Pond spill
Routine Acute Scenarios		
Adult female	Vegetation contact	Direct spray
Adult female	Contaminated fruit	Direct spray
Adult female	Contaminated vegetation	Direct spray
Adult female	Swimming	Peak 1-day EEC
Child	Water consumption	Peak 1-day EEC
Adult male	Fish consumption	Peak 1-day EEC
Subsistence male	Fish consumption	Peak 1-day EEC
Routine Chronic Scenarios		
Adult female	Contaminated fruit	Direct spray, degradation
Adult female	Contaminated vegetation	Direct spray, degradation
Adult male	Water consumption	1-year EEC
Adult male	Fish consumption	1-year EEC
Subsistence male	Fish consumption	1-year EEC

BLM uses numerous exposure scenarios and handles chronic exposure scenarios differently from Forest Service risk assessments. As summarized in Table 2 and detailed further in Table 4, BLM risk assessments use 40 acute accidental scenarios and 30 acute routine scenarios. Note that the list of scenarios in Table 4 is taken from Table 4-4 in ENSR (2005b). To some extent, the differences in the number of specific scenarios are overstated, because not all scenarios for all receptors are used for each of the chemicals covered in ENSR (2005b). Nonetheless, while the general classes of exposure scenarios used in Forest Service and BLM risk assessments are similar, BLM generally uses a greater number of acute exposure scenarios, relative to the number used in Forest Service risk assessments, owing to the tendency of BLM to apply each of the general exposure scenarios to each of the receptors.

A more important and substantial difference between the acute exposure scenarios used in Forest Service risk assessments and those in BLM risk assessments involves the classification of scenarios as *accidental* versus *routine*. BLM risk assessments classify exposures based on direct spray of a pond or consumable (e.g., fruit) as an accidental scenario. In Forest Service risk assessments, the routine (i.e., non-accidental) exposure scenarios involving contaminated vegetation are based on a direct spray of the vegetation. In other words, BLM risk assessments seem to assume that individuals are not expected to come into contact with contaminated vegetation.

Note that the summary of BLM exposure assessments provided in Table 4 covers only acute exposures. BLM risk assessments appear to classify all long-term exposure scenarios as unlikely. As stated in the recent human health risk assessment prepared for BLM:

> *While it is possible that public receptors use public lands under intermediate- and long-term time frames, it is unlikely that public receptors would be exposed to herbicides under the routine use scenario for more than a short-term exposure, which is defined as 1 day to 1 month (USEPA 2001h). Therefore, short-term dose-response values are used to evaluate the public receptors under the routine use exposure scenario. To account for the unlikely possibility that public receptors could repeatedly enter areas that have been recently sprayed, the Uncertainty Analysis (Section 5.5) includes an evaluation of the public receptors under an intermediate and a long-term exposure scenario.*
>
> ENSR 2005a, p. 4-10

The uncertainty analysis referenced above is included in Section 5.5.6.1 of ENSR (2005a) and the estimates of longer-term risk are included in appendices. As discussed further in the following subsections, the nature of the longer-term exposure scenarios are generally similar to those used in Forest Service risk assessments but differ in terms of specific exposure inputs and assumptions. These specifics differences are discussed in the following subsections.

Table 4: Standard acute exposure scenarios used in BLM HHRAs

Modified from Table 4-4 in ENSR (2005b)

Receptor	Scenario	Modifier	Source
Hiker	Dermal Contact (spray)	Routine	Spray drift
		Accidental	Direct spray
	Dermal Contact (veg)	Routine	Spray drift
		Accidental	Direct spray
	Water ingestion	Routine	Spray drift
		Accidental	Direct spray
		Accidental	Spill
Berry picker *Adult and child*	Dermal Contact (spray)	Routine	Spray drift
		Accidental	Direct spray
	Dermal Contact (veg)	Routine	Spray drift
		Accidental	Direct spray
	Water ingestion	Routine	Spray drift
		Accidental	Direct spray
		Accidental	Spill
Angler	Dermal Contact (spray)	Routine	Spray drift
		Accidental	Direct spray
	Dermal Contact (veg)	Routine	Spray drift
		Accidental	Direct spray
	Water ingestion	Routine	Spray drift
		Accidental	Direct spray
		Accidental	Spill
Swimmer *Adult and child*	Dermal contact (water) and ingestion of water	Routine	Spray drift
		Accidental	Direct spray
		Accidental	Spill
Resident *Adult and child*	Dermal Contact (spray)	Routine	Spray drift
		Accidental	Direct spray
	Dermal Contact (veg)	Routine	Spray drift
		Accidental	Direct spray
	Ingestion (berries)	Routine	Spray drift
		Accidental	Direct spray
Native American *Adult and Child*	Dermal Contact (spray)	Routine	Spray drift
		Accidental	Direct spray
	Dermal Contact (veg)	Routine	Spray drift
		Accidental	Direct spray
	Swimming: Dermal contact (water) and ingestion of water	Routine	Spray drift
		Accidental	Direct spray
		Accidental	Spill
	Ingestion (berries)	Routine	Spray drift
		Accidental	Direct spray
	Fish Consumption	Routine	Spray drift
		Accidental	Direct spray
		Accidental	Spill

The EPA typically uses fewer exposure scenarios than either Forest Service or BLM risk assessments. Specifically, the EPA does not include accidental exposure assessments similar to those included in either Forest Service or BLM risk assessments. The EPA exposure assessments generally focus on dietary and drinking water exposures. As detailed in Section 4.2.3.1, the dietary exposure assessments are very different from those used in NEPA risk assessments in that the exposure assessments are based on the use of pesticides on crops. Neither the Forest Service nor BLM risk assessments cover pesticide applications to crops. The drinking water assessments used by the EPA are conceptually similar to those used by the Forest Service and BLM but are based on different exposure models and treatment assumptions (Section 4.2.3.2). Some EPA risk assessments do include exposures associated with swimming; however, these assessments are based on a somewhat elaborate model developed by EPA (Section 4.2.3.4). In addition, some EPA pesticide risk assessments do include scenarios associated with uptake from contaminated turf or other contaminated surfaces; however, these assessments are generally limited to insecticides rather than all pesticides.

The most remarkable difference between the FIFRA risk assessments prepared by EPA and the NEPA risk assessments prepared by Forest Service or BLM involves the number of different uses that are considered. Typically, Forest Service and BLM risk assessments focus on relatively few non-agricultural uses. The EPA risk assessments, however, must cover a broad range of uses that include and are typically dominated by crop applications. Thus, while EPA risk assessments will generally not use as broad a range of exposure scenarios as Forest Service or BLM risk assessments, the EPA risk assessments will often involve a larger number of specific exposure assessments, based on the large number of crops and associated application rates that must be considered.

4.2.3. Types of Exposure Scenarios

4.2.3.1. Food

Although both the Forest Service and BLM risk assessments develop exposure assessments for the ingestion of contaminated fruits, they use different methods to estimate dose. In the acute and longer-term exposure scenarios used in Forest Service risk assessments, the concentration of the chemical on contaminated vegetation is typically estimated using the empirical relationships between the application rate and the concentration on vegetation, developed by Fletcher et al. (1994). The concentration on vegetation is then used along with assumptions about food consumption rates to estimate a daily dose (SERA 2007a, Section 3.2.3.6). In addition to residues on fruit, Forest Service risk assessments also consider the consumption of contaminated broadleaf vegetation because the concentration rates developed by Fletcher et al. (1994) indicate that residues on broadleaf vegetation are likely to be higher than those on contaminated fruit. For each exposure scenario, the estimated doses are expressed as a central value along with upper and lower bounds of exposure.

The BLM risk assessment use a different algorithm based on an EPA method for assessing toddler ingestion of pesticide treated grass (ENSR 2005a, Section 5.2.2.6, p, 5-11 ff). Although the algorithms used by BLM and the Forest Service differ, the resulting exposure estimates appear to be quite similar. For example, BLM estimates that chemical exposure for an adult male who consumes berries following pesticide application at a rate of 0.25 lbs/acre (the maximum rate used for dicamba) is 0.00256 mg/kg bw (ENSR 2005c, Accidental Exposures). When the same application rate is used in Forest Service risk assessments to estimate exposure, the central estimate of dose is 0.00294 mg/kg bw, which only modestly exceeds (by a factor of 1.034) the value derived in the BLM risk assessment. The upper bound on the exposure estimate made using the Forest Service methodology, however, is 0.04665 mg/kg bw, which substantially exceeds (by a factor of 15) the central estimate derived by BLM. Upper bounds are provided in Forest Service risk assessments to reflect both the variability in the residue rates reviewed by Fletcher et al. (1994) as well as documented variability in food consumption rates.

As noted above, Fletcher et al. (1994) also include estimates of residues rates on contaminated broadleaf vegetation, and these rates are higher than rates on fruit. Using the Forest Service methodology and an application rate of 0.25 lb/acre, the central estimate of exposure is 0.0405 mg/kg bw and the upper bound estimate is 0.3375 mg/kg bw. These are factors of about 16 and 132 higher than the single oral dose of 0.00256 mg/kg derived using the BLM method. Thus, while BLM provides a larger number of exposure assessments, their methodology (i.e., a point estimate using the EPA algorithm) results in much lower estimates of exposure, compared with upper bound estimates and the broadleaf vegetation scenario used in Forest Service risk assessments.

While the general methods for assessing dietary exposures used in Forest Service and BLM risk assessments are relatively simple to compare, this is not the case with dietary exposures conducted by the EPA, which uses a very different approach to dietary exposure. Several different models have been developed that allow for age-specific assessments of total dietary exposures based on crop residue data, age-specific consumption values, and age-specific body weight. For example, in the EPA risk assessment on carbaryl (U.S. EPA/OPP 2007, p. 7), the U.S. EPA uses the Food Commodity Intake Database (DEEM-FCID™, Version 1.3). As noted in the EPA risk assessment, this database … *incorporates consumption data from USDA's Continuing Surveys of Food Intakes by Individuals (CSFII), 1994-1996 and 1998. The 1994-96, 98 data are based on the reported consumption of more than 20,000 individuals over two non-consecutive survey days*. DEEM or other similar models base the estimates of consumption on residues in each commodity for which the pesticide is labeled; however, the residue levels are based values in commodities as purchased and consumed by the general public.

The dietary estimates used by the EPA are remarkably different from the estimates associated with the direct consumption of treated items like fruits or broadleaf vegetation taken from the field. Thus, the dietary exposures in EPA risk assessments are typically much lower than those used in Forest Service risk assessments. For example, the highest upper bound acute dietary exposure modeled by the EPA for carbaryl is 0.006589 mg/kg bw/day—i.e., 99.9% for children 1-2 years old (U.S. EPA/OPP 2007, Table 5.2.1, p. 30); whereas, in the recent Forest Service risk assessment on carbaryl, the upper bound acute exposure estimate for the consumption of contaminated fruit is 0.14 mg/kg bw and the corresponding value for contaminated broadleaf vegetation is about 1 mg/kg bw (**Carbaryl (Leaf Beetle) Worksheets.xls**). The upper bound Forest Service values are higher than the upper bound EPA values by factors of about 21 and 152. Similar differences are apparent in the longer-term dietary assessments typically included in both EPA and Forest Service risk assessments.

The differences between the EPA and Forest Service dietary exposure assessments reflect differences in the programmatic goals. The EPA is concerned with regulating pesticide use with a strong emphasis on agricultural uses, which are most often the predominant uses for pesticides. Accordingly, the EPA is concerned with consumer exposures to processed commodities. Forest Service risk assessments, on the other hand, are concerned with exposures that could occur following a forestry application. Although most individuals do not forage in forests, some individuals do, and these individuals constitute a group of concern in Forest Service risk assessments.

4.2.3.2. Water Consumption

4.2.3.2.1. Expected Environmental Concentrations (EECs)

The general approach to estimating EECs in surface water is similar in Forest Service, BLM, and EPA risk assessments in that environmental fate models are used to estimate concentrations of the pesticide in surface water. There are differences, however, regarding the use and application of the specific models.

Both the Forest Service and BLM use GLEAMS, a root zone model designed to examine the fate of chemicals in various types of soils under different meteorological and hydrogeological conditions. The GLEAMS model provides estimates of pesticide loss from a treated field in terms of runoff, sediment, and percolation. Both Forest Service and BLM risk assessments then use post-processing algorithms to estimate concentrations in surface water. In the BLM risk assessment, the post-processing algorithms were developed specifically for the recent BLM risk assessments (ENSR 2005a). Forest Service risk assessments currently use Gleams-Driver, a computer program for handling user input and post-processing (SERA 2007b). Gleams-Driver is a standard Windows program. Earlier Forest Service risk assessments used similar algorithms, which were implemented in an XBASE dialect (SERA 2000; SERA 2004).

The BLM risk assessments use GLEAMS to model a small pond and a river. The SERA risk assessments model a small pond and a small stream. BLM uses a static 0.25-acre pond with a 10-acre drainage area that is 1 meter deep. The Forest Service risk assessments typically use a 1-acre pond with a 10-acre drainage area that is 2 meters deep, which is proportional to the "standard farm pond" defined by the U.S. EPA/OPP (i.e., a 1-hectare pond with a 10-hectare drainage area that is 2 meters deep). A static pond volume was also used in early Forest Service risk assessments (SERA 2000) but it has been replaced with a variable volume pond (SERA 2004). Although Gleams-Driver accommodates either a fixed or variable volume pond, the variable volume pond is used in all recent Forest Service risk assessments.

Both BLM and Forest Service risk assessments involve GLEAMS runs in clay, loam, and sand. In GLEAMS, rainfall is the primary mechanism of pesticide transport. Earlier Forest Service risk assessments and BLM risk assessments, as well, conducted several GLEAMS runs with annual rainfall rates of 5-250 inches. The earlier GLEAMS runs in Forest Service risk assessments partitioned rainfall in a pattern of every 10th day. The BLM risk assessments used daily rainfall data from 1990 in Medford, Oregon and scaled the rainfall rates. The more recent applications of GLEAMS in Forest Service risk assessments use Gleams-Driver weather simulations from CLIGEN, a weather generator developed by USDA, for nine locations consisting of combinations of arid, moderate, and heavy rainfall patterns as well as cool, moderate, and warm temperatures.

The EPA uses different models to estimate pesticide concentrations in water, according to the specific types of risk assessments. For Tier 1 assessments, the EPA uses GENEEC (Generic Estimated Environmental Concentrations) (GENEEC 2001), a straightforward model that estimates pesticide concentrations in a small pond. For more refined risk assessments, the EPA uses PRZM/EXAMS (Burns 2006). PRZM (Pesticide Root Zone Model) is another root zone model which was developed and is used by the EPA. EXAMS (Exposure Analysis Modeling System) is essentially a post-processor for PRZM (just as Gleams-Driver is a post-processor for GLEAMS) that uses output from PRZM to estimate pesticide concentrations in surface water. Because of the crop-specific focus of EPA risk assessments, the EPA typically conducts PRZM/EXAMS runs for different combinations of crop and location. For human health risk assessments, the EPA uses an Index Reservoir rather than a farm pond. The index reservoir is 13 acres in surface area with a depth of 9 feet and a drainage area of 427 acres.

The Forest Service risk assessments use 1-day peak concentrations for short-term exposures and 1-year average concentrations for longer-term exposures. These time periods are also used in EPA risk assessments (e.g., U.S. EPA/OPP 2007, p. 27). The BLM risk assessments use the maximum 7-day average concentration from the last year of a 10-year simulation for peak exposures. For intermediate and longer-term exposures, BLM risk assessments use 30-day and 1-year average concentrations.

4.2.3.2.2. Accidental Spill

Both Forest Service and BLM risk assessments use similar accidental spill scenarios. Until recently, Forest Service risk assessments generally assumed a spill of 200 gallons of a field solution into a 0.25-acre pond that is 1 meter deep. More recently, the Forest Service decided to use a variable spill volume with a central estimate of 100 gallons and a range of 20-200 gallons. BLM risk assessments use a 0.25-acre pond that is 1 meter deep. Spill volumes are taken as 200 gallons for spills from a batch truck and 140 gallons for spills from a helicopter (ENSR 2005a, p. 4-12).

4.2.3.3. Fish Consumption

Forest Service, BLM, and EPA risk assessments all handle bioconcentration in about the same way. The concentration in water is multiplied by the bioconcentration factor (BCF) to calculate the concentration of the chemical in fish. Assumptions are then made concerning the amount of fish that an individual might consume, and that is used to calculate the exposure dose. Generally, in Forest Service risk assessments for human health, the BCF for the edible portion of fish is used, if available, rather than the BCF for whole fish. Despite the nearly identical methods used in Forest Service, BLM, and EPA risk assessments to determine the pesticide concentrations in fish, the different methods used to estimate pesticide concentrations in surface water (Section 4.2.3.2.1) lead to different estimations of exposure.

4.2.3.4. Swimming

Up until the recent Forest Service risk assessment on aminopyralid, Forest Service risk assessments did not include exposure scenarios for swimmers. Except for pesticides that are applied directly to water (e.g., U.S. EPA/OPP 2004b), the EPA does not routinely include exposure assessments for swimmers. The recent BLM risk assessments include exposure scenarios for swimmers for all applications.

All exposure assessments for swimmers include a dermal component based on zero-order absorption. Both the EPA model (U.S. EPA/OPP 2003a) and the model used in Forest Service risk assessments (SERA 2007a) are based on the analysis developed by the U.S. EPA Office of Research and Development (U.S. EPA 1992b). The algorithm used in BLM risk assessments is slightly different and is based on an algorithm developed as part of the U.S. EPA's Superfund Guidance (ENSR 2005a, p. 4-57). The differences in the algorithms are inconsequential.

Forest Service risk assessments use only the dermal component to estimate swimmer exposure. BLM risk assessments take the same approach for Native American receptors under the rationale that the pond water is assumed to be used by Native American populations for drinking water … *and any incidental ingestion during swimming is therefore included in the drinking water scenario* (ENSR 2005a, p. 4-15). For members of the general public, however, the BLM risk assessments do incorporate the incidental ingestion of water during swimming (based on Superfund guidance) and assume an incidental ingestion rate of 50 mL/hour.

The EPA developed a much more elaborate model for assessing pesticide exposure for swimmers (U.S. EPA/OPP 2003a). This model includes routes for dermal absorption, ingestion, and inhalation, as well as separate exposure routes for buccal, nasal, and aural exposures. For incidental oral exposures, the EPA model uses lower values than those used in BLM risk assessments (i.e., 12.5 mL/hour for adults and 25 mL/hour for young children).

4.2.3.5. Direct Spray

As noted in Table 2, both Forest Service and BLM risk assessments include direct spray scenarios. The scenarios, however, are structured differently. In Forest Service risk assessments, the direct spray scenarios are considered to be accidental, and scenarios involve the direct spray with a field solution. In one scenario, it is assumed that a young child is completely covered with a field solution. This is a highly extreme scenario intended to serve as an essentially Tier 1 screen. In the other scenario, it is assumed that the lower legs of a young woman are sprayed. Each of these scenarios assumes that some fraction of the applied solution remains on the surface of the skin and is absorbed following first-order kinetics. In both of these scenarios, the amount of pesticide deposited on the skin depends solely on its concentration in the field solution and not on the application rate. In other words, this is not a spray deposition scenario.

The direct spray scenario used in BLM risk assessments is a spray deposition scenario. In other words, the assumption is made that the individual is sprayed at the nominal application rate. Thus, the amount of pesticide deposited on the skin is calculated as the product of the body surface area that is sprayed (e.g., cm^2) and the application rate (e.g., mg/cm^2). As in Forest Service risk assessments, the assumption of first-order absorption is used to estimate the absorbed dose. In BLM risk assessments, the direct spray scenarios (i.e., sprays at the application rate considered in the risk assessment) are treated as accidental scenarios. Deposition due to drift is treated as typical/non-accidental.

4.2.3.7. Contact with Contaminated Vegetation

Dermal exposure scenarios involving contact with contaminated vegetation are routinely included in Forest Service risk assessments as well as BLM risk assessments. Some EPA pesticide risk assessments may include scenarios for contact with contaminated surfaces (e.g., the tops of picnic tables); however, these kinds of scenarios are not routinely included in risk assessments conducted by U.S. EPA/OPP.

The methods used in Forest Service and BLM risk assessments differ substantially, and the nature of these differences is similar to the differences in exposure assessments involving the consumption of contaminated vegetation (Section 4.2.3.1). In Forest Service risk assessments, dermal transfer rates (in units of mg/cm^2 hr) are based on the dislodgeable foliar residue using the algorithm developed in Durkin et al. (1995). The amount transferred to the skin is then calculated as the product of the transfer rate, the exposed surface area for the individual, and the duration of exposure. The absorbed dose is then based on the assumption of first-order absorption. (These calculations are typically detailed in Worksheet D02 of the EXCEL worksheets produced by WorksheetMaker 5.)

The approach taken in BLM risk assessments is conceptually similar but takes a somewhat different approach. As detailed in ENSR (2005a, pp. 5-6 to 5-7), U.S. EPA/OPP (1997) developed an approach based on a transfer coefficient (*Tc*), expressed in units of cm^2/hour. The transfer coefficient can be viewed as an activity-specific clearance rate (i.e., the surface area of vegetation from which the pesticide is removed during a particular kind of activity by a particular receptor). The dose to the individual is then calculated as the product of the transfer rate, the dislodgeable foliar residue, and the duration of the activity.

The bases for the two algorithms are totally independent of one another; nevertheless, the results lead to similar dose estimates, similar to the pattern for central estimates in the consumption of contaminated vegetation discussed in Section 4.2.3.1. Again, using dicamba as an example, the BLM risk assessment for dicamba estimates absorbed doses of about 0.0026 mg/kg bw for all receptors based on a 2-hour exposure period after direct spray at 0.25 lb/acre (ENSR 2005c, Accidental Exposures). Using the same application rate and a 2-hour exposure period for a young woman, the Forest Service methodology results in a central estimate of 0.0014 mg/kg bw with a range of 0.0005-0.0042 mg/kg bw.

The central estimate using the Forest Service methodology is about a factor of 2 below the estimate given in the BLM risk assessment [0.0026 mg/kg bw/day ÷ 0.0014 mg/kg bw = 1.85]. In terms of differences in the methodologies, however, it is worth noting that the BLM risk assessment uses an absorption factor of 0.15 for dicamba (ENSR 2005a, Table 4-16, p. 57). The Forest Service risk assessment, however, uses a first-order dermal absorption rate of 0.0013 $hour^{-1}$ which leads to an absorption fraction of 0.03 over a 24-hour period. If an absorption fraction of 0.15 had been used in the Forest Service risk assessment, the central estimate of the absorbed dose would be 0.007 mg/kg bw. Thus, the differences in the estimates for dicamba are most markedly impacted by differences in the absorption rates rather than differences in the general algorithms.

4.3. Exposure Assessments for Nontarget Species

Ecological risk assessments are somewhat more variable than human health risk assessments because of the larger number of receptors (i.e., groups of nontarget organisms) that can be and often are involved in ecological risk assessments. The discussion given in this section is based on the general approach used in most Forest Service risk assessments (SERA 2007a) and the general methods document for the BLM ecological risk assessments (ENSR 2004) as well as the ecological risk assessments themselves (ENSR 2005g). The discussion of ecological risk assessments prepared by the EPA, focus on the general risk assessment methods used by the Environmental Fate and Effects Division (EFED) of the U.S. EPA Office of Pesticide Programs (U.S. EPA/OPP 2004a). Ecological risk assessments prepared by the EPA can differ substantially, depending on the tier level of the risk assessment (Section 2.2). The EPA's ecological risk assessment of carbaryl (U.S. EPA/OPP 2003b) is used as an example of a relatively detailed (Tier II/III) ecological risk assessment, and the risk assessment on dinotefuran (U.S. EPA/OPP 2006) is used as an example of a less detailed assessment.

Table 5: Standard receptors used quantitatively in ecological risk assessments			
Scenario Type	Forest Service	BLM	EPA
Terrestrial Invertebrates			
Honeybee	0.093 g	0.093 g	Not used quantitatively
Others	Variable		Variable
Mammals			
Small Mammal	20 g	20 g	15, 35, and 1000 g
Large grazing mammal	70 kg	70 kg	
Carnivorous mammal	5 kg	12 kg	
Birds			
Small bird	10 g	80 g	20, 100, and 1000g
Large herbivorous bird	4 kg	3.5 kg	
Large piscivorous bird	NS	5 kg	
Terrestrial Plants			
Variable based on toxicity data	Sensitive and tolerant species: direct spray, drift, runoff, and dust	Typical and RTE Species: direct spray, drift, runoff, and dust	Most sensitive species: direct spray and drift
Aquatic Organisms			
Fish	Sensitive and tolerant species	Warm water and cold water	Variable
Amphibian		Most sensitive species	
Aquatic Invertebrates			
Aquatic macrophytes		Most sensitive species	
Aquatic algae			

Table 5 provides an overview of the receptors typically included in Forest Service, BLM, and EPA risk assessments. For the most part, the receptors used in Forest Service and BLM risk assessments are quite similar. Some differences in the body weights selected for different groups of organism exist, with Forest Service risk assessments using lower body weights for a carnivorous mammal and small bird and BLM using a lower body weight for an herbivorous bird. As discussed below, the body weights are used to calculate both surface area and consumption values. Because of the allometric nature of these relationships, small organisms, relative to larger ones, are subject to higher doses (in terms of mg/kg bw). Thus, differences in the selection of body weights have an impact on dose estimates and associated risk.

Both Forest Service and BLM exposure assessments generally focus on estimating dose in units of mg/kg bw for invertebrates, mammals, and birds. The EPA generally takes a somewhat different approach and estimates risk based on comparisons of dietary concentrations in toxicity studies with expected environmental concentrations. The EPA has developed an EXCEL program that handles these calculations (U.S. EPA/OPP 2008), and the default body weights used EPA risk assessments are comparable to those in Forest Service and BLM risk assessments.

While the EPA considers toxicity data on the honey bee (Section 5), the agency does not typically conduct a quantitative exposure assessment for bees. As noted in a recent EPA risk assessment on dinotefuran,

> *EFED does not conduct a risk analysis for terrestrial invertebrates like the other nontarget organisms (fish, birds, small-mammals, etc.), however, the Agency is concerned about protecting nontarget terrestrial invertebrates. EFED does not usually assess risk to terrestrial invertebrates using RQs. A screening level RQ assessment method for estimating the risk to bees is not available because EFED has not developed an exposure design for bees.*
>
> – U.S. EPA/OPP 2006, p. 4.

In both Forest Service and BLM risk assessments, pesticide exposure levels for terrestrial plants are based on deposition (direct spray or drift) as well as offsite transport of the pesticide by runoff or soil transport by wind (i.e., dust). The EPA considers only deposition by direct spray and drift.

The Forest Service and BLM consider the effect of soil transport by wind, using different methods. As discussed in detail in SERA (2007a, Section 4.2.3.5), Forest Service risk assessments base the estimates of wind erosion by soil on soil loss rates from agricultural lands. As implemented in the Forest Service worksheets, the central estimate of daily pesticide loss, as a fraction of the application rate, is taken as about 6.8×10^{-5} with a range from 1.4×10^{-5} to 1.4×10^{-4}. As discussed in ENSR (2004), BLM risk assessments estimate the transport of contaminated soil by wind erosion using CalPUFF, an air quality dispersion model. Each BLM risk assessment involves nine separate CalPUFF runs based on different geographical locations and transport conditions. When expressed as a fraction of the application, the estimated offsite losses range from about 1.03×10^{-10} to 5.3×10^{-6}. While the values used in Forest Service risk assessments are higher (more conservative) than those used in BLM risk assessments, they seldom have any practical impact on the risk assessment because offsite drift is much greater than the effect of wind erosion of soil. Both Forest Service and BLM risk assessments use AGDRIFT to model exposures to nontarget plants based on drift.

As discussed in Section 4.2.3.1, Forest Service and BLM risk assessments for human health use different approaches to estimate pesticide concentrations in vegetation. In the ecological risk assessment, however, BLM uses the residue rates from Fletcher et al.

(1994), identical to the rates used in both the human health and ecological risk assessments prepared for the Forest Service.

For aquatic species, all exposure assessments are based on modeled concentrations of the pesticide in surface water. In Forest Service and BLM risk assessments, the concentrations in surface water are the same as those used in the human health risk assessment (Section 4.2.3.2.1). As discussed in Section 4.2.3.2.1, the EPA typically uses an Index Reservoir as the basis for drinking water assessments in higher tiered risk assessments. For nontarget species in the ecological risk assessment, the EPA uses a standard farm pond, similar to the ponds used in Forest Service and BLM risk assessments (i.e., the EPA standard pond is a 1-hectare pond with a 10-hectare drainage area that is 2 meters deep).

5. Dose-Response Assessment

5.1. Human Health Risk Assessments

There are very few differences among Forest Service, BLM, and EPA risk assessments in the dose-response assessments for human health effects. The EPA derives acute and chronic RfDs (reference doses). The RfD is calculated as an animal NOAEL or comparable endpoint divided by an uncertainty factor (UF):

Equation 3

$$RfD = \frac{NOAEL}{UF}$$

The selection of the uncertainty factor is relatively standard in all risk assessments across government agencies (SERA 2007a, Table 3-5).

Forest Service risk assessments defer to and use the acute and chronic RfDs derived by the EPA. It is conceivable that a Forest Service risk assessment might derive a surrogate RfD which is lower than the EPA RfD, but only when new data, not considered by the EPA, are available and clearly indicate that the current EPA RfD is not sufficiently protective.

In addition to acute and chronic RfD values, the EPA identifies NOAELs, sometimes referred to as *points of departure*, for oral, dermal, and inhalation exposures for acute, intermediate, and longer-term exposure. These values are typically margin of exposure calculations (MOEs) used in EPA risk characterizations and represent the different approaches taken by the EPA and the Forest Service to characterize risk (i.e., the MOE approach versus the HQ approach). BLM risk assessments follow the MOE approach used by the EPA. As discussed in Section 6 (Risk Characterization), the HQ and MOE approaches are conceptually identical but involve using different algorithms to characterize risk. As also discussed in Section 6, Forest Service use the HQ method in human health risk assessments to maintain consistency with the ecological risk assessment and because the HQ is simpler to apply and understand when applied to multiple routes of exposure and exposures to more than one chemical.

In addition to the use of acute and chronic RfDs, most recent Forest Service risk assessments provide a dose-severity assessment in the dose-response assessment for human health effects. The discussions of dose-severity relationships are designed to assist in assessing the consequences of exceeding the RfD.

5.2. Ecological Risk Assessment

5.2.1. Toxicity Endpoints

As summarized in Table 5, Forest Service, EPA, and BLM risk assessments cover similar groups of nontarget organisms. There are some fundamental differences between the approaches in Forest Service risk assessments and EPA risk assessments. Generally, BLM uses the same dose-response endpoints as the EPA; consequently, this section focuses on the differences between Forest Service and EPA dose-response assessments.

As explained in SERA (2007a, Section 4.3), the Forest Service prefers to use NOEC values rather than LC_{50} or EC_{50} values as the basis for assessing risks associated with acute exposures of nontarget species to pesticides. The U.S. EPA, however, generally prefers to use LC_{50} or EC_{50} values, and that approach that has been adopted by BLM. These differences are very closely related to differences in the risk characterization, as discussed in further detail in Section 6.2.

5.2.2. Studies Selection

Another general difference between Forest Service and U.S. EPA risk assessments involves the selection of studies that form the basis of the dose-response assessment. In general, the EPA uses registrant submitted studies rather than published studies as the basis for a dose-response assessment. Forest Service risk assessments, on the other hand, always consider both registrant submitted studies as well as studies from the open literature, tending to focus on whichever type of study appears to be the most appropriate (i.e., the best) and/or the most conservative. This is a fundamental difference between EPA and Forest Service risk assessments, and the rationale for this difference is not always well understood.

As noted in Section 3, the EPA has defined protocols for the types of studies required for pesticide registration under FIFRA, and chemical companies that submit data for pesticide registration must submit a subgroup of these studies. The EPA specifies the studies required for a particular pesticide, which may involve a tiered approach, as discussed in Section 2.2, in which more detailed studies (e.g., Tier 2) may be required, depending on the results of a preliminary study (Tier 1).

Reviews of the studies submitted to U.S. EPA/OPP typically take the form of a Data Evaluation Record (DER). Although DERs are not developed for all submissions, they are developed for all toxicity studies. Since the formation of the EPA in the 1970s, the nature of scientific reviews of submitted studies has evolved from early DERs entailing only a cursory review, to the current DERs involving an elaborate and detailed review (and often the reanalysis of data) with several stages of internal review by the EPA. Based on these reviews, the EPA classifies the quality of each study. Although the nomenclature is somewhat variable, the general classifications are Core (or Guideline or Acceptable), Supplemental, and Unacceptable (or Invalid). The EPA generally uses studies that are classified as Core/Guideline rather than Supplemental as the basis for the dose-response assessment.

As noted in Section 3, the EPA may consider some studies from the open literature (e.g., studies summarized in ECOTOX). A characteristic of studies from the open literature is that they are highly diverse. Many of these studies are conducted by academics who are attempting to understand a particular toxicological or biological process. In most cases, these published studies do not follow the protocols or guidelines defined by U.S. EPA/OPP. Even if the studies do assert that a particular guideline was followed, published studies do not provide the type of detail required in studies submitted to the EPA under FIFRA. Thus, the EPA seldom classifies a published study as Core/Guideline. Accordingly, published studies are seldom used to derive toxicity values in EPA risk assessments.

Under NEPA, the Forest Service has determined (at least partially through judicial decisions) that it is required to review the open literature. Thus, Forest Service risk assessments generally provide a much more detailed coverage of the open literature than is found in most EPA risk assessments.

The practical impact of the Forest Service use of open literature is highly dependent on the pesticide. Newer pesticides have less open literature (and sometimes virtually no open literature). For these pesticides, Forest Service risk assessments will be based on the same studies used by the EPA, and, in most cases, the studies selected for the dose-response assessment will be the same as those used by the EPA. The major differences in the toxicity values selected for use in the risk assessment will be based on differences in the endpoints that are considered, as discussed in Section 5.2.1. For other pesticides with a robust open literature, Forest Service risk assessments may use data from the open literature either because the literature identifies a more sensitive species than is contained in the FIFRA studies or because the type of study (e.g., invertebrate drift in streams) is relevant to the risk assessment.

5.2.3. Dose Metameter

For mammals and birds, an additional difference will involve the dose metameter (i.e., how the dose is expressed). As discussed in Section 4.3, the EPA uses dietary concentrations (i.e., mg agent/kg diet) rather than ingested doses (i.e., mg/kg bw) as the index of exposure; accordingly, the corresponding toxicity value is expressed as a dietary concentration. Forest Service risk assessments express dose in units of mg/kg bw. These doses are, in turn, calculated from the dietary concentration as well as measured values of food consumption (if available) or estimated values. The basis for this rationale is the potential discrepancy between the amount of food consumed by an organism in the wild and the amount consumed in a laboratory study either because of differences in the caloric value of the foods (e.g., laboratory chow as opposed to vegetation or prey species) or the possible organoleptic effects (unpleasant taste or odor) of laboratory chow, which may decrease the rate of food consumption.

6. Risk Characterization

6.1. Human Health Risk Assessment

The Forest Service, BLM, and the EPA use conceptually identical methods to quantitatively characterize risk in the human health risk assessment, except that the Forest Service uses an algorithm (the hazard quotient approach) that is different from that used by the EPA (the margin of exposure approach). This difference can be a source of confusion and sometimes controversy in terms of preference. The following comparison of the two methods, while mathematically simple if somewhat tedious, is intended to demonstrate the basic relationship of the two methods (i.e. either one is the reciprocal of the other).

Quantitative risk characterizations in Forest Service risk assessments are based on the hazard quotient (HQ) method in both the human health effects and ecological risk assessments. In the risk characterization for the human health risk assessment, the hazard quotient is defined as the ratio of the exposure (*Exp*) to the RfD. As defined in Equation 3 (Section 5.1), the RfD is calculated as the NOAEL divided by an uncertainty factor. Thus,

Equation 4

$$HQ = \frac{Exp}{RfD} = \frac{Exp}{NOAEL \div UF} \ .$$

If the exposure is less than RfD (i.e., the measure of an acceptable exposure), the HQ is less than 1 and there is no plausible basis for asserting that the exposure is likely to result in an adverse effect. If the HQ is greater than 1, there may be a plausible basis for asserting risk; in which case, the types of effects that might be expected are discussed based on the dose-severity assessment included in the Forest Service risk assessment (Section 5.1).

In many Forest Service risk assessments, one exposure pathway (most often contaminated vegetation) will dominate the risk characterization. In some risk assessments, however, different pathways may be combined. Using the HQ approach, the combination of pathways to get a combined HQ value (HQ_C) is very straightforward:

Equation 5

$$HQ_C = \sum_{i=1}^{n} HQ_i = \frac{Exp_1}{RfD_1} + \frac{Exp_2}{RfD_2} + ... + \frac{Exp_n}{RfD_n}$$

where each subscript represents a different exposure pathway. When this equation is used in Forest Service risk assessments, the RfDs in Equation 5 are usually identical because the HQs are based on the same route of exposure. Nonetheless, Equation 5 can be used equally well to develop a combined HQ for different routes of exposures using route-specific RfDs (e.g., oral, dermal, and inhalation). Equation 5 can also be used to combine the effects of different chemicals. When this is done, the combined HQ is typically called the Hazard Index (HI) (e.g., U.S. EPA, 1986).

The U.S. EPA/OPP takes a somewhat different approach based on the margin of exposure (MOE). The terms used to define the MOE approach are somewhat variable—e.g., the *margin of exposure* has also been referred to as the *margin of safety* (MOS). The following discussion is based on U.S. EPA/OPP (2001), and is representative of the approach used by the U.S. EPA/OPP as well as the approach used in BLM risk assessments.

The MOE is defined as the ratio of the animal NOAEL to the estimated exposure:

Equation 6

$$MOE = \frac{NOAEL}{Exp}$$

The MOE is most often evaluated relative to a level of concern (LOC). Most often, the LOC is equivalent to the uncertainty factor that would be used in deriving an RfD. If the MOE is greater than the LOC, no risk is apparent.

Note that the magnitude of the MOE is inversely related to the HQ. As the exposure increases, the HQ increases, but the MOE decreases. The relationship can be stated more precisely by rearranging Equation 4 to solve for the NOAEL,

Equation 7

$$NOAEL = \frac{Exp}{HQ \div UF}$$

and similarly rearranging Equation 6 to solve for the NOAEL:

Equation 8

$$NOAEL = MOE \times Exp\,.$$

Setting the right side of Equation 7 to the right side of Equation 8,

Equation 9

$$MOE \times Exp = \frac{Exp}{HQ \div UF}$$

and dividing both sides by the exposure (*Exp*),

Equation 10

$$MOE = \frac{1}{HQ \div UF} = \frac{UF}{HQ}\,.$$

The relationship can be expressed as a rearrangement of Equation 10 solving for HQ:

Equation 11

$$HQ = \frac{UF}{MOE}\,.$$

While the differences between the simple MOE and the simple HQ are relatively trivial, the EPA almost always derives risk characterizations for combined exposures involving dermal and inhalation pathways in occupational assessments and sometimes derives combined exposures for the general public.

So long as all individual MOE values are based on the same uncertainty factor or LOC, the combined MOE_C can be calculated as:

Equation 12

$$MOE_C = \frac{1}{\frac{1}{MOE_1} + \frac{1}{MOE_2} + \ldots + \frac{1}{MOE_n}}$$

Note that Equation 12 is taken from U.S. EPA/OPP (2001, p. 51) where the EPA uses the term *Total MOE* (MOE_T) rather than combined MOE_C.

In cases where the uncertainty factor for different routes of exposure is not the same, the U.S. EPA/OPP (2001, p. 53) will quantify the risk using an Aggregate Risk Index (ARI):

Equation 13

$$ARI = \frac{1}{\frac{1}{RI_1} + \frac{1}{RI_2} + \ldots + \frac{1}{RI_n}} = \frac{1}{\sum_i^n \frac{1}{RI_i}}$$

where RI_i is the *Risk Index* defined as the MOE divided by the uncertainty factor:

Equation 14

$$RI_i = \frac{MOE_i}{UF_i}.$$

Substituting the right hand side of Equation 11 for MOE_i in Equation 14, RI_i is also equivalent to the reciprocal of HQ_i:

Equation 15

$$RI_i = \frac{UF_i \div HQ_i}{UF_i} = \frac{1}{HQ_i}$$

thus

Equation 16

$$\frac{1}{RI_i} = HQ_i$$

Substituting Equation 16 into the denominator of Equation 13,

Equation 17

$$ARI = \frac{1}{HQ_1 + HQ_2 + \ldots + HQ_n} = \frac{1}{\sum_i^n HQ_i}.$$

then substituting the left hand side of Equation 5 for the denominator in the far right hand side of Equation 17,

Equation 18

$$ARI = \frac{1}{HQ_c}.$$

Thus, the Aggregate Risk Index used by U.S. EPA/OPP is simply the reciprocal of the combined Hazard Quotients. The simplicity of Equation 5 relative to the computations of the ARI (Equation 13 and Equation 14) is apparent.

6.2. Ecological Risk Assessment

The relationship of risk characterization methods in the ecological risk assessments conducted by the Forest Service and U.S. EPA/OPP are the converse of those in the methods used in the human health risk assessments. As noted in the previous section, the risk characterization methods in the human health risk assessment are conceptually identical but use different algorithms. In the ecological risk assessments, the risk characterization methods use identical algorithms; however, there are real and substantial differences in the implementation and conceptual framework of the algorithms.

In Forest Service ecological risk assessments, the HQ is used to characterize risk. Unlike the algorithm used in the human health risk assessment (Equation 4), however, the HQ used in the ecological risk assessment does not incorporate an uncertainty factor and is based only on the ratio of exposure to the toxicity value (*TV*), which is preferably a NOAEL:

Equation 19

$$HQ = \frac{Exp}{TV}.$$

The interpretation of HQ values in the ecological risk assessments is similar to that in the human health risk assessment (i.e., an HQ of 1 or less does not trigger concern but an HQ of greater than 1 suggests that effects might be plausible). The nature of the possible effects, however, is variable. NOAEL values can be based on many different types of endpoints such as mortality, gross signs of toxicity, as well as behavioral, physiological, or biochemical changes. Forest Service risk assessments attempt to select the most relevant and most sensitive endpoint; nonetheless, the endpoint selected for deriving the HQ is often based on limitations in the available data.

In some instances, NOAEL values may not be available for a receptor. In Forest Service risk assessments prepared to date, LD_{50} or LC_{50} values are used to calculate HQs. This practice is being discontinued. In future Forest Service risk assessments, any LD_{50} or LC_{50} used to develop a hazard quotient will be divided by a factor, essentially an uncertainty factor, to approximate an NOEC. The factors that will be used are related to the risk characterization methods used by U.S. EPA/OPP, as discussed in further detail below.

Detailed dose-severity assessments of ecological receptors are seldom provided in Forest Service risk assessments. Nonetheless, the discussions presented in the ecological risk characterization attempt to describe qualitatively the nature of potential adverse effects based on the available toxicity data.

U.S. EPA/OPP bases risk characterizations for the ecological risk assessment on a risk quotient (RQ) (U.S. EPA/OPP 2004), and this method is also used in BLM risk assessments. The algorithm for the RQ is identical to that of the HQ (Equation 19):

Equation 20

$$RQ = \frac{Exp}{TV}.$$

As summarized in Table 6, many RQs are calculated based on LD_{50}, LC_{50} or EC_{50} values rather than the NOECs preferred in Forest Service risk assessments.

As described in U.S. EPA/EFED (2004b), EPA risk assessments conducted by U.S. EPA/OPP are part of the pesticide registration process, and the specific LOCs presented in Table 6 are associated with different *risk presumption* categories. These categories may impact labeling requirements. In addition, if a particular risk assessment results in an HQ that exceeds the LOC, additional analyses may be conducted and may involve elaboration or refinement of the dose-response relationships or exposure assessments.

A major difference between an HQ and an RQ involves the way in which these values are interpreted. As discussed above, HQ values in Forest Service risk assessments are based on a level of concern (LOC) of 1. As noted in Table 6, a variable LOC is used by the EPA for all RQ values based on LD_{50}, LC_{50} or EC_{50} values. For example, if an RQ is based on an LC_{50} or EC_{50} value in aquatic species, the LOC is 0.5 for acute risk, 0.1 for acute restricted use, and 0.05 for endangered species.

Table 6: Risk characterization categories used by U.S. EPA/OPP (2004b)		
Risk Presumption	**RQ**	**LOC**
Mammals and Birds		
Acute Risk	EEC^{b}/LC_{50} or LD_{50}/sqft or LD_{50}/day	0.5
Acute Restricted Use	EEC/LC_{50} or LD_{50}/sqft or LD_{50}/day (or LD_{50} <50 mg/kg)	0.2
Acute Endangered Species	EEC/LC_{50} or LD_{50}/sqft or LD_{50}/day	0.1
Chronic Risk	EEC/NOEC	1
Aquatic Animals		
Acute Risk	EEC^{g}/LC_{50} or EC_{50}	0.5
Acute Restricted Use	EEC/LC_{50} or EC_{50}	0.1
Acute Endangered Species	EEC/LC_{50} or EC_{50}	0.05
Chronic Risk	EEC/NOEC	1
Terrestrial and Semi-aquatic Plants		
Acute Risk	EEC/EC_{25}	1
Acute Endangered Species	EEC/EC_{05} or NOEC	1
Aquatic Plants		
Acute Risk	EEC^{h}/EC_{50}	1
Acute Endangered Species	EEC^{g}/EC_{05} or NOEC	1

As noted above, Forest Service risk assessments used LD_{50} and other similar values in the development of HQs when NOAEL values were not available. This practice has been discontinued. Instead, Forest Service risk assessments will now divide an LD_{50} or similar value by an uncertainty factor if a suitable NOAEL is not available. The uncertainty factors used are 10 for terrestrial species and 20 for aquatic species, both of which are simply the reciprocals of the LOC values used by the EPA (Table 6) for these groups of organisms. As with other HQ values derived in Forest Service risk assessments, the LOC for interpreting the HQ is 1 (i.e., any value above 1 is taken as an indication that risk to the receptor is plausible).

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