DEPARTMENT OF THE NAVY
CRANE DIVISION
NAVAL SURFACE WARFARE CENTER
300 HIGHWAY 361
CRANE INDIANA 47522-5001

IN REPLY REFER TO
5090/S4.7.5
Ser RP3/5205

15 JUN 2005

U.S. Environmental Protection Agency, Region V
Waste, Pesticides, & Toxics Division
Waste Management Branch
Corrective Action Section
77 West Jackson Blvd.
Chicago, IL 60604

Dear Mr. Ramanauskas:

Crane Division, Naval Surface Warfare Center submits, as enclosure (1), responses to your March 23, 2005 comments on the Draft Conceptual Site Model (CSM) and RCRA Facility Investigation (RFI) for the Ammunition Burning Grounds (ABG), Solid Waste Management Unit 03. Enclosure (2) contains the minutes for the meeting held on March 30, 2005 to discuss the March 23, 2005 comments. The permit required Certification Statement is provided as enclosure (3).

If you require any further information, my point of contact is Mr. Thomas J. Brent, Code RP3-TB, at 812-854-6160, email thomas.brent@navy.mil.

Sincerely,

James M. Hunsicker

JAMES M. HUNSICKER
Manager, Environmental Protection
By direction of the Commanding Officer

Enclosures:
1. Responses to ABG CSM and RFI Comments
2. March 30 Meeting Minutes
3. Certification Statement

Copy to:
ADMINISTRATIVE RECORD
SOUTHNAVFACENGCOM (Code ES31) (w/o encl)
IDEM (Doug Griffin)
TTNUS (Ralph Basinski) (w/o encl)

I certify under penalty of law that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to assure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, the information submitted is, to the best of my knowledge and belief, true, accurate, and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations.

James M. Hunsucker
SIGNATURE

Manager, Environmental Protection — 6/15/05
TITLE — DATE

Enclosure (1)

Responses to EPA Comments
on the
Ammunition Burning Grounds
Conceptual Site Model
and
RCRA Facility Investigation

RESPONSE TO ADDITIONAL U.S. EPA COMMENTS
DATED MARCH 23, 2005 ON
THE CONCEPTUAL SITE MODEL (DATED FEBRUARY 2005)
AND RFI REPORT ECO (DATED JANUARY 2005)
FOR SWMU 3 – AMMUNITION BURNING GROUND
NAVAL SURFACE WARFARE CENTER
CRANE, INDIANA

Comments provided by the United States Environmental Protection Agency (U.S. EPA) are shown in bold font. Responses following each comment are shown in regular font. Changes to Conceptual Site Model or the RFI Report text are italicized and enclosed in quotation marks.

RFI Eco:

1) Barium was eliminated as COPC even though the maximum and average concentrations of barium, listed as 981 mg/kg and 262.9733 mg/kg in Appendix H.4, exceeded the Apparent Effects Threshold (AET) of 48 mg/kg. Concentrations greater than an AET always result in toxic effects to the benthic organisms. Barium concentrations in shallow and deep sediments were elevated in every sampling location downstream of the ammunition burning grounds (ABG), including downstream of Spring C where LSC supports a variety of benthic macroinvertebrates and small fish. Therefore, the risks to sediment invertebrates from barium in the sediment may be unacceptable. Although Barium is not considered a bioaccumulative chemical, food chain modeling may provide a line of evidence to drop Barium from further consideration as a COPC. Barium is also elevated at the ABG MTA. The ecological effects of Barium at both OJT and MTA should be evaluated.

The following specific issues were identified in this comment.

Barium was eliminated as COPC even though the maximum and average concentrations of barium, listed as 981 mg/kg and 262.9733 mg/kg in Appendix H.4, exceeded the Apparent Effects Threshold (AET) of 48 mg/kg.

Barium was eliminated as a chemical of potential concern (COPC) for sediments for the following reasons:

- Barium concentrations are greatest in samples collected within the Ammunition Burning Ground (ABG) Main Treatment Area (MTA). Barium concentrations decreased further downngradient although they remained above the Apparent Effects Threshold (AET) and exceeded concentrations in samples collected upgradient of the MTA.
- Portions of Little Sulphur Creek (LSC) upgradient of Spring C are poor habitat for aquatic receptors and significant impacts are unlikely because of the number of receptors is limited by the poor habitat.
- Perennial portions of LSC (Below Spring C) are currently supporting a variety of aquatic receptors indicating impacts from barium are not significant.

Concentrations greater than an AET always result in toxic effects to the benthic organisms.

Although the AET is defined as the concentration above which toxic effects are expected to occur; the value of 48 mg/kg for barium is conservative. The co-author of the briefing report to the "EPA SAB: the apparent effects threshold approach" was contacted in order to better understand the source of the barium AET as well as any uncertainties that may be associated with its use. The original AET values were developed from the Commencement Bay Superfund project in 1984 and later updated with data collected from projects in Puget Sound. The co-author indicated that the AETs are site-specific data and that care should be used when applying at other project locations, in particular, that AETs "should not be considered off-the-shelf numbers". Therefore, although barium concentrations in sediment at SWMU 3 exceed the AET of

48 mg/kg, it does not appear that the AET is the most appropriate value in which to base risk conclusions, particularly because site-specific macro invertebrate samples have been collected in LSC.

In addition, barium concentrations from Wente (1994) were reviewed in order to further evaluate the conservative nature of the barium AET. Wente (1994) identifies background concentrations of 172 potential pollutants across the state of Indiana by county. The report identifies background concentrations as chemical concentrations that are present in sediment in the absence of any particular pollutant source; therefore, the background concentration can be useful in determining whether point sources of pollutions have potentially contributed to a given sediment concentration. The maximum 95th percentile barium concentration for Martin County is 250 mg/kg, which is greater than five times the AET. The AET of 48 mg/kg implies that the naturally occurring barium concentrations throughout the state of Indiana are causing toxic effects to benthic organisms, which is unlikely.

With these considerations and the considerations discussed below, it does not appear that the AET is the most appropriate value for comparing sediment concentrations at SWMU 3 when site-specific data are available (Tetra Tech NUS, Inc., 1999). For these reasons, although barium concentrations are elevated at SWMU 3 with respect to the AET, the AET is a conservative value. Barium in the sediment does not appear to be causing adverse toxicity to aquatic invertebrates and concentrations in SWMU 3 sediments do not warrant retaining barium as a COPC for further evaluation.

Barium concentrations in shallow and deep sediments were elevated in every sampling location downstream of the ammunition burning grounds (ABG), including downstream of Spring C where LSC supports a variety of benthic macro invertebrates and small fish. Therefore, the risks to sediment invertebrates from barium in the sediment may be unacceptable.

Three macro invertebrate samples were collected in LSC along the OJT during the Current Contamination Conditions Risk Assessment (TtNUS, 1999). One sample was co-located with a sediment sample analyzed for barium. The sediment sample is 03SD13 and the macro invertebrate sample is identified as that collected from Spring C, the riffle area. The barium concentration at 03SD13 was 262 mg/kg. The barium concentration at the next upstream sample, 03SD12, was 373 mg/kg. These concentrations are similar to those found in samples collected within this vicinity during the SWMU 3 RFI (TtNUS, 2005). Barium concentrations in shallow sediment (0 to 6 inches) at locations 03SD12 and 03SD13 collected during the SWMU 3 RFI were 353 mg/kg and 326 mg/kg.

Four species were found in the macro invertebrate sample collected from Spring C, the riffle area. Of these, two were EPT (pollution sensitive) species. The Current Contamination Conditions Risk Assessment concluded that the low flow conditions of LSC at the time of the survey contributed to the observed species composition. However, because pollution sensitive species were found in the riffle area, it does not appear that chemicals in the sediment are impacting sediment invertebrates. The barium concentrations have not significantly changed (i.e., increased) since samples were collected for the Current Contamination Conditions Risk Assessment. Therefore, there is no reason to believe that the macro invertebrate community in LSC is now being adversely impacted by barium in the sediment when it was previously concluded that the macro invertebrate community is diverse and pollution sensitive species are represented in this area.

During a site visit in June 2004, it was observed that the creek becomes perennial downstream of Spring C, where the flow of water increases. Halfway downstream to the next bridge, it was noted that the substrate was rocky with gravel with very little sediment. There were a few deeper pools/runs in this portion of the creek that were several feet deep. Various aquatic insects were observed under rocks, and fish, crayfish, and frogs were present in the water; some of the fish were three to five inches long. Also, a few salamanders were observed under the rocks. Therefore, little viable habitat exists for aquatic receptors upstream of Spring C; LSC does

support a variety of aquatic species downstream of Spring C. As such, from further qualitative observations, it does not appear that barium concentrations (or metals concentration in general) are adversely impacting the aquatic community downstream of Spring C where barium concentrations in the sediment samples (03SD15 through 03SD19) were elevated (ranging from 57.3 to 353 mg/kg) when compared to the conservative AET (48 mg/kg).

Although Barium is not considered a bioaccumulative chemical, food chain modeling may provide a line of evidence to drop Barium from further consideration as a COPC. Barium is also elevated at the ABG MTA. The ecological effects of Barium at both OJT and MTA should be evaluated.

It is not clear how food chain modeling would provide evidence to support dropping barium from further consideration as a COPC for aquatic invertebrates. However, barium should be dropped as a COPC based upon the reasons which were previously discussed.

2) There is some additional concern about impacts of metals transport (barium, cadmium, copper, lead, zinc) off-site to Sulphur Creek and the East Fork of the White River where endangered species studies and reintroductions are taking place. Generally, it appears that there is greater contamination from metals downstream of Spring C where more permanent aquatic habitat exists (e.g., lead exceeds both TEC and PEC). Without more extensive sediment sampling data, it is difficult to know the full spatial extent of (e.g. lead) contamination throughout LSC; therefore, the risks to sediment invertebrates from lead may be unacceptable. In addition to sampling data, toxicity information on metals could reduce the level of uncertainty in some cases based on laboratory or field studies. For example, certain COPCs may not be toxic in aquatic environments. This was alluded to very briefly in the case of barium, but no supporting evidence for the statement "barium...is not generally associated with significant toxicity" was provided. Site conditions, such as water hardness, may also affect metals toxicity.

Several issues have been identified in this comment. These are discussed as follows.

Generally, it appears that there is greater contamination from metals downstream of Spring C where more permanent aquatic habitat exists (e.g., lead exceeds both TEC and PEC). Without more extensive sediment sampling data, it is difficult to know the full spatial extent of (e.g. lead) contamination throughout LSC; therefore, the risks to sediment invertebrates from lead may be unacceptable.

After reviewing the data for barium, cadmium, copper, lead, and zinc, it does not appear that the greatest concentrations were detected in samples collected downstream of Spring C. For most of the metals identified, the greatest concentrations are in samples collected upstream of Spring C. For example, the greatest detected concentrations of lead in shallow sediment (i.e., those that were greater than the PEC) were all found in samples upstream of Spring C (see figure 5-8 of the SWMU 3 RFI). Although barium was detected at a concentration of 353 mg/kg at location 03SD15 (which is just downstream of Spring C) in deep sediments, the greatest concentrations of barium in deep sediment were detected upstream of Spring C. For example, barium was detected in deep sediments at concentrations of 981 mg/kg (location 03SD11), 744 mg/kg (location 03SD12), and 419 mg/kg (location 03SD10). Similarly, zinc was detected in deep sediments at a concentration of 319 mg/kg at location 03SD15; however, the greatest concentrations were detected at locations upstream of Spring C at 412 mg/kg, 1,120 mg/kg, and 421 mg/kg (locations 03SD10, 03SD11, and 13SD12, respectively). These are just a few examples; however, this is generally the case for most of the metals.

When comparing concentrations in samples 03SD15 through 03SD19, it appears that the greatest concentrations of metals in these samples are found at the most upstream location (13SD15). For example, copper was detected at 198 mg/kg at location 13SD15 and decreased to less than the consensus-based TEC of 31.6 mg/kg in the furthest downstream samples. In the case of lead, the greatest concentration in these samples was also detected at location 13SD15

with generally decreasing concentrations downstream. In most cases, the concentrations of metals in the furthest downstream sediment samples were less than or just slightly greater than the TEC. Therefore the Navy does not believe that it is necessary to collect additional downstream samples to better define the spatial extent of these metals.

In addition to sampling data, toxicity information on metals could reduce the level of uncertainty in some cases based on laboratory or field studies. For example, certain COPCs may not be toxic in aquatic environments. This was alluded to very briefly in the case of barium, but no supporting evidence for the statement "barium...is not generally associated with significant toxicity" was provided.

Macro invertebrate samples have been previously collected at SWMU 3 as a part of the Current Contamination Conditions Risk Assessment report (see Response to Comment RFI Eco 1). Three macro invertebrate samples were collected in LSC. The sediment location collocated with one of the macro invertebrate samples had metals concentrations greater than or similar to those found in samples collected downstream of Spring C; therefore, the Navy does not believe that additional field studies are needed.

The comment regarding "barium...is not generally associated with significant toxicity" was based on the fact that barium is likely to precipitate out of solution as an insoluble salt in aquatic media and that barium in sediments is typically found in the form of barium sulfate (ATSDR, 1992). Additionally, the insoluble salts are typically less toxic than the soluble barium salts (USEPA, http://www.epa.gov/safewater/dwh/c-ioc/barium.html).

Site conditions, such as water hardness, may also affect metals toxicity.

Water hardness is typically used to adjust water quality criteria for evaluating surface water concentrations for certain metals. It is unclear how water hardness can be used to evaluate metals concentrations in sediment.

ABG/OJT CSM:

1) Provide a reference for the calculation of the barium site-specific soil screening value for protection of groundwater noted on page 1-33.

The site-specific barium screening value for protection of groundwater was calculated for the groundwater MCL of 2 mg/l as described in the following.

U.S. EPA, 2002. Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites.

OSWER 9355.4-24. Office of Solid Waste and Emergency Response, Washington, D.C., December.

The actual screening concentration is 1,648 mg/kg. A copy of the calculation sheet is included below.

Equation B-13
Soil Screening Level Partitioning Equation for Migration to Ground Water

$$\text{Screening Level in Soil (mg/kg)} = C_w \left[K_D + \frac{(\theta_w + \theta_a H')}{\rho_b} \right]$$

Parameter/Definition (units)	Default
C_w/target soil leachate concentration (mg/L)	(nonzero MCLG, MCL, or HBL)[a] × dilution factor
K_d/soil-water partition coefficient (L/kg)	organics = $K_{oc} \times f_{oc}$ inorganics = see Appendix C[b]
K_{oc}/soil organic carbon/water partition coefficient (L/kg)	chemical-specific[c]
f_{oc}/fraction organic carbon in soil (g/g)	0.002 (0.2%)
θ_w/water-filled soil porosity (L_{water}/L_{soil})	0.3
θ_a/air-filled soil porosity (L_{air}/L_{soil})	$n - \theta_w$
ρ_b/dry soil bulk density (kg/L)	1.5
n/soil porosity (L_{pore}/L_{soil})	$1 - (\rho_b/\rho_s)$
ρ_s/soil particle density (kg/L)	2.65
H'/dimensionless Henry's law constant	chemical-specific[c] (assume to be zero for inorganic contaminants except mercury)

[a] Chemical-specific (see Appendix C).
[b] Assume a pH of 6.8 when selecting default K_d values for metals.
[c] See Appendix C.

Input Parameters:

DAF	=	20		
Cw	=	2	mg/L	(U.S. EPA MCL)
		40	mg/L	
Kd	=	4.10E+01	L/kg	
Koc	=	NA	L/kg	
foc	=	NA	g/g	
□w	=	0.3	L/L	
□a	=	0.134	L/L	
Pb	=	1.5	kg/L	
n	=	0.434	L/L	
Ps	=	2.65	kg/L	
H'	=	0.00E+00		

SSL = 40 mg/L x [41 L/kg + (0.3 L/L + 0.134 L/L x 0)/1.5 kg/L|

SSL = 1,648 mg/kg

2) Page 3-2 states that barium was found in significant concentrations only in surface soils. What is the potential of air deposition of barium from OB/OD operations at the ABG/OJT and DBG (barium was elevated in surface soils there as well)? Was this evaluated in the Air Risk Assessment and/or CCCRA for the OB/OD units?

Yes, deposition was estimated in the AEHHRA using the U.S. EPA approved mass-balance approach. This approach was used to estimate the accumulation of barium in soil over time for the Current and Future Land Use Scenarios. The estimated soil concentration for the Current Land Use Scenario is shown in AEHHRA Table 6-3 for a receptor location at the ABG. The estimated barium concentration in soil is 0.0048 mg/kg. The result for the same receptor in the Future Land Use Scenario is given in Table 6-4 from the AEHHRA. The estimated barium concentration in soil is also 0.0048 mg/kg.

All 35 surface soil samples analyzed at the MTA contained barium, which was expected, because barium is naturally occurring in soils. The soils background study (TtNUS 2000) showed that naturally occurring barium concentrations ranged form 46 to 153 mg/kg. Based on the results of the AEHHRA potential air deposition of barium (0.0048 mg/kg) would be insignificant compared to concentrations naturally occurring in soils.

The April 2004 sampling data showed that surface soil barium concentrations ranged from a low of 25.5 mg/kg to a high of 4,120 mg/kg. Subsurface soil barium concentrations ranged from a low of 18.3 to 652 mg/kg. As shown in Figure 1-22 from the Conceptual Site Model, the locations where the highest surface soil barium concentrations were found (2,120 mg/kg at 03SB086, 2,920 mg/kg at 03SB120, and 4,120 mg/kg at 03SB116) were all found in the same general area.

3) The 6th bullet in Section 3.5 makes the statement that RDX is present above RBTLs in LSC surface waters and unknown risk to off-site receptors exists. These risks must be quantified.

As noted in the comments risk to off-site receptors was unknown. Additional evaluations have been conducted to determine risks to off-site receptors. Indiana water quality standards (WQS) for surface waters were used as the basis for evaluating risks to off-site receptors resulting from the presence of RDX in LSC surface waters. Based on this evaluation it was determined that even under worst-case conditions RDX concentrations LSC resulting from Spring A and Spring C discharges do not present any risk to off-site receptors. Following is the detailed evaluation.

Title 327 Water Pollution Control Board establishes regulations applicable to surface waters in the state of Indiana. 327 IAC 2-1-3 (Surface water use designations; multiple uses) Sec. 3(a) states the following:

(1) Surface waters of the state are designated for full-body contact recreation as provided in section 6(d) of this rule.
(2) All waters, except as described in subdivision (5), will be capable of supporting a well balanced, warm water aquatic community and, where natural temperatures will permit, will be capable of supporting put-and-take trout fishing. All waters capable of supporting the natural reproduction of trout as of February 17, 1977, shall be so maintained.
(3) All waters which are used for public or industrial water supply must meet the standards for those uses at the points where the water is withdrawn. This used designating and its corresponding water quality standards are not to be construed as imposing a user restriction on those exercising a desire to exercise the use.
(4) All waters which are used for agricultural purposes must, at a minimum, meet the standards established in section 6(a) of this rule.
(5) All waters in which naturally poor physical characteristics (including lack of sufficient flow), naturally poor chemical quality, or irreversible man-induced conditions, ... Specific waters of the state designated for exceptional use are listed in section 11(a) of this rule.
(6) All waters which provide unusual aquatic habitat, which are an integral .... Specific waters of the state designated for exceptional use are listed in section 11(b) of this rule.

LSC and the East Fork White River between LSC and Shoals are not listed in either 327 IAC 2-1-3 Sections 11(a) or 11(b).

Based on the above, RDX concentrations in LSC must be limited, at a minimum, to concentrations which would ensure maintenance of a balanced warm water aquatic community, and to protect public water supplies, wherever water is withdraw for drinking water or industrial use. These uses require the most stringent criterion.

327 IAC 2-1-6 (Minimum surface water quality standards) states the following in 327 IAC 2-1-6(a)(2):

(2) At all times, all surface waters outside of mixing zones shall be free of substances in concentrations that on the basis of available scientific data are believed to be sufficient to injure, be chronically toxic to or be carcinogenic, mutagenic, or tetratogenic to humans, animals, aquatic life, or plants. To assure protection against the adverse effects identified in this subdivision, the following requirements are established:
(A) A toxic substance or pollutant shall not be present in such waters in concentrations that exceed the most stringent of the following continuous criterion concentrations (CCCs):

> *(i) A chronic aquatic criterion (CAC) to protect aquatic life from chronic toxic effects.*
> *(ii) A terrestrial life cycle safe concentration (TLSC) to protect terrestrial organisms from toxic effects that may result form the consumption of aquatic organisms or water from the waterbody.*
> *(iii) A human health life cycle safe concentration (HLSC) to protect human health from toxic effects that may result from the consumption of aquatic organisms or drinking water from the waterbody.*
> *(iv) For carcinogenic substances, a criterion to protect human health from unacceptable cancer risk of greater than one (1) additional occurrence of cancer per one hundred thousand (100,000) population.*

327 IAC 2-1-6(a)(2)(B) references Table 6-1, which contains a list of surface water quality criteria for specific substances. Criteria are listed for aquatic life (4-day average) and for human health (30-day average) and at the point of water intake (30-day average). RDX is not one of the substances that are listed.

327 IAC 2-1-6(a)(2)(C) references procedures for calculation of criterion for substances which are not listed in Table 6-1. 327 IAC 2-1-8.3 [Determination of chronic aquatic criteria (CAC)] contains procedures for the development of water quality criteria for protection of aquatic receptors. The water quality criterion of 3,070 µg/L was determined by a methodology in accordance with these procedures. A terrestrial life cycle safe concentration (TLSC) was also calculated. The calculated TLSC was 2,800 µg/L. Attachment 1 contains the details.

327 IAC 2-1-6(a)(2)(E) states the following:

(E) The CAC and TLSC for a substance apply in all surface waters outside a mixing zone for a discharge of that substance. Similarly, in waters where a public water system intake is not present or is unaffected by the discharge of a substance, the HLSC and the carcinogenic criterion for that substance based on consumptions of organisms from the waterbody and only incidental ingestion of water shall apply to all surface waters outside the mixing zone for a discharge of that substance. In surface waters where a public water system intake is present, the HLSC and the carcinogenic criterion for a substance based on consumption of organisms and potable water from the waterbody shall apply at the point of the public water system intake.

327 IAC 2-1-8.6 (Determination of concentration providing an acceptable degree of protection to public health for cancer) contains procedures for the development of water quality criteria for protection of public water supply at the point of intake. For public water supply, the water quality criterion, 3.0 µg/L, was determined in accordance with these procedures. For surface waters at

locations where public water supply intakes are not located, the calculated water quality criterion is 240 µg/L. Attachment 2 contains the detailed calculations. Note that a WQS could also be calculated according to 327 IAC 2-1-8-5 for the noncarcinogenic health effects of RDX (on the prostate). However, calculations show that the noncarcinogenic WQS would be approximately 35 times greater than WQS based on cancer. Only the more conservative carcinogenic WQS (3 µg/L) is presented.

In accordance with Indiana requirements, these surface water quality criteria must be met at the location of the point of use. For protection of aquatic receptors, this location would be below Spring C, where LSC becomes a perennial stream. For carcinogenic substances this location would be LSC outside of the location where Springs A and C mix with LSC. For protection of public water supplies, this location would be the closest location, where LSC surface waters are withdrawn for public water supply.

The discharges from Springs A and C can be considered to be equivalent to National Pollution Discharge Elimination System (NPDES) discharge points. Indiana regulations contain provisions for calculation of water quality based effluent limitations. 327 IAC 5-2-11.1 (Establishment of water quality-based effluent limitations for dischargers not discharging to waters within the Great Lakes system) was used to determine the water quality-based effluent limit (WQBEL) for RDX. 327 IAC 5-2-11.1 states the following:

(b)(4) The criterion to provide an acceptable degree of protection for public health for cancer effects shall apply outside of the mixing zone if the criterion is based on the consumption of organisms and incidental water intake and at the point of the public water system intake if based on the consumption of organisms and drinking water, if this would not cause the criterion based on the consumption of organisms and incidental water intake to be exceeded outside of the mixing zone. For calculation of allowable dilution, one-fourth (1/4) of the fiftieth percentile flow of the receiving stream shall be used if the criterion is based on consumption of organisms and incidental water intake, and the fiftieth percentile flow of the receiving stream at the point of the public water system intake can be used if the criterion is based on the consumption of organisms and drinking water.

The surface water features of LSC and the streams into which it flows after it exits NSWC Crane property are shown in Figure 1. LSC flows into Sulphur Creek, which in turn flows into Indian Creek. Indian Creek flows into the East Fork of the White River. IDEM databases were used to determine location of public water supply intakes. No public water supply intakes were identified on Sulphur Creek, or Indian Creek.

United States Geological Survey (USGS) databases were used to obtain stream flow data. The closest stream gauging information, that was available from the USGS, was at the East Fork White River at Shoals. Attachment 3 contains the monthly average flow rate in terms of gallons per minute (gpm) for the last ten years (1993 through 2003). The monthly flow rates at this location ranged from a low of 142,279 gpm (November 1999) to a high of 15,762,951 gpm (May 1996).

All RDX in LSC originates from Spring A (from Main Treatment Area) and Spring C (from Jeep Trail). As noted previously, under base flow conditions, all of the surface waters in LSC are comprised of Springs A and C discharges. Attachment 4 contains information regarding concentrations of RDX in LSC below Springs A and C. The information presented in this attachment demonstrates that RDX concentrations generally decrease with increasing flow. The worst-case combination of RDX concentrations and flow rates was a Spring A RDX concentration of 120 µg/L at an estimated flow rate of 4 gpm. This is well below the aquatic life criterion of 3,070 µg/L and the public health criterion (incidental water intake only) of 240 µg/L. However, it is above the public water supply intake criterion of 3 µg/L which applies at the point of intake.

RDX concentrations at the nearest downstream gauging location (East Fork White River at Shoals), were calculated under the following assumptions:

- Spring C comprises the entire flow of LSC for 30 days
- Spring C RDX concentrations are 63 µg/L for 30 days
- LSC flow rate is 6 gpm for 30 days
- Public water supply intake is located on East Fork White River at Shoals
- Monthly flow rate at Shoals is 142,279 gpm
- No photolytic or biological degradation of RDX occurs.

Under these assumptions, which are more stringent than allowed by IDEM regulations, RDX concentrations at Shoals would be 0.0026 µg/L. This concentration is well below the applicable Indiana RDX criterion for protection of public water supply of 3.0 µg/L.

Based on the above evaluation RDX concentrations in LSC are not presenting significant risks to off-site receptors and are not affecting state-designated uses of the Little Sulphur Creek.

4) The Navy should evaluate risks to human health and ecological receptors in light of the new data obtained in April 2004 at both the ABG MTA and OJT. For example, if the Navy wishes to recommend no removal or treatment of surface or subsurface soils because they do not appear to be a source of ongoing groundwater contamination, the risks to actual/potential receptors of leaving such soils in place must be presented. Even if exposure risks are found to be acceptable and a source remains, EPA prefers source removal unless it can be shown impractical to remove source materials.

The Main Treatment Area (MTA) is an active operating RCRA-permitted hazardous waste treatment facility. All RCRA-permitted facilities are subject to RCRA closure requirements for permitted facilities. At the end of the active life of the MTA, it will either be clean closed or closed as a RCRA landfill.

Human Health Risk Assessment: A screening level health risk assessment is being conducted for current actual/potential receptors that could occur under the existing land use (open burning facility). These receptors include the site worker and trespasser. The results will be included in the CMS and used to determine whether evaluations of corrective measures are necessary to protect current actual/potential receptors. Future receptors (residents, visitors, construction worker) will be not be evaluated at this time.

Ecological Risk Assessment: Currently, and for the foreseeable future, OB/OD operations occur at the active portion of ABG. As seen from Figure 2, the site consists of roads and other areas covered by gravel, pavement, concrete, etc., which is surrounded by mowed grass. The active area is surrounded by a heavily wooded area as seen in Figure 2. With the exception of soil invertebrates, grass and other herbaceous plants, few ecological receptors would be present in the active area except in passing, because of the industrial nature of the site. Therefore, there is not an ecologically significant ecological community that would be impacted by site contaminants under these conditions. After activities cease at the site, a risk-based closure plan would need to be prepared, and risks to ecological receptors will be evaluated at that time for the uses that will be designated in the closure plan.

5) The second bullet on page 3-10 states that remediation of soils to address RDX, TCE, and barium contamination should not be extensively evaluated. Evidence exists that TCE contaminated soils are present at a subsurface depth and may continue to contribute to groundwater contamination. Even if exposure risks to subsurface TCE are found to be acceptable and a source remains, EPA prefers source removal unless it can be shown impractical to remove source materials.

The Main Treatment Area (MTA) is an active operating RCRA-permitted hazardous waste treatment facility. All RCRA-permitted facilities are subject to RCRA closure requirements for permitted facilities. At the end of the active life of the MTA it will either be clean closed or closed as a RCRA landfill.

The TCE contamination in groundwater presents a risk to users of the groundwater. TCE in soils appears to be a continuing source of contamination in groundwater. However, groundwater from the contaminated Beech Creek aquifer, Beech Creek will not be used for as long as the MTA is an active treatment area and/or under control of the Navy. Groundwater from the MTA and the OJT discharges from Springs A and C, respectively, into LSC. TCE is generally not detected or only detected at trace concentrations in the springs and LSC surface waters. The TCE appears to be volatizing in the karst system and any TCE that would be present in the springs would volatize in LSC. Therefore, TCE in groundwater does not present any risk to existing onsite or offsite receptors whether it originates in the MTA or the OJT.

Barium has been detected in wells, springs, and surface waters in excess of the RBTL (3.9 µg/L) established in the Permit. The most likely media cleanup standard (MCS) for protection of human health that would be established in the CMS is the Maximum Concentration Level (MCL), which is 2,000 µg/L. Groundwater data for the Old Jeep Trail, Main Treatment Area, and Springs A and C was evaluated to determine whether this MCL was exceeded.

Old Jeep Trail: Groundwater barium data (2001 and 2004) and Spring C (2001, 2002, 2003, and 2004) is available from samples collected during RFI and RCRA-Permit monitoring programs.

The following table summarizes the average and maximum sampling results for barium.

Calendar Year	Filtered Results (µg/L)				Unfiltered Results (µg/L)			
	Average Value	Maximum			Average Value	Maximum		
		Value	Location	Date		Value	Location	Date
2001	48.3	58.3	Spring C	9/17/2001	71.1	132.	03C02P2	9/18/2000
2002	53.9	63.3	Spring C	6/24/2002	55.3	66.7	Spring C	6/24/2002
2003	47.3	63.3	Spring C	9/9/2003	46.6	60.7	Spring C	9/9/2003
2004	55.2	67.5	Spring C	9/16/2004	54.8	66.1	Spring C	9/16/2004

The data from 1994 through 2004 indicates that the barium MCL, 2,000 µg/L, has never been exceeded at the Old Jeep Trail monitoring wells or in Spring C.

Main Treatment Area: Groundwater wells and springs at the Main Treatment Area are routinely analyzed for metals including barium in accordance with the RCRA Permit for open burning operations. The following table summarizes the average and maximum sampling results for barium from 1998 through 2004:

Calendar Year	Filtered Results (µg/L)				Unfiltered Results (µg/L)			
	Average Value	Maximum			Average Value	Maximum		
		Value	Location	Date		Value	Location	Date
1998	44.2	94.8	03C02P2	11/9/1998	43.9	96.2	03C02P2	11/9/1998
1999	50.9	154.	Spring A	9/13/1999	47.1	142.	Spring A	9/13/1999
2000	46.8	123.	03C02P2	9/18/2000	48.4	132.	03C02P2	9/18/2000
2001	42.3	131.	Spring A	9/17/2001	43.	130.	Spring A	9/17/2001
2002	36.5	105.	03C02P2	11/7/2002	38.7	106.	Spring A	6/20/2002
						106.	03C02P2	11/7/2002
2003	36.1	142.	Spring A	9/9/2003	37.7	134.	Spring A	9/9/2003
2004	37.2	133.	Spring A	9/16/2004	39.8	130.	Spring A	9/16/2004

The data from 1998 through 2004 demonstrates that the barium MCL of 2,000 µg/L has never been exceeded at the Main Treatment Area monitoring wells or Spring A..

Little Sulphur Creek: Surface water samples have also been collected from Little Sulphur Creek for metals analysis during the MNA monitoring program and during RFI investigations since 1998. The following table summarizes the average and maximum sampling results for barium from 1998 through 2004:

Calendar Year	Filtered Results (µg/L)				Unfiltered Results (µg/L)			
	Average Value	Maximum			Average Value	Maximum		
		Value	Location	Date		Value	Location	Date
1998	113.4	146.	Creek A	11/5/1998	117.1	158.	Creek A	11/5/1998
1999	63.4	158.	Creek B	9/13/1999	63.	158.	Creek B	9/13/1999
2000	69.	128.	Creek B	6/9/2000	54.7	70.7	Creek A	3/8/2000

The data from 1998 through 2004 demonstrates that the barium MCL of 2,000 µg/L has never been exceeded in Little Sulphur Creek in samples collected at locations below the discharge points of Springs A and C.

Based on the above data, the MCL for barium is attained under current conditions at SWMU 3. Therefore, no remedial actions are necessary to address barium in groundwater.

6) The last bullet on page 3-11 states that evaluations of remedial alternatives for surface water and groundwater should focus on explosives rather than TCE. Why? Where? At OJT and MTA? Groundwater remedial alternatives for explosives and TCE should be evaluated in a CMS. The previous bullet indicates in-situ alternatives for remediation of contaminated groundwater at the MTA and OJT will be evaluated at locations before groundwater has entered the karst systems.

As noted in the response to Comment 5, TCE does not present a risk to existing onsite or offsite receptors whether it originates in the MTA or OJT. As noted in the response to comment 3, RDX does not present a risk to existing onsite or offsite receptors whether it originates in the MTA or OJT. The CMS will evaluate any remedial alternatives to existing receptors.

ATTACHMENT 1

CALCULATION OF SURFACE WATER QUALITY CRITERION FOR

RDX

AQUATIC RECEPTORS

SURFACE WATER QUALITY CRITERION FOR

RDX

AQUATIC RECEPTORS

A water quality screening level for RDX using water quality standard methodology which is equivalent to Indiana Department of Environmental Management (IDEM) procedures described in 327 IAC 2-1-8.3 [Determination of chronic aquatic criteria (CAC)] has recently been developed for the U.S. Army (Parametrix and ENSR, 2005). The U.S. Army document is attached. Note that although the title of this document indicates that it is for marine organisms, water quality criterion (WQC) developed for both marine and freshwater are presented in the document.

In the U.S. Army document, Parametrix and ENSR, (2005) compiled a dataset of toxicity data that was adequate to develop acute and chronic WQC for RDX using U.S. EPA guidance for deriving ambient water quality criterion for protection of aquatic life (U.S. EPA, 1985). The same U.S. EPA guidance is cited in the IDEM water quality standards regulation, 327 IAC 2-1-6, Minimum surface water quality standards, as one approach used to develop acute and chronic water quality criteria for Indiana. The acute and chronic WQC that were developed for RDX are 3,100 and 3,070 µg/L respectively (Parametrix and ENSR, 2005). The chronic value of 3,070 µg/L will be used as the screening level for RDX in surface water at Naval Surface Warfare Center Crane, Crane, Indiana.

A terrestrial life cycle safe concentration (TLSC) was also calculated according to the IDEM water quality standards. The attached table presents the calculation of the TLSC, including the equation and parameters. The TLSC was only calculated for mammals because no avian toxicity data were available. The calculated TLSC value is 2,800 µg/L.

References

Parametrix, Inc. and ENSR International. 2005. Derivation of Toxicity Reference Values for the Acute and Chronic Toxicity of Hexahydro-1,3,5-Trinitro-1,3,5-Triazine (RDX) to Marine Aquatic Organisms Final Report. Prepared for the U.S. Army Public Works, Fort Lewis, Washington. January.

U.S. EPA. 1985. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their Uses. U.S. Environmental Protection Agency, Washington, DC. PB85-227049.

CALCULATION OF TERRESTRIAL LIFE CYCLE SAFE CONCENTRATION FOR RDX
INDIANA DEPARTMENT OF ENVIRONMENTAL MANAGEMENT WATER QUALITY STANDARDS BASIS
SWMU 3 CONCEPTUAL SITE MODEL
NSWC CRANE
CRANE INDIANA

NOAEL (mg/kg-day)	Body Weight (BW) (kg)	Drinking Water rate (DW) (L/day)	Uncertainty Factor (U)	Fraction of Days Dosed Per Week (Fw)	Terrestrial Life Cycle Safe Concentration (TLSC) (mg/L)
7	0.03	0.0075	10	1	2.8

TLSC = [NOAEL * (BW/DW) * Fw]/U

- The NOAEL is based on a chronic study (2 year) with mice. The endpoint was reproduction (testicular degeneration). The source of the study was Lish et al., 1984 as cited in Talmage et al., 1999.
- The source of the mouse body weight and drinking water rate is Sample et al., 1996
- It is assumed that the fraction of days doses per week is 1 because the RDX was administered in the diet.
- An uncertainty factor of 10 was assumed because it was a long-term chronic study with a reproductive endpoint.

Lish P.M., B.S. Levine, E.M. Furedi, E.M Sagartz, and V.S. Rac. 1984. Determination of the chronic mammalian toxicological effects of RDX: twenty-four month chronic toxicity/carcinogenicity study of hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) in the B6C3F1 hybrid mouse. Phase VI. Vol. 1. AD A160774. IIT Research Institute, Chicago, IL. U.S. Army Medical Research and Development Command, Frederick, MD.

Sample, B.E., D.M. Opresko, and G.W. Suter II. 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Oak Ridge National Laboratory. June. ES/ER/TM-86/R3.

Talmage, Sylvia S., Dennis M. Opresko, Christopher J. Maxwell, Christopher J.E. Welsh, F. Michael Cretella, Patricia H. Reno, and F. Bernard Daniel, 1999. "Nitroaromatic Munition Compounds: Environmental Effects and Screening Values." Rev. Environ. Contam. Toxicol. 161:1-156.

ENSR INTERNATIONAL

DERVATION OF TOXICITY REFERENCE VALUES FOR THE ACUTE AND CHRONIC TOXICITY OF HEXAHYDRO-1,3,6-TRINITRO-1,3,5TRIAZINE (RDX) TO MARINE AQUATIC ORGANISMS

FINAL REPORT

U.S. Army Public Works

Fort Lewis, Washington

Derivation of Toxicity Reference Values for the Acute and Chronic Toxicity of Hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX) to Marine Aquatic Organisms

Final Report

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January 2005

ENSR Project No. 09000-279-400
U.S. Army Contract No. DACA67-00-D-2009
Delivery Order No. 04

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APPENDICES

Appendix A. Toxicity Test Reports

ABBREVIATIONS AND ACRONYMS

ACR	Acute-to-chronic ratio
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
AWQC	Ambient Water Quality Criteria
CAS#	Chemical Abstract Number
CCC	Criterion Continuous Concentration
CMC	Criterion Maximum Concentration
CWA	Clean Water Act
DMSO	Dimethylsulfoxide
EC_{50}	Concentration estimated to have a non-lethal effect on 50 percent of a population
ELS	Early Life Stage
FACR	Final acute-to-chronic ratio
FAV	Final Acute Value
FCV	Final Chronic Value
FDA	Food and Drug Administration
FT	Flow-through
GC/MS	Gas Chromatography/Mass Spectrometry
GMAV	Genus Mean Acute Values
HPLC	High Performance Liquid Chromatography
hr	Hour
L	Liter
LC	Full life cycle
LC_{50}	Concentration estimated to kill 50 percent of a test population
LOEC	Lowest observable effect concentration
mg	Milligram
min	Minute
ml	Milliliter
NAS	Northwestern Aquatic Sciences
nm	Nanometer
NOEC	No observable effect concentration
NPDES	National Pollutant Discharge Elimination System
PLC	Partial Life Cycle
ppb	Parts-per-billion
ppt	Parts-per-thousand
PS	Present study
QA/QC	Quality Assurance/Quality Control
R	Static Renewal
RDX	Royal Detonation Explosive
SMAV	Species Mean Acute Values
Sps	Species
SPE	Solid Phase Extraction
ST	Short-term chronic
TRV	Toxicity Reference Value
USEPA	U.S. Environmental Protection Agency
UV	Ultraviolet

1.0 INTRODUCTION

Royal Detonation Explosive (RDX; hexahydro-1,3,5-trinitro-1,3,5-triazine; (Chemical Abstract Number [CAS#] 121-82-4) has been used as an explosive by the military for nearly 100 years. This white, crystalline solid is considered to be the most powerful and brisant of the military high explosives, and it is found in most munitions used for military training by the U.S. Army at Fort Lewis, Washington. There is concern that RDX residues from stored/discarded/exploded ordnance may be harmful to aquatic organisms found in aquatic habitats near Fort Lewis, and to the humans that consume them. The Artillery Impact Area on Fort Lewis, in particular, receives heavy munitions usage during training exercises. Also, the Artillery Impact Area is adjacent to Muck Creek, a waterway that provides habitat for resident cutthroat trout (*Oncorhynchus clarki*) and anadromous chum salmon (*O. keta*). Upon detonation, RDX is converted into mostly harmless byproducts. However, in the event that a munition does not detonate, some RDX will remain in the Artillery Impact Area. Because the explosive is very mobile in water, undetonated RDX can potentially be carried from the Artillery Impact Area via stormwater runoff to Muck Creek, the Nisqually River, and ultimately to Puget Sound.

To address these issues, a literature review was conducted in 2001 on the toxicity and bioaccumulation potential of RDX with respect to aquatic organisms (ENSR 2001). One of the objectives of that study was to evaluate the current scientific literature to determine if there was sufficient information on the toxicity of RDX to freshwater and marine species. Enough relevant freshwater RDX toxicity studies were obtained to derive a provisional acute Ambient Water Quality Criterion (AWQC) for the protection of freshwater aquatic life. This provisional freshwater acute criterion was calculated to be 3.096 milligrams (mg) RDX/Liter (L) using standard U.S. Environmental Protection Agency (USEPA) methods (ENSR 2001). According to USEPA methods, RDX concentrations less than this criterion concentration should not pose a significant acute hazard (i.e., mortality) to aquatic organisms. Even though there was insufficient data to derive a criterion continuous concentration (CCC; or "chronic" criterion) according to standard USEPA guidance, the ENSR report used alternative USEPA methods (USEPA 1995a) with the available data to suggest a provisional CCC of 1.032 mg RDX/L. Concentrations of RDX that are less than this value should not pose a significant chronic hazard (i.e., inhibition of growth or reproduction) to aquatic organisms.

ENSR was not able to derive AWQC for marine organisms because of the lack of toxicity data for marine species at that time (ENSR 2001). Since the ENSR report, one new study of RDX toxicity to marine organisms has been conducted (Nipper et al. 2001), but sufficient data still did not exist to derive an AWQC for marine organisms. Therefore, additional toxicity testing was needed to derive a marine AWQC (both acute and chronic) for RDX. The goals of this study included the following:

- Conduct acute RDX toxicity tests with three species of marine fish and five species of marine invertebrates.
- Conduct chronic RDX toxicity tests with one species of marine fish, one species of freshwater invertebrate, and one marine alga.
- Calculate toxicity reference values (TRVs) using USEPA guidance for derivation of AWQC for protection of aquatic life (USEPA 1985).
- Compare marine TRVs to existing freshwater toxicity values.

2.0 DERIVATION OF TRVS/CRITERIA

There are several methods available for deriving TRVs that can be used to establish maximum levels of contaminant exposure below which there should not be significant harm to aquatic biota. Some of the most widely accepted and detailed methods are those used to derive AWQC for the Protection of Aquatic Life (USEPA 1985). These methods were developed by the USEPA for setting water quality criteria and standards for compliance with the Clean Water Act (CWA), but they are also useful in situations where strict CWA compliance is not the immediate concern, or where no officially-promulgated criterion is available. These methods are well-tested, widely-applied, and are arguably more sophisticated and scientifically-meaningful than most other methods for TRV derivation.

National AWQC set maximum threshold concentrations of contaminants for both freshwater and marine environments. These criteria are derived from empirical toxicity data and are designed to be stringent enough to protect most sensitive species potentially exposed to a contaminant in any water body in the U.S. Below these thresholds, no adverse effects are anticipated. The thresholds derived in each AWQC are designed to protect all but 5 percent of the most sensitive species. If data suggest that a commercially or recreationally important species is not protected at this level, then these values can be adjusted to provide sufficient protection for these species as well.

Each AWQC is derived using a well-defined process that relies on the collection of an adequate and reliable set of toxicity data from the open literature or "gray-literature" reports from verifiable sources (USEPA 1985). These data are reviewed for acceptability, and then used to calculate both an acute and a chronic criterion for marine and freshwater organisms using the steps outlined in Figure 2-1.

Furthermore, these criteria concentrations can be made a function of water quality characteristics (such as pH or hardness), if scientifically justified.

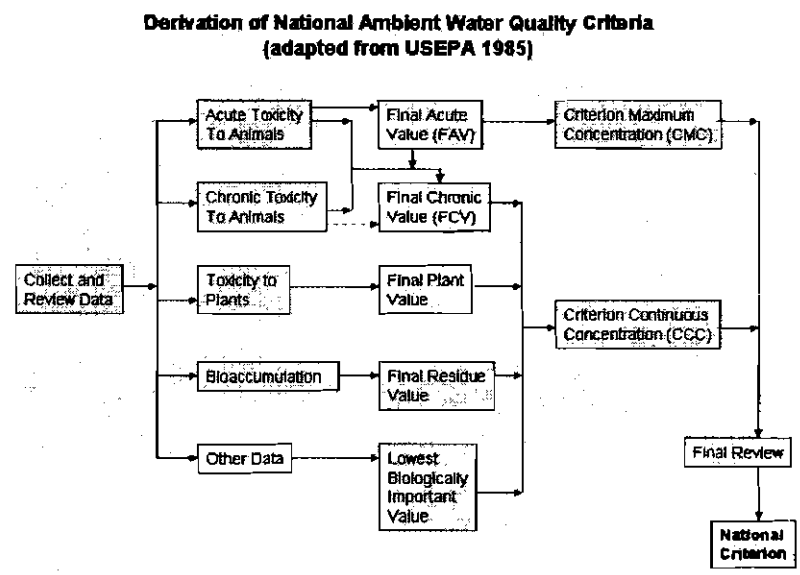

Figure 2-1. Flow Diagram for AWQC Development

2.1 DATA COLLECTION AND SCREENING

Data are collected from all available published and non-published sources, but only when sufficient documentation exists to judge the reliability and accuracy of these data. Types of data collected include toxicity and bioaccumulation values for animals and plants, Food and Drug Administration (FDA) action levels for tissue residue concentrations, and chronic feeding and long-term field studies with wildlife species that consume aquatic organisms. These data are then screened for technical reliability to ensure that toxicity values obtained are likely to be correct. The screening criteria address generally accepted characteristics of a well-conducted scientific study, and include considerations such as control treatment and organism performance criteria (e.g., control mortality was

not excessive), purity of the material used in toxicity testing, and use of only North American resident species for AWQC derivation (USEPA 1985).

USEPA guidance further specifies that, at least for acute toxicity data, a minimum dataset must be constructed to represent the impacts of contaminants on several distinct taxonomic groups. Acceptable acute tests must be available for at least eight different families, with the specific taxonomic requirements for freshwater and marine animals shown in Table 2-1. These data then are used to derive the Final Acute Value (FAV) as detailed below. Final Chronic Values (FCVs) also ean be derived using the same procedure, but given the cost and complexity of chronic toxicity tests, Acute-to-Chronic Ratios (ACR) calculated from a minimum of three different species are more commonly used (Table 2-1). In many cases, the same three chronic toxicity values can be used to calculate ACRs for both freshwater and marine organisms, as long as data also exist for at least one acutely sensitive marine and freshwater species.

Additional data must be collected for at least one acceptable toxicity test with an alga or vascular plant (both freshwater and marine), and at least one acceptable bioconcentration factor must be collected (freshwater and marine) if a FDA action level for maximum permissible tissue concentrations is available for that particular chemical.

Table 2-1

Minimum Taxonomic Requirements for Derivation of Freshwater and Marine Ambient Water Quality Criteria

Test Type	Freshwater Families (Eight required)	Marine Families (Eight required)
Acute	• Salmonidae family (Osteichthyes) • Second family in Osteichthyes • Third family in phylum Chordata • Planktonic crustacean • Benthic crustacean • Aquatic insect • Family in phylum other than Chordata • Family in any order of insect or any phylum not already represented	• Family in phylum Chordata • Second family in phylum Chordata • One family in phylum other than Arthropoda or Chordata • One member of Mysidae or Penaeidae • Family not in phylum Chordata • Second family not in phylum Chordata • Third family not in Chordata • Any other family
Chronic	• At least one fish • At least one invertebrate • At least one acutely sensitive species	• At least one fish • At least one invertebrate • At least one acutely sensitive species

2.2 FINAL ACUTE VALUE (FAV)

The FAV is designed to represent the contaminant concentration at which all but the most sensitive 5 percent of species are protected against acute toxicity (i.e., levels resulting in 50 percent mortality of individual organisms). Data collected from at least eight families as designated above (Table 2-1) are reduced down to a ranked set of genus mean acute values (GMAVs), which are the geometric means of all Species Mean Acute Values (SMAV; geometric mean of all toxicity data for a given species) for each genus. A statistical curve-fitting procedure is then used to fit a log-triangular distribution model to the GMAV data. If there are less than 59 GMAVs available (which is the case for all criteria to date), one only needs to enter the total number of GMAVs, and the toxicity values for the lowest *four* GMAVs in a particular dataset. This procedure has been shown to provide the most accurate estimate of an FAV that corresponds to a concentration below which all but 5 percent of the species are protected (USEPA 1988). A graphical example of an acute dataset—and the FAV concentration estimated from these data—is presented for copper in Figure 2-2.

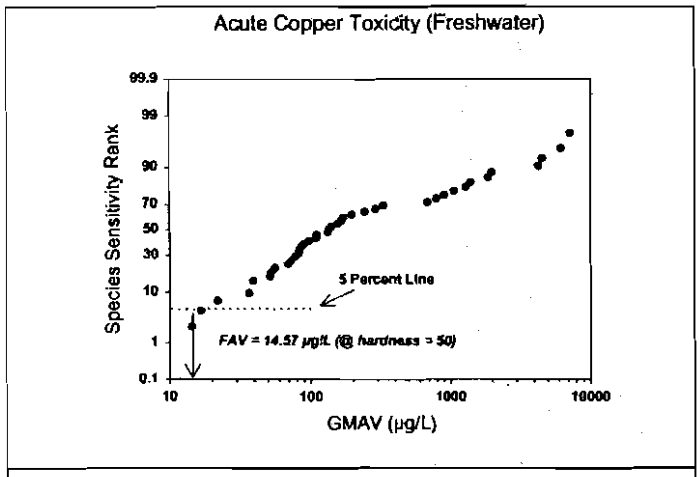

Figure 2-2. Relationship Between Final Acute Value and Percent Species Sensitivity to Copper (from Copper Ambient Water Quality Criteria)

2.3 FINAL CHRONIC VALUE (FCV)

The FCV can be calculated using the same methods as for FAVs, but acceptable chronic values for at least eight families (Table 2-1) are rarely achieved. The more common method is to use the ACR, which essentially "corrects" an acute value to provide an estimate of chronic toxicity. In most AWQC, this ratio is used to compare chronic values—of which few typically exist—against acute toxicity values as a means of estimating an FCV. Individual ACRs are derived by dividing each acceptable chronic value (derived from flow-through life-cycle, partial life-cycle, or early life-stage toxicity test results) into an acute toxicity value for the same species, preferably taken from the same study or at least the same laboratory (Figure 2-3). After compiling all acceptable species' ACRs, a final ACR (FACR) is calculated by one of several methods, the most common of which is to take the simple geometric mean of all individual ACRs. The FCV is then derived by dividing the FACR into the FAV (Figure 2-3), which was already derived using the curve-fitting procedure described in Section 2.2.

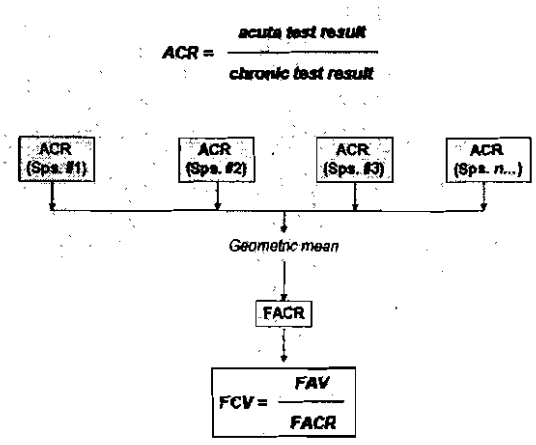

Figure 2-3. Derivation of a Final Chronic Value using the Acute-to-Chronic Ratio Method

2.4 FINAL PLANT VALUE

A Final Plant Value is used to ensure that toxicity thresholds derived from animal toxicity data are sufficiently protective of aquatic plants. No proeedures analogous to FAV or FCV derivation are used; instead the lowest acceptable aquatic plant toxicity test value is simply compared to the FCV. This lowest plant value is selected from an important plant species in which chemical exposure concentrations were measured, and the endpoint is biologically important (e.g., inhibition of growth or photosynthesis). A plant value is usually the result of a 96-hour test conducted with an alga or a chronic test with an aquatic plant.

2.5 FINAL RESIDUE VALUE

The Final Residue Value is designed to prevent tissue contaminant concentrations from affecting marketability of aquatic species owing to exceedence of FDA action levels. Similar to the Final Plant Value, Final Residue Value derivation is a simple proeess, and is set to the lowest residue value obtained for any species. Each residue value is calculated by dividing maximum permissible tissue concentrations (i.e., FDA action levels) by a bioconcentration or bioaccumulation factor for that species. These factors in turn are simple ratios of exposure concentrations divided into tissue residue concentrations.

However, few AWQC ultimately incorporate a Final Residue Value, predominantly owing to the lack of appropriate/relevant FDA action levels or the laek of significant bioaccumulation by the criteria chemical. Furthermore, in a revision to AWQC derivation guidance for the Great Lakes, Final Residue Values are not considered because the potential effects of food chain transfer from aquatic organisms to wildlife and humans are evaluated using a separate procedure (USEPA 1995a). Final Residue Values, thus, are of limited relevance to final derivation of AWQC in most cases. For RDX, a Final Residue Value will not be considered because of its limited bioaccumulation potential and because no FDA action levels have been established (ENSR 2001).

2.6 CRITERION MAXIMUM AND CRITERION CONTINUOUS CONCENTRATIONS

Each AWQC consists of two toxicity thresholds: the CMC (criterion maximum concentration' also called an "acute criterion"), and the CCC (also called a "chronic criterion"). The CMC is designed to protect aquatic organisms from short-term, more severe impacts (i.e., mortality) resulting from relatively high concentrations of contaminants. Its derivation is relatively simple, and usually is based entirely on the FAV value. The CMC is set to one half of the FAV so that all but the most sensitive 5 percent of species are protected at levels substantially lower than 50 percent acute impact.

The CCC is designed to protect aquatic organisms from longer-term, less-severe impacts (i.e., inhibition of growth or reproduction) of relatively low concentrations of a contaminant. Its derivation is based upon the FCV, but also involves consideration of other toxicity data for recreationally or commercially important species. The FCV is often based upon a "correction" of the FAV using an ACR approach (Figure 2-3). The CCC is equivalent to the FCV unless any one of the following is lower than the FCV:

- Final Plant Value
- Final Residue Value (if calculated)
- Any acceptable toxicity value for a recreationally or commercially important species

2.7 WATER QUALITY CORRECTIONS

Ambient Water Quality Criteria concentrations can be made a function of water quality characteristics if scientifically justified. This is because the toxicity of many contaminants can be modified in the presence of certain water quality characteristics. Perhaps the most widely used water quality correction is water hardness, which is used to correct AWQC concentrations for several metals. The bioavailability and, hence, toxicity of many metals is diminished in waters of increasing hardness, and so the CCC and CMC concentrations are expressed in terms of an equation rather than a single value.

2.8 FINAL CRITERION

The final criterion is a combination of the CMC and CCC for both freshwater and saltwater organisms. Each criterion is stated in terms of a threshold magnitude (concentration), averaging period (duration), and a frequency of allowed excursions. Although a criterion magnitude determines what concentration should not be exceeded to protect aquatic organisms, criteria implementation (e.g., using criteria or standards to set National Pollution Discharge Elimination System [NPDES] permit limits) also depends upon the duration and frequency components of an AWQC. The averaging period (duration) and frequency components were designed as alternatives to a criterion being based on a single instantaneous concentration. This is because water quality is not a static condition in ecological systems; instead it depends on natural variability in contaminant concentrations (i.e., not unusual or accidental spills), as well as variability in receiving water and effluent flows. In addition, aquatic organisms often can tolerate higher concentrations of a contaminant for shorter periods of time than they can tolerate at a constant level throughout their entire life cycle. Thus, the magnitude-duration-frequency approach for AWQC derivation is designed to be a reasonably realistic means of protecting aquatic life without being overly conservative (USEPA 1991).

Typical averaging periods for most AWQC are 1 hour for acute, and 4 days for chronic criteria, and all AWQC use the default 3-year frequency of allowed excursions. Final criterion concentrations can be adjusted upwards or downwards if laboratory or field evidence with important species warrants such an adjustment. Adjustments can also made on a site-specific basis, but are typically not considered by USEPA in derivation of the AWQC (USEPA 1985).

3.0 FRESHWATER TOXICITY DATA FOR RDX

3.1 ACUTE TOXICITY OF RDX TO FRESHWATER ORGANISMS

Most of the freshwater toxicity tests included in the interim AWQC (Etnier 1986) were conducted in a series of studies by Bentley et al. (1977) and Liu et al. (1983). Collectively, these studies evaluated the toxicity of RDX to four microalgae, four invertebrate species, and four species of freshwater fish (Table 3-1). Under these test conditions, RDX was not acutely toxic to any of the invertebrate species, with LC_{50} values greater than maximum solubility. Similarly, none of the algal studies exhibited significant impacts as determined by EC_{50} calculations. In contrast, RDX was acutely toxic to all four species of fish, and at levels that were remarkably similar (ranging from 4.1 to 13 mg RDX/L). Many of these studies used dimethylsulfoxide (DMSO) as a delivery solvent, and reported toxicity as nominal[1], rather than measured, concentrations of RDX. However, nominal concentrations were reported to be stable and similar to measured concentrations (Bentley et al. 1977).

Since the review by Etnier (1986), studies have been conducted that provide data from at least two additional families. With this additional information, an interim freshwater AWQC for RDX can be derived. Peters et al. (1991) conducted static-renewal toxicity tests with a cladoceran (*Ceriodaphnia dubia*) and a hydra (*Hydra littoralis*), and flow-through tests with a midge (*Paratanytarsus parthenogeneticus*). No delivery solvents were used, RDX concentrations were confirmed analytically, and all relevant Quality Assurance/Quality Control (QA/QC) acceptance criteria (e.g., sufficiently low control mortality and adequate and consistent water quality conditions) were met. As in the other invertebrate studies, RDX was not acutely toxic to any of these three invertebrate taxa (Table 3-1; Bentley et al. 1977). Additional flow-through acute testing was also conducted with the fathead minnow, *Pimephales promelas,* and acute toxicity was nearly 50 percent less (12.7 mg RDX/L) than in earlier studies (Peters et al. 1991). More recent toxicity studies have also been conducted with the bacterium *Vibrio fischeri* (Drzyzga et al. 1995) and the freshwater alga *Selenastrum capricornutum* (Burton et al. 1994a). As indicated by EC_{50} values, RDX is not acutely toxic to either of these taxa at, or even above (for the bacterium), limits of aqueous solubility (Table 3-1).

[1] Nominal concentrations refer to the amount of chemical added to test chambers, which may or may not be an accurate representation of the actual concentration of dissolved chemical to which the organism is exposed. Many chemicals can adhere to test chamber materials, or perhaps be taken up by the organisms themselves, so it is generally preferred to measure exposure concentrations using an appropriate analytical method.

Table 3-1

Acute Toxicity of RDX to Freshwater Aquatic OrganismsTest Species

	Method[1]	Duration[2]	LC_{50}/EC_{50} (mg/L)	Genus Mean Acute Value (mg/L)	Reference
Bacteria					
Vibrio fischeri	S	30 min	74.6	N/A[3]	Drzyzga et al. 1995
Algae					
Selenastrum capricornutum	S	96 hr	> 36.7[4]	N/A	Burton et al. 1994b
Selenastrum capricornutum	S	96 hr	> 32		Bentley et al. 1977
Microcystis aeruginosa	S	96 hr	> 32	N/A	Bentley et al. 1977
Anabaena flosaquae	S	96 hr	> 32	N/A	Bentley et al. 1977
Navicula pelliculosa	S	96 hr	> 32	N/A	Bentley et al. 1977
Invertebrates					
Aselius militaris	S	48 hr	> 100	100	Bentley et al. 1977
Ceriodaphnia dubia	R	48 hr	> 17	17	Peters et al. 1991
Chironomus tentans	S	48 hr	> 100	38.7	Bentley et al. 1977
Chironomus tentans	FT	48 hr	> 15		Bentley et al. 1977
Daphnia magna	S	48 hr	> 100	38.7	Bentley et al. 1977
Daphnia magna	FT	48 hr	> 15		Bentley et al. 1977
Gammarus fasciatus	S	48 hr	> 100	100	Bentley et al. 1977
Hydra littoralis	R	48 hr	> 32	32	Peters et al. 1991
Paratanytarsus parthenogeneticus	FT	48 hr	> 29	29	Peters et al. 1991
Fish					
Ictalurus punctatus	S	96 hr	4.1	7.3 (4)[5]	Bentley et al. 1977
Ictalurus punctatus	FT	96 hr	13		Bentley et al. 1977
Lepomis macrochirus	S	96 hr	6	6.75 (2)	Bentley et al. 1977
Lepomis macrochirus	FT	96 hr	7.6		Bentley et al. 1977
Pimephales promelas	S	96 hr	4.5	6.84 (3)	Liu et al. 1983
Pimephales promelas	S	96 hr	5.8		Bentley et al. 1977
Pimephales promelas	FT	96 hr	6.6		Bentley et al. 1977
Pimephales promelas	FT	96 hr	12.7		Burton et al. 1994a
Oncorhynchus mykiss	S	96 hr	6.4	6.4 (1)	Bentley et al. 1977

1 S = static, R = static renewal, and FT = flow-through.

2 Duration reported as minutes (min) or hours (hr).

3 N/A = not applicable. GMAVs from these taxa are not used in calculating final acute values according to USEPA Guidance (USEPA 1985).

4 Values with > indicate that toxicity was greater than maximum solubility.

5 Numbers in parentheses represent ranking of four lower GMAVs.

3.2 CHRONIC TOXICITY OF RDX TO FRESHWATER ORGANISMS

3.2.1 Invertebrates

Only two studies were available that conducted chronic toxicity tests on aquatic invertebrates (Bentley et al. 1977, Peters et al. 1991). Collectively, these studies conducted tests on four different species, including the cladocerans *C. dubia* and *Daphnia magna*, and the midges *Chironomus tentans* and *P. parthenogeneticus* (Table 3-2). It was found that RDX was not chronically toxic to either species of midge, with no effects observed in flow-through life-cycle studies in solutions of up to 21 mg RDX/L. Flow-through tests with *D. magna* reported No Observable Effect Concentration (NOEC) and Lowest Observable Effect Concentration (LOEC) values of 2.2 and 4.8 mg RDX/L, respectively. However, these results should be interpreted with caution because effects were only observed between 7 and 14 days, and mortality exceeded 20 percent in some of the controls. Similar, but more definitive, chronic toxicity values were observed using *C. dubia* in which 7-day static tests achieved NOEC and LOEC values of 3.6 and 6 mg RDX/L, respectively (Peters et al. 1991). From these data, both a chronic value (ChV) of 4.7 mg RDX/L, and an acute-to-chronic ratio (ACR) of 3.6325 for this species can be calculated (Table 3-2).

3.2.2 Fish

Two sets of chronic toxicity tests with RDX were conducted with fathead minnows (Bentley et al. 1977, Burton et al. 1994b). In the first set of flow-through tests, both Early Life Stage and Partial Life Cycle tests achieved similar NOEC and LOEC values of approximately 3 to 6 mg RDX/L, respectively (Table 3-2). More recent ELS tests by Burton et al. (1994b) reported somewhat greater RDX toxicity with NOEC and LOEC values of 1.4 and 2.4 mg RDX/L, respectively. From these data, chronic values could be calculated, which ranged from 1.8 to 4.3 mg RDX/L (Table 3-2). Acute-to-chronic ratios ranged from 1.5349 to 7.1152, achieving a species geometric mean ACR of 3.3047 (similar to the ACR for *C. dubia*; Table 3-2).

3.2.3 Algae

As described in Section 3.1, Bentley et al. (1977) conducted algal toxicity tests using four different taxa (Table 3-1), and reported that EC_{50} values could not be estimated owing to minimal growth inhibition in all taxa. However, a subsequent reanalysis of these data using chronic endpoints suggested that growth was inhibited in all species, with NOECs ranging from < 0.32 mg RDX/L for *S. capricornutum*, to 10 mg RDX/L for *Microcystis aeruginosa* (Sullivan et al. 1979). Similarly, when chronic endpoints were derived for the more recent algal study (Burton et al. 1994a), a NOEC and a LOEC of 0.5 and 4.8 mg RDX/L, respectively, were observed. These data suggest that *S. capricornutum* is the most sensitive algae tested, with NOEC values less than 1 mg RDX/L. However, LOEC values in this species are similar to acute toxicity concentrations observed for freshwater fishes (Table 3-1).

Table 3-2

Chronic Toxicity of RDX to Freshwater Aquatic Organisms

Test Species	Method[1]	Duration (days)	Observed Effects	NOEC[2] (mg/L)	LOEC[3] (mg/L)	Chronic Value (mg/L)	Acute to Chronic Ratio (ACR)[5]	Reference
Invertebrates								
Ceriodaphnia dubia	R, ST	7	survival/reproduction	3.6	6	4.7	3.632	Peters et al. 1991
Chironomus tentans	FT, LC	23	growth/survival/emergence			> 21		Bentley et al. 1977
Daphnia magna	FT, ELS	21	reproduction	2.2[4]	4.8[4]			Bentley et al. 1977
Paratanytarsus parthenogeneticus	FT, LC	17 (?)	growth/survival			> 21		Peters et al. 1991
Fish								
Pimephales promelas	FT, ELS	30	growth	3	5.8	4.2		Bentley et al. 1977
Pimephales promelas	FT, PLC	240	survival	3	6.3	4.3	1.535	Bentley et al. 1977
Pimephales promelas	FT, ELS	28	growth	1.4	2.4	1.8	7.115	Burton et al. 1994

[1] R = static renewal, FT = flow-through, ST = short-term chronic, ELS = early life stage, LC = full life cycle, PLC = partial life cycle.

[2] NOEC = No Observable Effect Concentration (i.e., highest concentration at which no statistically significant adverse effect was observed).

[3] LOEC = Lowest Observable Effect Concentration (i.e., lowest concentration at which a statistically significant adverse effect was observed).

[4] Values should be interpreted with caution because effects were only observed between 7-14 days, and mortality exceeded 20 percent in some controls.

[5] Acute values obtained from same study.

4.0 MARINE TOXICITY DATA FOR RDX

At the time of the initial literature review, no RDX toxicity studies with marine organisms had been conducted (ENSR 2001). Since that time, a single study has been conducted that tested RDX toxicity to five species of marine organisms (Nipper et al. 2001). This study included both acute and chronic toxicity tests, but not all were of the type that would be considered acceptable for use in deriving AWQC (USEPA 1985). The following summarizes the RDX toxicity studies of Nipper et al. (2001) and their relevance to AWQC development according to USEPA guidance.

All toxicity tests were designed to test toxicity using concentrations no higher than the maximum experimentally achievable solubility. Test solution salinity was 30 parts per thousand (ppt), and test temperatures were 20 °C except for the redfish test, which used 25 °C waters. All test endpoints were derived using typical statistical procedures, and were based on analytically-verified exposure concentrations. Adequate experimental details (including QA/QC considerations such as control survival and maintenance of adequate dissolved oxygen concentrations) were provided to evaluate test reliability. Tests were conducted in complete darkness to prevent RDX loss from photolysis.

- **Redfish (*Sciaenops ocellatus*) short-term chronic test**. A 48-hour embryo hatching and larval survival test was used to evaluate the short-term chronic (i.e., sublethal) toxicity of RDX to redfish. No adverse effects to these very early life stages were observed up to a maximum test solubility of 68 mg RDX/L. This was consistent with fathead minnow (*P. promelas*) studies by Bentley et al. (1977) that showed these life stages to be relatively resistant to RDX. However, because the test duration is substantially less than the shortest early life-stage test allowed for use by USEPA (30 days) and only encompasses two early life stages that may be relatively resistant to RDX toxicity, this redfish test could not be used in derivation of a chronic AWQC (USEPA 1985).

- **Sea urchin (*Arbacia punctulata*) embryo-larval acute test**. A 48-hour fertilization and embryo-larval development test (Carr and Chapman 1992), was used to evaluate the short-term toxicity of RDX to sea urchins. No adverse effects of RDX were observed up to a maximum test solubility of 75 mg RDX/L. This sea urchin test cannot be used as an acute test for AWQC derivation because it was not clear if the test evaluated the percentage of organisms killed in addition to those with incomplete development (USEPA 1985).

- **Polychaete (*Dinophilus gyrociliatus*) short-term chronic test**. A 7-day survival and reproduction test (Carr et al. 1989), starting with 1- to 2-day post-emergent females in which mortality and number of eggs per surviving female were recorded after 4 and 7 days, demonstrated chronic toxicity of RDX to polychaetes. No mortality was observed up to a maximum test solubility of 49 mg RDX/L, but the number of eggs per surviving female was adversely affected by a concentration—estimated to have a non-lethal effect on 50 percent of the population (EC_{50})—of 26 mg RDX/L. This test is likely not appropriate for use in derivation of a chronic AWQC because no indication was given that this test either represents—or is a reasonable predictor of—a chronic life-cycle toxicity test with this species (USEPA 1985).

- **Algal (*Ulva fasciata*) germination and early growth test**. A 96-hour zoospore germination and germling growth test (Hooten and Carr 1998) was chosen as an indicator of RDX toxicity to a marine alga. In this test, RDX was toxic to *U. fasciata* at 12, 9.8, and 8.1 mg RDX/L for percent germination, cell number, and germling length, respectively. However, the relevance of this particular study for use as a Final Plant Value in the development of a chronic AWQC is open to question. Guidance developed by the USEPA (1985) only states that such a test can be the result of a 96-hour test conducted with an alga, and that the "Final Plant Value should be obtained by selecting the lowest result from a test with an important aquatic plant species in which the *concentrations of test material were measured* and the endpoint was *biologically important*" (emphases added). While these criteria may appear to be met, it is not clear

whether the zoospore stage is the most sensitive portion of this species' life cycle with respect to RDX toxicity. Given that the Nipper et al. (2001) study shows the marine alga to be more sensitive than any of the animal taxa tested, it would be prudent to conduct another 96-hour toxicity test using a standard population-based microalgal test that is not dependent on life cycle-specific toxicity.

- **Mysid shrimp (*Americamysis bahia*) acute survival test**. Nipper et al. (2001) used a standard USEPA 96-hour test to quantify the acute toxicity of RDX to this marine crustacean. No adverse effects were recorded up to a maximum solubility of 47 mg RDX/L. Given that they used a standard USEPA acute toxicity test method, this test is clearly appropriate for use in derivation of an acute AWQC (USEPA 1985).

Therefore, even though acute RDX toxicity data were available for five marine species, only one of these (acute toxicity to the mysid shrimp) was clearly appropriate for use in deriving a marine AWQC, with perhaps one other also being appropriate (marine algal test). Additionally, no data are available regarding the potential dependence of RDX toxicity to marine organisms on external environmental factors or water quality.

5.0 MARINE TOXICITY TESTING METHODS

As discussed in Section 2.1, an acute criterion requires toxicity data from at least eight specific marine families (Table 2-1), and a chronic criterion requires toxicity data from at least three species, only one of which must be an acutely sensitive marine species (USEPA 1985). An additional algal toxicity test may also be required to ensure the chronic criterion concentration is likely to be protective of marine plants. In light of the toxicity tests reviewed in Section 4, we first summarize additional toxicity tests (Section 5.1) that were needed to complete minimum data requirements according to USEPA guidance. Detailed protocols for each of the tests were included in Appendix A of ENSR (2003).

5.1 ACUTE TESTS

Even though one acceptable acute marine toxicity test was available for RDX (for the mysid shrimp *A. bahia*, see Section 4), we conducted all the necessary toxicity tests to satisfy the eight-family minimum data requirements of USEPA.

Table 5-1

Summary of Testing Required to Derive an Acute Marine Criterion for RDX

Marine Families Required	Tests to Fulfill Requirement
• Family #1 in phylum Chordata	• Sheepshead minnow, *Cyprinodon variegatus*
• Family #2 in phylum Chordata	• Silverside, *Menidia beryllina*
• Family iu phylum other than Arthropoda or Chordata	• Blue mussel, *Mytilus galloprovincialis*
• One crustacean from either the Mysidae or Penaeidae	• Mysid shrimp, *Americamysis bahia*
• Family #1 not in phylum Chordata	• Dungeness crab, *Cancer magister*
• Family #2 not in phylum Chordata	• Sand dollar, *Dendraster excentricus*
• Family #3 not in phylum Chordata	• Polychaete worm, *Neanthes arenaceadentata*
• Any other family	• Speckled sand dab, *Citharichthys stigmaeus*

5.1.1 General Study Plan

All toxicity testing was conducted by Northwestern Aquatic Sciences (NAS) in Newport, Oregon (Dr. Richard Caldwell, Study Director). All appropriate licenses and training were obtained for the safe use and storage of RDX. In most instances, tests were conducted consecutively. Much of the testing was conducted in a designated flow-through constant temperature room (12 feet x 8 feet). A Mount-Brungs siphon dilutor provided test solutions to the exposure vessels except in algal (*Skeletonema costatum*), mussel (*Mytilus galloprovincialis*), and echinoderm (*Dendraster excentricus*) studies. Five test concentrations and a control, each with four replicates, were used in the studies. Exposure solutions were prepared from saturated stocks containing 99.9 percent pure RDX (courtesy of Dr. S. Caulder, Naval Surface Warfare Center, Maryland) dissolved in ~30 percent filtered seawater collected from Yaquina Bay, Newport, Oregon. No test concentrations higher than aqueous solubility under test conditions were employed. When toxicity was not observed under this condition, the results were reported as greater than the saturation concentration.

Concentrations of RDX were monitored in all test concentrations and the control at test initiation and termination in the acute toxicity tests. For chronic studies, test concentrations were measured at test initiation, weekly thereafter, and at test termination. Some additional analyses were required for calibration of the Mount-Brungs dilutor. Water samples were preserved and shipped to Columbia Analytical Laboratory (Kelso, Washington) for RDX analysis. Because photolysis is one of the primary factors responsible for losses of RDX in surface water (Etnier 1986, Agency for Toxic Substances and Disease Registry [ATSDR] 1995), tests were conducted in subdued light, if possible, to prevent unacceptable losses of RDX.

5.1.2 Flow-Through Acute Toxicity Tests

The experimental design of all flow-through acute toxicity tests incorporated five test concentrations and a dilution water control. All were of 96-hour duration. Four replicates were tested for each concentration, and the control had a minimum of five organisms per replicate. The endpoint was mortality, and it was determined by enumeration of survivors at the conclusion of the test.

Dungeness crab, *C. magister*, first instar zoeae (a larval form) were obtained by hatching and release from ovigerous (egg-bearing) female crabs in the laboratory. Testing of this species utilized a flow-through apparatus of Buchanan et al. (1975). The Buchanan et al. (1975) method allowed a gentle flushing of a sereened 250 milliliter (ml) beaker held in a low-sided aquarium equipped with an auto-siphon flushing mechanism.

The acute toxicity tests with the polychaete worm, *Neanthes arenaceadentata*, hegan with 3- to 4-week post-emergence worms purchased from Dr. Don Reish at Long Beach State University, Long Beach, California. The 96-hr acute tests with *Menidia beryllina* and *Cyprinodon variegatus* were initiated with 13 to 15 day-old larvae purchased from commercial test organism suppliers. Mysid shrimp, *A. bahia*, and speckled sand dab, *C. stigmaeus*, were obtained from a commercial supplier also, and were 6 days old or juveniles (<40 mm in length) at test initiation, respectively. The 96-hour exposures to RDX for these speeies, with the exception of *Citharichthys stigmaeus*, employed the same screened eompartment flow-through apparatus used in the tests with crabs.

5.1.3 Embryo-Larval Tests of Mussels and Echinoderms

A 48-hour embryo-larval test of the hlue mussel, *M. galloprovincialis*, was conducted in accordance with the USEPA West Coast marine testing manual (USEPA 1995b). In this static test, a saturated RDX solution was manually diluted and dispensed (10 ml/replicate) into 30 ml test vials. Four replicates with 16.9 test organisms per ml were employed. Extra replicates with larvae were prepared for monitoring RDX concentrations during the test.

An embryo-larval test of the sand dollar, *D. excentricus*, also was conducted in accordance with the USEPA West Coast marine testing manual (USEPA 1995b). Adults of both species in spawning condition were purchased from a reputable test organism supplier. This test was performed in essentially the same manner as the mussel test, but it was terminated after 65 hours because pluteus development was complete within that time. For both tests, EC_{50}s were based on the percentage of organisms with incompletely developed shells, plus the percentage of organisms killed, to ensure that results were acceptable for use in deriving acute AWQC (USEPA 1985).

5.2 CHRONIC TESTS

In the case of RDX, chronic toxicity data already exist for one species of fish and three species of freshwater invertebrates (Table 3-2). Even though the *C. dubia* study reported an ACR remarkably similar to that of the fathead minnow, it was believed that this test was not acceptable for use in derivation of a chronic criterion because a short-term (7-day) chronic study method was used (USEPA 1985). However, USEPA now allows the use of full 7-day *C. dubia* tests in deriving chronic criteria (USEPA 1999).

It was still considered prudent to conduct a full life-cycle test with a freshwater invertebrate to supplement the short-term chronic test that already exists for *C. dubia* because in addition to the short-length of this test, problems were also reported with the *D. magna* study (see footnote in Table 3-2). Therefore, a new chronic study was conducted with the freshwater invertebrate *D. magna*, and a new chronic study with an acutly sensitive marine fish, *C. variegates*, was also conducted. In both cases, NAS conducted acute toxicity tests to enable calculation of ACRs using the same laboratory and test conditions for both acute and chronic toxicity values according to USEPA guidance (USEPA 1985). In addition to the existing ACR for fathead minnows (Table 3-2), these data provided at least the minimum of three ACRs required for derivation of an FCV.

5.2.1 Acute and Life-Cycle Test of *Daphnia magna*

A life-cycle (chronic) test with the cladoceran, *D. magna*, was conducted in order to derive an acceptable ACR for this species. To generate the chronic value for *D. magna* for use in ACR calculation, a standard 21-day life-cycle test was conducted. Such chronic tests also allow for the 96-hr mortality data to be used to generate an LC_{50} for use in the ACR calculation as well. In the life-cycle test with *D. magna*, newly released neonates (<24-hr old) were exposed to RDX test solutions in a static renewal system for 21 days (American Society for Testing and Materials [ASTM] 1996a). Test results were based on survival and reproduction, the latter based on numbers of live neonates produced in the F_1 generation. *Daphnia magna* were obtained from cultures maintained at NAS.

5.2.2 Early Life-Stage Test of Sheepshead Minnow

Both acute and chronic tests were conducted with sheepshead minnow, *C. variegatus*, in order to derive an acceptable ACR for this species. The acute toxicity test followed the procedures described in Section 5.1.2. The 28-day early life stage test with *C. variegatus* was conducted in accordance with American Society for Testing and Materials guidance (ASTM 1996b). This test began with fertilized eggs at 24 to 48 hours post-fertilization. Each of the four replicates that started with 36 fertilized eggs each was culled to 15 larval fish per replicate on day 7. Pre-fertilized eggs were purchased from a commercial supplier. The exposure system was the same flow-through design and apparatus employed in the flow-through acute tests. The results were based on hatch, survival, and growth rates of the larval fish.

5.2.3 Marine Algal Toxicity Test

As described in Section 2.4, a Final Plant Value is needed to ensure that the FCV would be protective of aquatic plants. Given that the marine algal test by Nipper et al. (2001) may not be the most appropriate test for this purpose, an algal toxicity test was conducted with the marine microalga *S. costatum*. The test was conducted in general accordance with the USEPA guideline for conducting algal toxicity tests with *S. capricornutum* (USEPA 1994). In this test, algae in logarithmic-phase growth[2] were inoculated into test chambers at a density of 2.3×10^4 cells/ml. The test was maintained under a 16:8 hr, L:D photoperiod for 96 hours, at which time cell density in each of the test replicates/treatments was enumerated using a Neubauer counting chamber. The results were based on a reduction in cell density relative to the control.

5.3 ANALYTICAL CHEMISTRY

Water samples were taken from all toxicity test solutions at the start and conclusion of each test to analytically confirm RDX exposure concentrations (see Appendix A test protocols for details; ENSR 2003). Concentrations of RDX were monitored in all test concentrations and the control at test initiation and termination in the acute toxicity tests. For chronic studies, test concentrations were measured at test initiation, weekly thereafter, and at test termination. Some additional analyses were required for calibration of the Mount-Brungs dilutor. Water samples were preserved and shipped on ice to Columbia Analytical Services for analysis.

Concentrations of RDX were analyzed by High Performance Liquid Chromatography (HPLC) using USEPA Method 8330 for Nitroaromatics and Nitramines, which provides HPLC detection of parts-per-billion (ppb) levels of explosives residues in water. Aqueous samples were first extracted using Solid Phase Extraction (SPE) to clean up and concentrate the sample (see USEPA methods 8330A and 3535). Samples were passed through a pre-conditioned SPE sorbent tube, and then RDX was extracted using acetonitrile, which was then concentrated. The concentrate was diluted 1:1 with aqueous calcium chloride, filtered, and then analyzed. The HPLC analysis used a C-18 reverse phase column and an ultraviolet (UV) detector at a wavelength of 254 nanometers (nm). Analytes

[2] Logarithmic-phase growth denotes maximum population growth rate in which the logarithm of biomass or cell numbers increase linearly as a function of time.

were confirmed using a cyano column. Spectral analysis using peak purity and qualitative Gas Chromatography/Mass Spectrometry (GC/MS) confirmation were used to assist in identifications. The HPLC was calibrated for each analytical run using certified analytical standards. Typical/standard QA/QC procedures (e.g., use of surrogate standards and matrix spike/duplicate matrix spike analysis) were used to ensure accuracy and precision of the analysis.

6.0 RESULTS

6.1 ACUTE TOXICITY

The results of the acute toxicity of RDX to marine species, including the Nipper et al. (2001) result for *A. bahia*, are presented in Table 6-1. Similar to freshwater organisms (Table 3-1), RDX was only acutely toxic to fish, and no invertebrates were acutely sensitive even at maximum RDX solubility. The marine algae *S. costatum* was intermediate in sensitivity between fish and invertebrates (i.e., about two times less sensitive than any of the fish species and about two times more sensitive than any of the invertebrates). The similarity between the toxicity of RDX to both freshwater and marine species is shown in the strong overlap between species sensitivity distributions of acute toxicity values (Figure 6-1).

Table 6-1

Acute Toxicity of RDX to Marine Species

Test Species	Method[1]	Duration	LC_{50}, EC_{50} (mg RDX/L)	GMAV[2] (mg RDX/L)	Reference
Invertebrates					
Mysid shrimp, *Americamysis bahia*	FT	96 hr	> 53	50	This Study
Mysid shrimp, *Americamysis bahia*	S	96 hr	> 47		Nipper et al. 2001
Dungeness crab, *Cancer magister*	FT	96 hr	> 41	41	This Study
Sand dollar, *Dendraster excentricus*	S	65 hr [3]	> 44	44	This Study
Blue mussel, *Mytilus galloprovincialis*	S	48 hr	> 47	47	This Study
Polychaete worm, *Neanthes arenaceadentata*	FT	96 hr	> 43	43	This Study
Vertebrates					
Sheepshead minnow, *Cyprinodon variegatus*	FT	96 hr	9.82	9.82	This Study
Silverside, *Menidia beryllina*	FT	96 hr	7.08	7.08	This Study
Speckled sand dab, *Citharichthys stigmaeus*	FT	96 hr	2.39	2.39	This Study
Algae					
Marine alga, *Skeletonema costatum*	S	96 hr	17.9	17.9	This Study

[1]S = static, FT = flow-through.

[2]GMAV = Genus Mean Acute Value (geometric mean).

[3]Pluteus development complete at this time; test terminated early.

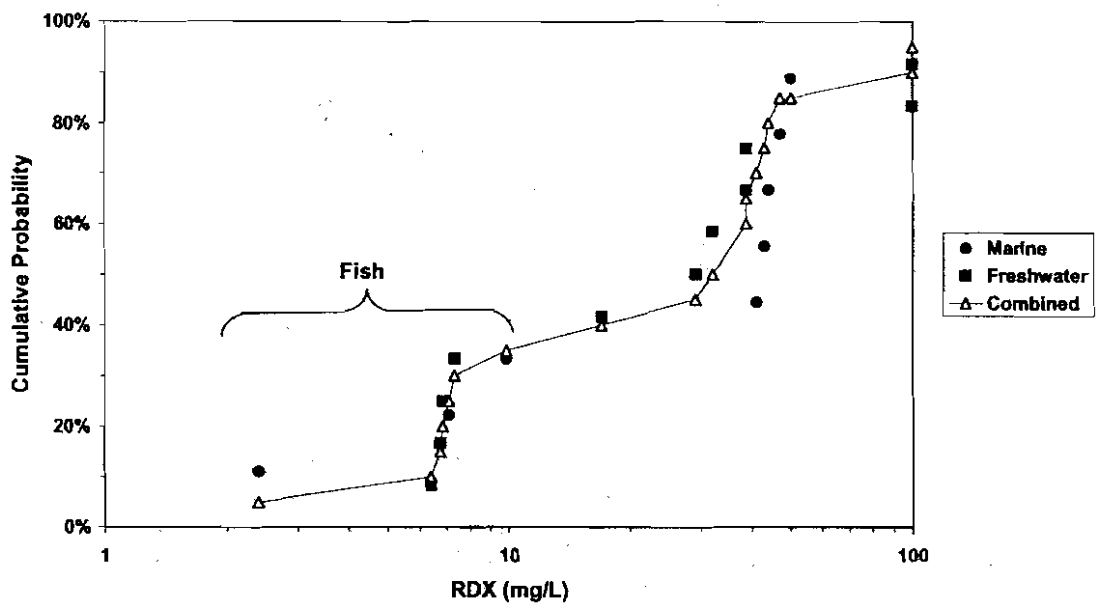

Figure 6-1. Acute Toxicity of RDX to Aquatic Animals. Data Represent Acute Toxicity Values (LC_{50} or EC_{50}) Plotted as a Function of their Cumulative Probability Distribution.

6.2 CHRONIC TOXICITY

The results of the chronic toxicity of RDX to aquatic animals (both freshwater and marine) are presented in Table 6-2. As per USEPA guidance (USEPA 1985), acceptable chronic toxicity data for both freshwater and marine organisms were combined, and ACRs calculated where possible (Table 6-2). Unlike earlier studies, RDX was not chronically toxic to *D. magna*. Compared to freshwater fish, *C. variegatus* was much less chronically toxic, with an ACR <1 (i.e., chronic value was higher than acute LC_{50} in Table 6-1). The final ACR equals 2.015, and it was calculated as the geometric mean of genus mean ACRs from *C. dubia*, *P. promelas* (mean of both values in Table 6-2), and *C. variegatus*.

6.3 WATER CHEMISTRY DATA

The water chemistry parameters that were measured as part of the studies described in Section 5 are presented in Table 6-3. All values were within acceptable ranges for each organism according to test protocols (ENSR 2003).

Table 6-2

Chronic Toxicity of RDX to Aquatic Animals

Test Species	Method[1]	Duration (days)	Observed Effects	NOEC[2] (mg/L)	LOEC[3] (mg/L)	Chronic Value (mg/L)	Acute to Chronic Ratio (ACR)[5]	Reference
Invertebrates								
Ceriodaphnia dubia	R, ST	7 d	survival/reproduction	3.6	6.0	4.7	3.6325	Peters et al. 1991
Chironomus tentans	FT, LC	23 d	growth/survival/ emergence			> 21		Bentley et al. 1977
Daphnia magna	FT, ELS	21 d	reproduction	2.2[4]	4.8[4]			Bentley et al. 1977
Daphnia magna	R, LC	21 d	reproduction	> 41	> 41	> 41		This Study
Paratanytarsus parthenogeneticus	FT, LC	17 d (?)	growth/survival			> 21		Peters et al. 1991
Fish								
Pimephales promelas	FT, ELS	30 d	growth	3.0	5.8	4.2		Bentley et al. 1977
Pimephales promelas	FT, PLC	240 d	survival	3.0	6.3	4.3	1.5349	Bentley et al. 1977
Pimephales promelas	FT, ELS	28 d	growth	1.4	2.4	1.8	7.1152	Burton et al. 1994
Cyprinodon variegatus	FT, ELS	28 d	survival/growth	9.7	21.4	14.4	0.6816	This Study

[1] R = static renewal, FT = flow-through, ST = short-term chronic, ELS = early life stage, LC = full life cycle, PLC = partial life cycle.

[2] NOEC = No Observable Effect Concentration (i.e., highest concentration at which no statistically significant adverse effect was observed).

[3] LOEC = Lowest Observable Effect Concentration (i.e., lowest concentration at which a statistically significant adverse effect was observed).

[4] Values should be interpreted with caution because effects were only observed between 7-14 d, and mortality exceeded 20 percent in some controls.

[5] Acute values obtained from same study, as per USEPA Guidance for deriving Ambient Water Quality Criteria.

Table 6-3

Water Chemistry Parameters

Test Species	Duration	Temperature (° C) (range)	Dissolved Oxygen (mg/L) (range)	pH (range)	Salinity (%) (range)
Invertebrates					
Americamysis bahia	96 hr	24.8 (24.4 – 25.0)	6.1 (5.6 – 6.4)	8.0 (7.9 – 8.1)	31.1 (30.0 – 32.0)
Cancer magister	96 hr	15.1 (14.5 – 15.9)	8.2 (8.0 – 8.3)	8.0 (7.8 – 8.1)	30.6 (30.0 – 31.0)
Dendraster excentricus	65 hr	16.5 (15.8 – 16.9)	8.0 (7.8 – 8.1)	8.0 (7.9 – 8.1)	30.4 (30.0 – 31.5)
Mytilus galloprovincialis	48 hr	15.8 (15.3 – 16.3)	7.9 (7.7 – 8.0)	8.0 (7.9 – 8.1)	30.5 (30.0 – 32.0)
Neanthes acrenaceadentata	96 hr	20.3 (20.0 – 20.8)	7.4 (6.8 – 7.6)	8.0 (7.8 – 8.1)	30.1 (29.5 – 30.5)
Daphnia magna	21 d	20.3 (19.8 – 23.2)	9.2 (9.0 – 9.4)	7.2 (7.0 – 7.4)	Not Applicable
Vertebrates					
Cyprinodon variegates	96 hr	24.5 (23.8 – 25.4)	6.3 (5.3 – 7.0)	8.0 (7.8 – 8.1)	31.3 (30.0 – 32.0)
Cyprinodon variegates	28 d	24.7 (24.0 – 25.4)	6.3 (6.0 – 6.8)	7.9 (7.4 – 8.1)	30.0 (29.0 – 31.0)
Menidia beryllina	96 hr	23.4 (22.9 – 23.7)	6.1 (5.2 – 7.0)	7.9 (7.8 – 8.0)	32.5 (31.5 – 33.0)
Citharichthys stigmaeus	96 hr	14.7 (14.5 – 15.0)	7.7 (7.4 – 8.0)	8.0 (7.9 – 8.2)	30.9 (30.5 – 31.5)
Algae					
Skeletonema costatum	96 hr	20.7 (20.2 – 21.1)	Not Reported	8.3 (8.0 – 8.7)	30.6 (30.0 – 32.0)

7.0 TOXICITY REFERENCE VALUE/CRITERIA CALCULATIONS AND CONCLUSIONS

7.1 TOXICITY REFERENCE VALUE CRITERIA CALCULATIONS

As we reviewed in Section 2, the primary basis of a TRV calculated according to USEPA guidance for derivation of AWQC is the Final Acute Value (FAV). This FAV is the concentration below which all but 5 percent of the most sensitive species would be protected from significant mortality from short-term RDX exposures. According to USEPA guidance, this number is calculated on the basis of the four lowest Genus Mean Acute Values (GMAVs) and the total number of available GMAVs (Section 2.2).

For marine organisms, the four lowest GMAVs consisted of all three fishes tested and one invertebrate GMAV (*C. magister*; Table 7-1). These data result in an FAV of 0.860 mg RDX/L which is almost 7 times lower than an FAV based on freshwater species (Table 7-1; ENSR 2003). This comparatively low marine FAV results from the low LC_{50} for *C. stigmaeus*, and because only three acutely sensitive marine species were tested (i.e., the crab *C. magister* was not acutely sensitive, and so its GMAV is set by default to maximum RDX solubility). However, given the strong overlap in species sensitivity (Figure 6-1) it is reasonable to combine freshwater and marine acute data to derive a single FAV. Using this combined dataset, the four lowest GMAVs are for three freshwater fish and the marine fish *C. stigmaeus* (Table 7-1), which yield an FAV of 2.719 mg RDX/L; this is only about two times lower than an FAV derived using freshwater data alone. To complete acute TRV calculations, the FAV is divided by two to derive the "acute criterion" or the Criterion Maximum Concentration (CMC; Section 2.6). For RDX, CMCs range from 0.430 mg RDX/L using marine data alone to 3.096 mg RDX/L for freshwater species (Table 7-2) . Using a combined dataset yields an intermediate CMC of 1.360 mg RDX/L (Table 7-2).

Table 7-1

Lowest Four GMAVs for the Marine Only, Freshwater Only, and Combined Datasets

Test Species	GMAV (mg RDX/L)	FAV (mg RDX/L)	Reference
Marine only			
Cancer magister	41		This Study
Cyprinodon variegatus	9.82		This Study
Menidia beryllina	7.08		This Study
Citharichthys stigmaeus	2.39		This Study
		0.860	
Freshwater only			
Ictalurus punctatus	7.3		Bentley et al. 1977
Pimephales promelas	6.84		Bentley et al. 1977; Liu et al. 1983; Burton et al. 1994b
Lepomis macrochirus	6.75		Bentley et al. 1977
Oncorhynchus mykiss	6.4		Bentley et al. 1977
		6.193	
Combined			
Pimephales promelas	6.84		Liu et al. 1983
Lepomis macrochirus	6.75		Bentley et al. 1977
Oncorhynchus mykiss	6.4		Bentley et al. 1977
Citharichthys stigmaeus	*2.39*		This Study
		2.719	

For most chemicals, chronic TRVs or criteria are usually calculated from applying an ACR to the FAV (Section 2.6), and this ACR is typically derived using both marine and freshwater species according to USEPA guidance (USEPA 1985). Using the final ACR of 2.015 we derived for RDX in Section 6.2, chronic TRVs (calculated as a "Criterion Continuous Concentration" or CCC) ranged from 0.427 to 1.349 mg RDX/L, depending on whether freshwater or marine acute data were combined (Table 7-2).

The final step in TRV derivation is to compare the calculated CMC and CCC values to any other available toxicity data (i.e., data not generated according to fully "acceptable" methods) to ensure that most important organisms would be protected by the calculated TRV (Section 2.6). For RDX, the calculated TRV concentrations were lower than toxicity endpoints reported in other marine organism studies, meaning that these TRVs are also protective of these other organisms. In these studies (detailed in Section 4), Nipper et al. (2001) found no adverse effects at maximum RDX solubility concentrations in short-term embryo-larval tests with redfish (*S. ocellatus*) and sea urchin (*A. punctulata*). Sublethal effects of RDX were observed in a 7-d chronic test with polychaete worms (*D. gyrociliatus*) at 26 mg RDX/L, and in an algal germination and early growth test (*U. fasciata*) in as low as 8.1 mg RDX/L, but these would still be protected by the acute and chronic TRVs proposed here (Table 7-2).

Table 7-2

Acute and Chronic TRV Calculations Using AWQC Methods

Acute dataset	FAV[1] (mg RDX/L)	CMC[2] (mg RDX/L)	ACR[3] (mg RDX/L)	CCC[4] (mg RDX/L)
Marine only	0.860	0.430	2.015	0.427
Freshwater only	6.193	3.096	2.015	3.073
Combined	2.719	1.360	2.015	1.349

[1]FAV = Final acute value.

[2]CMC = Criterion maximum concentration, or "acute criterion" (FAV/2).

[3]ACR = Acute-to-chronic ratio; geometric mean of all ACRs from Table 2.

[4]CCC = Criterion chronic concentration, or "chronic criterion" (FAV/ACR).

7.2 CONCLUSIONS

Using the combined marine and freshwater toxicity datasets, an acute TRV of 1.360 mg RDX/L would be considered protective of aquatic life according to methods used in derivation of national AWQC (USEPA 1985). Similarly, a chronic TRV of 1.349 mg RDX/L would be considered protective of aquatic life. These TRVs are also lower than available toxicity data for marine (Table 6-1) and freshwater plants (Table 3-1), and so would be protective of plant life according to USEPA guidance (USEPA 1985). Because RDX is not likely to bioaccumulate to a significant degree (ENSR 2001), these TRVs also would be considered protective of organisms dependent on aquatic life for food.

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Toxicity Test Reports

ATTACHMENT 2

CALCULATION OF SURFACE WATER QUALITY CRITERION FOR

RDX

PUBLIC WATER SUPPLY

Derivation of a Surface Water Quality Standard for RDX

Surface water quality standards (WQS) for RDX were calculated according to the guidance provided in Indiana Administrative Codes 327 IAC 2-1-8-5 and 2-1-8-6. Since RDX is classified as a carcinogen by the U.S. EPA, the derivation of the WQS was performed primarily according to Code 327 IAC 2-1-8-6 (Determination of concentrations providing an acceptable degree of protection to public health for cancer). The following general formula for deriving a WQS for carcinogens is provided in 327 IAC 2-1-8-6:

$$C = \frac{D \times W_h}{WC + (F \times BCF)} \quad \text{(Equation 1)}$$

Where:

C = derived surface water quality standard for RDX
D = dose
W_h = average human adult body weight = 70 kg
WC = daily water consumption
= 0.01 L per day for surface water not protected for drinking water supply
= 2 L per day surface water protected for drinking water supply
F = daily fish consumption rate = 0.0065 kg per day
BCF = water to fish tissue bioconcentration factor

The dose (D) can be derived using one of several options provided in 327 IAC 2-1-8-6 (b) (1) (A), (B), and (C). Subsection (B) states that the goal for cancer "shall be a concentration estimated to cause one (1) additional cancer over the background rate in one hundred thousand (100,000) individuals exposed to that concentration". This corresponds to a 1×10^{-5} cancer risk.
Subsection (B)(ii) indicates that the dose (D) can be determined by dividing the cancer slope factor, known as q_1^* (or CSF) by 1×10^{-5}. Therefore, the U.S. EPA CSF for RDX currently published in IRIS ($0.11\ (mg/kg/day)^{-1}$) was used to calculate D. This CSF was has been used to assess risks and to derive cleanup concentrations in other media at Crane and it is appropriate that the CSF be used to derive the WQS for surface water. This CSF has also been used by U.S. EPA Regions 3 and 9 to develop their soil and water remediation goals. Using the method recommended in Subsection (B)(ii), the value of D is:

$$D = \frac{1 \times 10^{-5}}{0.11 (mg/kg/d)^{-1}} \quad \text{(Equation 2)}$$

$$D = 9.1 \times 10^{-5}\ mg/kg/day$$

The BCF for RDX was calculated according to Code 327 IAC 2-1-8-7. There are a number of BCFs for RDX published in the literature. For example TOXNET (online at http://toxnet.nlm.nih.gov/) provides a range of measured BCFs of 4 to 5.9 L/kg for RDX. However, according to this rule, the measured BCFs must be normalized by the percent lipid content of the fish used in the measurements. Since the percent lipid content was not known, it was necessary to estimate the BCF using equations provided in Code 327 IAC 2-1-8-7.

The 1st step in deriving a BCF is to derive a calculated BCF from the octanol/water partition coefficient (K_{ow}) using the following equation:

$$\text{Log } BCF_c = 0.847 \log K_{ow} - 0.628 \quad \text{(Equation 3)}$$

Where:
BCFc = the calculated BCF
Log Kow for RDX = 0.87 (TOXNET, online, May 2005)

Using Equation 3, BCF_c = 1.28 L/kg

The 2nd step in deriving a BCF is to normalize BCF_c for lipid content using the formula:

$$BCF_f = BCF_c\ (9.6/4.8)\ \text{(Equation 4)}$$

Where BCF_f = the final bioconcentration factor = 1.28 x 2 = 2.56 L/kg

Based on the above equation and exposure factors, the following surface water WQS were calculated for RDX:

- WQS for surface water protected for drinking water supply = 3 µg/L
- WQS for surface water not protected for drinking water supply = 240 µg/L

<u>Example Calculation for Protected Water Supply</u>

$$C = \frac{9.1\text{x}10^{-5}\ \text{x}\ 70\,\text{kg}}{2\text{L/d} + (0.0065\ \text{kg/day x } 2.56\ \text{L/kg})}$$

$$= \frac{0.00636\ \text{mg}}{2.017\ \text{L}}$$

$$= \mathbf{0.003\ mg/L}$$

$$= \mathbf{3\ \mu g/L}$$

Note that a WQS could also be calculated according to 327 IAC 2-1-8-5 for the noncarcinogenic health effects of RDX (on the prostate). However, calculations show that the noncarcinogenic WQS would be approximately 35 times greater than WQS based on cancer. Only the more conservative carcinogenic WQS (3 µg/L) is presented.

ATTACHMENT 3

AVERAGE MONTHLY FLOW RATES

EAST FORK WHITE RIVER

1993 – 2003

USGS 03373500 East Fork White River at Shoals, Indiana												
Monthly mean streamflow (gpm)												
Year	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
1993	4,681,309	2,733,382	5,762,543	6,157,515	2,798,462	2,017,047	1,937,604	1,123,424	1,652,596	2,576,291	8,245,029	4,798,005
1994	3,309,681	5,179,512	2,118,483	5,359,044	5,385,974	894,521	1,028,721	440,303	342,907	225,313	601,434	2,093,797
1995	2,016,598	2,957,797	3,358,155	4,389,120	8,357,236	3,530,057	1,214,088	2,159,327	383,302	363,104	514,809	1,344,698
1996	4,991,003	2,359,057	4,618,473	6,750,421	15,762,951	6,799,792	2,302,953	826,747	750,895	731,146	1,731,142	5,745,039
1997	4,622,961	5,134,629	7,167,834	4,171,437	3,766,591	8,657,953	1,438,055	1,066,423	492,817	223,518	214,092	531,416
1998	1,847,389	2,252,684	3,596,933	8,339,283	6,220,800	8,007,148	3,221,261	1,905,737	391,830	293,536	324,056	776,029
1999	5,008,956	6,117,569	4,739,657	2,325,843	1,796,222	954,664	657,538	246,408	162,477	157,091	142,279	284,110
2000	1,251,341	3,118,479	2,425,035	4,157,074	1,889,579	3,196,576	1,646,313	1,995,055	1,099,188	2,707,350	2,281,409	3,976,195
2001	1,618,934	4,375,655	1,819,562	1,149,457	1,116,243	2,397,656	1,916,060	756,281	1,080,785	3,940,289	2,776,918	9,694,753
2002	2,537,243	3,852,767	5,789,922	7,872,499	13,339,262	3,529,608	1,138,685	432,673	352,781	549,818	1,250,444	2,292,630
2003	3,349,627	3,106,361	5,493,694	3,091,998	5,964,517	3,505,820	3,496,844	1,021,989	1,877,012	ND	ND	ND

Source: USGS Website (http://nwis.waterdata.usgs.gov/in/nwis/monthly/?site_no=03373500&agency_cd=USGS)

June 9, 2005

Water System Number	Water System Name	Intake Type	Source Type	Principal City Served
ACTIVE				
IN5251002	CRANE WATER WORKS	C	GW	CRANE
IN5251003	CRANE DIV, NAVAL SURFACE WARFARE CENTER	C	SW	CRANE
IN5251004	EAST FORK WATER	C	GW	SHOALS
IN5251005	LOOGOOTEE WATER WORKS	C	GW	LOOGOOTEE
IN5251006	PERRY WATER SYSTEM, INC.	C	GW	LOOGOOTEE
IN5251007	SHOALS WATER COMPANY	C	GW	SHOALS
INACTIVE				
IN2140006	HICKORY RIDGE AMISH SCHOOL	NTNC	GW	
IN2140015	WASHINGTON BOAT CLUB AND CAMGROUND 0	NC	GW	
IN2140828	FAIRVIEW AMISH SCHOOL	NC	GW	
IN2510002	U.S. GYPSUM CO.	NTNC	GW	
IN2510800	MARTIN STATE FOREST	NC	SW	SHOALS
IN2510801	ZION AMISH CHRISTIAN SCHOOL	NTNC	GW	SHOALS
IN2510812	IMMANUEL MISSIONARY CHURCH	NTNC	GW	SHOALS
IN5251001	BURNS CITY WATER DEPT	C	SW	LOOGOOTEE
IN5251008	EAST FORK WATER COMPANY-	C	GW	

Intake Type
C — (Community) Serves at least 15 service connections used by year-round residents or regularly serves 25 year-round residents.
NC — (Transient Non-Community) Regularly serves at least 25 non-residential individuals (transient) during 60 or more days per year.
NTNC — (Non-Transient Non-Community) Serves at least the same 25 non-residential individuals during 6 months of the year.

Source Type
GW — Groundwater
SW — Surface water

ATTACHMENT 4

SPRING A, SPRING C, AND LITTLE SULPHUR CREEK

RDX CONCENTRATIONS AND FLOW RATES

SUMMARY OF RDX DATA FROM LITTLE SULPHUR CREEK BELOW ABG

Tables 1, 2, 3, and 4 present RDX and flowrate data by location. Table 1 presents data for Little Sulphur Creek (LSC) surface water samples collected at the boundary (Creek B). Table 2 presents data for Little Sulphur Creek (LSC) surface water samples collected below the discharge from Spring A. Table 3 presents data for Spring A. Table 4 presents data for Spring C.

The September 1999 Ground Water Monitoring Plan called for eight rounds of creek samples to be taken in support of the monitored natural attenuation demonstration project. However, nine rounds of samples were taken, as shown in Tables 1 and 2. Tables 3 and 4 present data from Springs A and C to date. Monitoring events 10 through 18 are post-creek sampling.

Table 1
RDX RESULTS FOR CREEK B
NSWC CRANE
CRANE, INDIANA

EVENT	QTR-YR	DATE	RESULTS (µg/L)	ESTIMATED QUANTITY (GPM)
1	4-98	11/05/98	20	40
2	1-99	02/28/99	0.97	500
3	2-99	05/20/99	3.1	550
4	3-99	09/13/99	3.8	20
5	4-99	12/28/99	19	10
6	1-00	03/08/00	7.2J	100
7	2-00	06/09/00	11	20
8	3-00	09/20/00	11	50
9	4-00	12/12/00	1.5	300

QTYR-YR = quarter –year.
GPM = gallons per minute.
U = undetected at the reported limit.

Table 2
RDX RESULTS FOR CREEK A
NSWC CRANE
CRANE, INDIANA

EVENT	QTR-YR	DATE	RESULTS (µg/L)	ESTIMATED QUANTITY (GPM)
1	4-98	11/05/98	28	20
2	1-99	02/28/99	1.6	800
3	2-99	05/20/99	3.6	300
4	3-99	NA	NA	Dry
5	4-99	01/10/00	11 P	50
6	1-00	03/8/00	4.5	50
7	2-00	06/9/00	23	20
8	3-00	09/20/00	17	50
9	4-00	12/18/00	1.1 P	300

QTYR-YR = quarter –year.
GPM = gallons per minute.
U = undetected at the reported limit.

Table 3
RDX RESULTS FOR SPRING A
NSWC CRANE
CRANE, INDIANA

EVENT	QTR-YR	DATE	RESULTS (µg/L)	ESTIMATED QUANTITY (GPM)
1	4-98	11/05/98	63	6
2	1-99	02/28/99	1.5	150
3	2-99	05/20/99	6.3	250
4	3-99	09/13/99	120	4
5	4-99	01/11/00	33	25
6	1-00	03/29/00	8.0	50
7	2-00	06/26/00	11	50
8	3-00	09/29/00	9.2	NR
9	4-00	12/18/00	1.9	150
10	1-01	04/10/01	10	50
11	2-01	07/10/01	5.0	50
12	3-01	09/17/01	110 PE	25
13	4-01	12/20/01	3.2	NR
14	1-02	03/12/02	3.4 P	70-80
15	2-02	06/20/02	36	50
16	3-02	09/10/02	140 E	NR
17	4-02	12/9/02	73	75
18	1-03	03/07/03	1.6	150

QTYR-YR = quarter -year
GPM = gallons per minute
U = undetected at the reported limit.

Table 4
RDX RESULTS FOR SPRING C
NSWC CRANE
CRANE, INDIANA

EVENT	QTR-YR	DATE	RESULTS (µg/L)	ESTIMATED QUANTITY (GPM)
1	4-98	11/05/98	1.4	6
2	1-99	02/28/99	1.7	300
3	2-99	05/20/99	1.9	175
4	3-99	09/13/99	0.8	4
5	4-99	01/11/00	6.0	10
6	1-00	03/14/00	1.8	50
7	2-00	06/28/00	4.6 P	50
8	3-00	09/29/00	3.2	25
9	4-00	12/19/00	1.7	100
10	1-01	04/10/01	1.4	50
11	2-01	07/10/01	4.9	50
12	3-01	09/17/01	3.6 P	25
13	4-01	12/18/01	1.6	100-125
14	1-02	03/11/02	2.5 P	75
15	2-02	06/24/02	3.9	50
16	3-02	09/09/02	0.6 U	20
17	4-02	12/12/02	3.1	50
18	1-03	03/10/03	1.7	125

QTYR-YR = quarter -year
GPM = gallons per minute
U = undetected at the reported limit.

Two rounds of surface water samples were taken during the ABG Old Jeep Trail/Little Sulphur Creek RFI in 2001. An attempt was made during this RFI to collect low and high-flow samples from Little Sulphur Creek (LSC). Samples for 03SW14 through 03SW19 are presented in Table 5. Samples locations for 03SW17 and 03SW19 are very close to the Creek A and Creek B samples locations, respectively. Sample 03SW15 is located in LSC just below where Spring C feeds in and therefore determines the contribution of RDX from Spring C. It's worth noting here that the reported RDX values are not attributable to sediment, since RDX was not detected in shallow sediments. RDX was detected in deep sediment only at two locations (03SW11 and 03SW18) at low (J) levels (0.45 mg/kg each). This data is plotted in the associated Excel Spreadsheet as SW01 (06/11/01 event) and SW02 (09/09/01 event).

Table 5
LSC RDX RESULTS FROM RFI
(Upstream → Downstream)
NSWC CRANE
CRANE, INDIANA

Sample	Distance Above Creek B[1] (feet)	Date: 06/11/01		Date: 09/09/01	
		Results	Flow	Results	Flow (gpm)[2]
03SW14	4,640	NA	Dry	0.35U	3
03SW15	3,670	2.4J	30	0.84	30
03SW16	2,320	2.3	45	0.62	45
03SW17	1,440	11	170	63	180
03SW18	590	9.5	180	39J	200
03SW19	0	7.7J	200	10	250

1 - Approximate distances as measured along Little Sulphur Creek – not in a straight line.
2 - Gallons per minute

SUMMARY:

Spring A causes LSC to consistently exceed the 0.61 µg/L RBTL at the boundary.

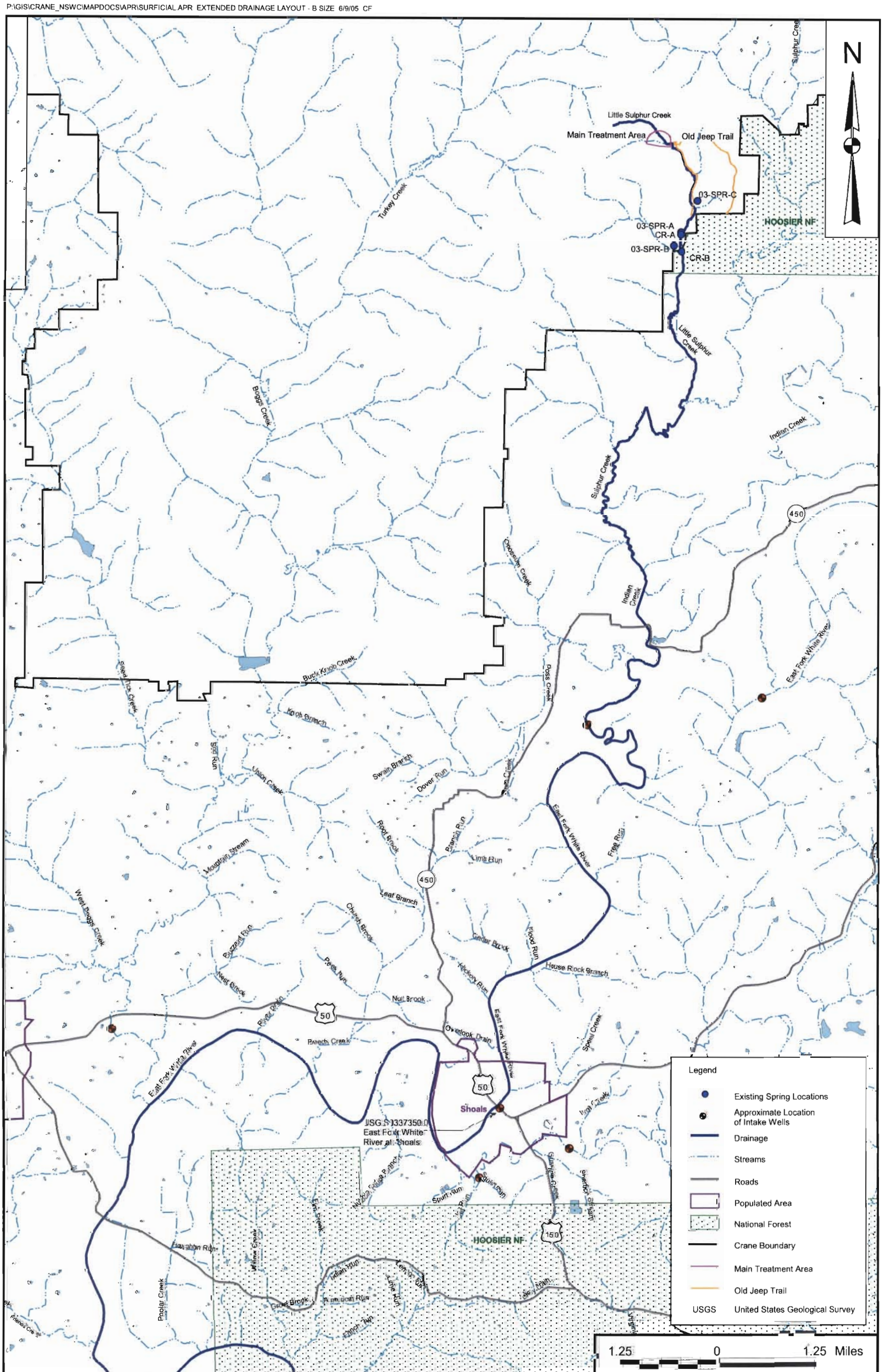
A quick, intuitive look at the change in concentrations from Creek A to Creek B for the eight sampling events available shows the following:

a. RDX concentrations changes ranged from an increase of 2.7 [monitoring event 6 (03/08/00)] to a decrease of 53 µg/L (SW02, 09/09/01).

b. RDX concentrations decreased from Creek A to Creek B in 7 of the 8 events.

c. The highest value of RDX in Creek B was 20 µg/L (110598). The lowest was 0.97 µg/L (022899). The average RDX concentration of 9 samples from Creek B was 8.3 µg/L.

P:\GIS\CRANE_NSWC\MAPDOCS\APR\SURFICIAL.APR EXTENDED DRAINAGE LAYOUT - B SIZE 6/9/05 CF



DRAWN BY	DATE
C. FOSTER	6/8/05
CHECKED BY	DATE
V. PLACHY	6/8/05
COST/SCHEDULE-AREA	
SCALE	
AS NOTED	

NAVFAC

SURFACE WATER FEATURES ASSOCIATED
WITH LITTLE SULPHUR CREEK
NSWC CRANE
CRANE, INDIANA

CONTRACT NUMBER	
8383	
APPROVED BY	DATE
APPROVED BY	DATE
DRAWING NO.	REV
FIGURE 1	0

P:\GIS\NSWC_CRANE\APR\SWMU3.APR SB 53-121 LOCATION LAYOUT CF 6/8/05

CONTRACT NO.
8383

APPROVED BY ___ DATE ___

APPROVED BY ___ DATE ___

DRAWING NO. FIGURE 2

REV ___

MAIN TREATMENT AREA AERIAL
SOIL BORING LOCATIONS
SWMU 3
CRANE NSWC
CRANE, INDIANA

NAVFAC

DRAWN BY C. FOSTER DATE 3/28/05

CHECKED BY V. PLACHY DATE 6/8/05

COST/SCHED-AREA

SCALE AS NOTED

Enclosure (2)
Minutes for the
March 30, 2005 Meeting

MINUTES FOR MEETING HELD MARCH 30, 2005
ON
EPA COMMENTS DATED MARCH 23, 2005
ON
CONCEPTUAL SITE MODEL AND FIELDWORK REPORT FOR AMMUNITION BURNING GROUNDS

A meeting was held on March 30, 2005 among the Navy, Crane Navy and Army, TtNUS, and U.S. EPA Region 5 to review EPA's comments, dated March 23, 2005 on the SWMU 3 Conceptual Site Model and proposed responses. Following is a list of the attendees.

Bill Gates – Southern Divisions
Tom Brent – NSWC Crane
Mike Singletory – Southern Division
Keith Henn – TtNUS
Ralph Basinski – TtNUS
Ramanauskas – U.S. EPA Region 5
Doug Johnson – Crane Army

Following is a summary of the results of the meeting.

Protected User: Selecting a remedy for the "current user" not future user and addressing the site further upon RCRA closure of the site is acceptable. However, the April 2004 data should be used to assess risks to current actual/potential receptors.

Land Use Controls: LUCs must include drinking water restrictions on BC/BC groundwaters.

Source Treatment: While EPA generally prefers source treatment, Peter Ramanauskas he is agreeable with the recommendations in the CSM based on circumstances that are specific to the ABG. The CSM recommends that evaluations focus on the "end-of-pipe" spring waters

Source / Groundwater Treatment: EPA would like to see further support in the CMS why source or groundwater treatment will not benefit the site.

Site-Specific Criteria: Site specific criteria for COC's may be generated.. EPA recognizes that the RDX criteria of 0.61 ug/L is a Region 9 value. More appropriate site-specific criteria may include criteria established by Indiana water quality standards.

Point-of-Compliance: The point-of-compliance (POC) must be established for groundwater and surface water. The POC must be established at locations were current actual/potential risks exist. Indiana water quality standards will be used to determine uses for surface waters. (Note: This subject is addressed in the response to EPA's march 23, 2005 Comment No. 3).

CMS Consistency: EPA would like to see consistency in formats for NSWC Crane CMS studies. The MGBG CMS was cited as an example of a good format.