

A Critical Analysis of the Draft Report, “Amendments to the Water Quality Control Plan for the Sacramento River and San Joaquin River Basins for the Control of Diazinon and Chlorpyrifos Runoff into the Lower San Joaquin River” (Karkoski et al. 2004) and Supporting Documents

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Specific Comments Related to the Three Stated Issues

Issue 1:

Is the proposed rule based upon sound scientific knowledge, methods, and practices when it relies on the use of the California Department of Fish and Game’s [CDFG] chlorpyrifos water quality criteria document as the basis for site-specific water quality objectives?

The validity of relying on the CDFG WQ (water quality) criteria as a basis for promulgating the San Joaquin River (SJR) TMDLs can be assessed by analysis of three issues. First, has CDFG faithfully followed EPA’s established protocol (guidelines) for determining national water quality criteria? Second, given the best available acute and chronic toxicity data, is CDFG WQ criteria protective enough such that when extrapolated to a TMDL it is likely to be protective of at least 95% of the species present in the aquatic ecosystem needing protection? Third, is there a more conservative scientifically sound method that could have substituted for the CDFG? The latter issue could be restated as “Is there another method relying on a risk characterization approach that could be used to validate the adequacy of the CDFG criteria?”

First, CDFG was transparent in its derivation of the proposed WQ criteria and they faithfully followed the EPA 1985 Guidance document. CDFG analyzed individual toxicity studies and gave a rationale for either rejecting or accepting a study for calculation of the WQ criteria. Given that the WQ criteria are probabilistic in nature and are based on a range of species it is very conservative and should be quite protective of indirect effects through food web disruption as well as acute and reproductive toxicity.

Second, many aquatic toxicology studies have thus far concurred that Cladocerans (specifically the species *Ceriodaphnia dubia*) are the most susceptible organisms to acute toxicity by diazinon and chlorpyrifos. Given the range of concentrations posted in Appendix C of the Karkoski report and the general trend for reductions in diazinon and chlorpyrifos use and residue detections, the proposed CDFG WA criteria should be adequately protective. Indeed, projections of the proportion of water samples annually exceeding the proposed WQ criteria indicated comparatively few violations (Karkoski report, Tables 1.5 – 1.8). Bioassays of *Ceriodaphnia* in water collected from the San Joaquin River (SJR) at Vernalis during 1993 showed that toxicity did not occur when diazinon concentrations were below ~150 ng/L (Kuivila and Foe 1995). Similar lack of

toxicity was observed in water collected from the same location during February of 1994 and 1995 (Werner et al. 2000). All of the chlorpyrifos concentrations (obtained from Appendix C) were less than 0.025 µg/L, so acute toxicity would not have been expected. However, many of the concentrations detected were greater than 0.014 µg/L, but reproduction was not affected (Werner et al. 2000). In sum, empirical evidence suggests that the WQ criterion for chlorpyrifos is more than adequately protective of the most sensitive organisms in the SJR.

Third, the EPA risk characterization method can be used to ground truth (validate) the derivation of the CDFG WQ criteria. Based on the indicator species chosen by EPA for conducting its typical ecological risk characterization, similar conclusions of adequate protection could be drawn in comparison to the CDFG criteria. For example, in the Registration Eligibility Decision document (RED) for chlorpyrifos, EPA (2000) identified *Daphnia magna* as the most sensitive organism for both acute toxicity (LC50 @ 0.10 µg/L) and chronic toxicity (NOAEC @ 0.04 µg/L). EPA uses a deterministic risk quotient (RQ) approach (ratio of exposure concentration to toxicity endpoint) to determine if exposure exceeds their levels of concern (LOC). For restricted use products the RQ should not exceed 0.2 (i.e., a 5-fold safety factor is applied to the acute toxicity endpoint). Thus, any water concentration estimated to exceed 0.02 µg/L would be above EPA's LOC and thus require risk mitigation. Pertinently, 0.02 µg/L is virtually the same concentration as the CDFG criterion. For endangered species, the RQ should not exceed 0.1, so any chlorpyrifos concentrations of 0.01 µg/L would exceed EPA's LOC. For assessment of the risk of chronic toxicity, EPA's LOC are equal to the NOAEC. The concentration of 0.04 µg/L is about three times higher than the proposed CDFG WQ criterion. Thus, the CDFG WQ criterion of 0.014 µg/L is likely to be below the threshold for any toxicological effects on the most sensitive invertebrates.

With the EPA RQ approach, a similar analysis to that used for chlorpyrifos can be applied to the proposed diazinon criterion to determine its validity. EPA (1999) considered the scud (*Gammarus fasciatus*) to be the most susceptible invertebrate exhibiting an LC50 of 0.2 µg/L. Application of the equivalent 5-fold safety factor indicates that any diazinon residues greater than 0.04 would exceed EPA's LOC. Thus, the proposed criterion of 0.16 µg/L is within a factor of ~4. The 21-day NOAEC in a chronic toxicity assay with *Daphnia magna* was 0.17 µg/L. This concentration is about twice as high as the 4-day WQ criteria proposed in the TMDL plan.

In conclusion, reliance on the CDFG WQ criteria for implementation of the SJR Basin TMDLs would give a degree of protection that seems reasonably validated by EPA's ecological risk characterization approach. Pertinently, ecological impairment is unlikely even if some toxicity should occur in the SJR drainage areas. Studies with chlorpyrifos in outdoor field ditches showed definitive recovery of *Daphnia pulex* populations after exposure to concentrations circa the LC50 (Van der Hoeven and Gerritsen 1997). Outdoor mesocosm studies using ditches and artificial pools exposed to different diazinon or chlorpyrifos concentrations also show recovery of invertebrate populations after initial declines (Giddings et al. 1996; Van der Brink 1996).

Issue 2:

Use of a toxic unit formula for the loading capacity and allocation to account for the additive effects of diazinon and chlorpyrifos.

The formula for joint toxicity that is used to determine whether the loading capacity is exceeded seems to be a derivative of the formula used to determine whether the response of an organism simultaneously exposed to two or more compounds having identical biochemical modes of action is the result of an interaction that is neutral, additive, synergistic, or antagonistic (e.g., Bailey et al. 1997). The formula is not meant to indicate the degree of exposure but whether simultaneous exposures result in interactive responses (and the nature of those responses). Thus, the formula proposed for determining whether the load capacity of simultaneously occurring residues of diazinon and chlorpyrifos exceeds the water quality criteria does not reflect additive toxicity. Indeed, the formula is divorced from measurement of toxicity by virtue of the false assumption that residues at the water quality standard, which has a 2-fold uncertainty factor built in, are linearly related to toxicity. However, the interactive response to simultaneous exposures of diazinon and chlorpyrifos residues is likely neutral when the concentrations are below a threshold for any effect. A critical analysis of the cited paper by Bailey et al. (1997) reveals the following statement at the end: “No mortalities were observed in concentrations of the mixtures that totalled between 0.47 and 0.58 total TUs.” In other words, when the concentrations were approximately 25% of the LC50 (i.e., ~0.25 TUs), mortality was not observed. Thus, assuming that all concentrations are additive is not valid when they are below a certain threshold of response.

The application of the formula for calculating toxicity units to account for additive effects is inconsistent with other determinations of joint toxicity from residues of compounds having different potencies. The denominator in these formulas has to be based on a definitive toxicity endpoint, not a value that already includes a safety factor. The formula should be based on either the LC50 for the most sensitive species (wherein the confidence intervals are lowest) or a potency factor (i.e., a benchmark for a measurable effect). Furthermore, to estimate the combined residues, all residues should be expressed as an equivalency of a chosen reference compound. Choosing chlorpyrifos toxicity equivalences is logical considering that chlorpyrifos residues may contribute the most toxicity at locations such as SJR at Vernalis (Werner et al. 2000).

The inadequacy for prediction of effects by using the proposed toxic unit formula is illustrated in Table 1.7 on page 22 of Karkoski et al. 2000. During the years 1993, 1994, and 1995, 15%, 22%, and 7.1%, respectively, of the combined residue values exceeded an S of 1. However, examination of toxicity assays on water samples collected from the SJR at Vernalis (see Werner 2000) failed to show evidence of acute toxicity (with the exception of August 1993 samples) to *Ceriodaphnia* nor adverse reproductive effects. Thus, it is unlikely that joint toxicity was an important factor. Alternatively, the calculation of S by the formula given is not a valid indicator of ambient effects.

One solution to the dilemma of cumulating exposure to two or more compounds with the same biochemical mode of action is to normalize the potency of all residues. For example, if the potency of both diazinon and chlorpyrifos were known, then all diazinon residues could be transformed into chlorpyrifos equivalent residues. This procedure is analogous to human and ecological exposure assessment for dioxin congeners (as well as PCB congeners) (Van den Berg et al. 1998) and the human health cumulative exposure assessment for OP insecticides that was required under the mandates of the Food Quality Protection Act (EPA 2002). For both dioxin congeners and OP insecticides, the toxicity of any homologous compound is expressed as a toxic equivalency of a reference compound. The potency for a defined toxicological endpoint is first compared between a reference compound and the homolog of interest. The ratio of potencies constitutes a toxic equivalency factor (TEF in the case of dioxin congeners) or a relative potency factor (RPF in the case of OP insecticides). For dioxin congeners, TCDD is chosen as the reference compound and the residues of all other congeners multiplied by the appropriate toxic equivalency factor to express all congeners as TEQs (toxic equivalencies of TCDD). The specific toxicological endpoint for TCDD varies depending on whether mammals, birds, or fish are the subject of the exposure assessment (Van den Berg et al. 1998). For OP insecticides, the most sensitive toxicological endpoint is inhibition of acetylcholinesterase, and methamidophos is chosen as the reference compound (EPA 2002). All OP residues co-occurring in food, water, or at a residence are multiplied by the RPF to express all residues as methamidophos toxic equivalencies.

While it is most desirable to compare potencies among compounds based on a specific toxicological endpoint, earlier iterations of the RPF approach as applied to OP insecticides relied on the NOAEL for acetylcholinesterase inhibition in rodent subchronic (90-day exposure) or chronic (two-year exposure) dietary bioassays (NAS 1993). Of course, a cholinesterase inhibition database is not available for invertebrates, but the LC50 is a definitive toxicological endpoint that represents the response with greatest precision for the whole dose-response curve. For both diazinon and chlorpyrifos, the CDFG (Siepmann and Finlayson 2000) reported a GMAV (genus geometric mean) that could be used as a toxicological endpoint. To be most protective, the most sensitive species should be chosen, and for both OPs, *Ceriodaphnia dubia* data are available. Furthermore, because the biochemical mode of action for OP insecticides is similar, the slopes of the dose-response curves for both diazinon and chlorpyrifos are likely to be similar (assuming the test organisms are homozygous for susceptibility).

An alternative to choosing the LC50 for the most sensitive species in the database would be to choose the FAV. The FAV is lower than the observed toxicological endpoint in *Ceriodaphnia* testing, and thus it should be quite protective. Furthermore, the FAV is probabilistic in origin and considers numerous species, and thus it would be protective of ecologically relevant species in the SJR Basin. The FAV for diazinon was inferred to be 0.32 (based on Karkoski et al. 2004) and for chlorpyrifos the FAV was reported as 0.05 (Siepmann and Finlayson 2000). Thus, the RPF should be the ratio of FAV (chlorpyrifos) to FAV (diazinon) or 0.15625. Diazinon residues are changed into chlorpyrifos equivalents residues by multiplying all concentrations of diazinon by 0.15625. The resulting chlorpyrifos equivalent residues are added to the measured

chlorpyrifos residues to determine if the daily load (1-h or 4-day criteria) has been exceeded.

Use of the RPF approach to make a decision about exceedance of WQ criteria is illustrated in Table 1 using part of the residue dataset presented in Appendix C. Table 2 show the likely management decision based on the “joint toxicity” formula presented in Karkoski et al. (2004). The management decision regarding whether the loads have been exceeded are virtually the same, but the use of the RPF method would be on sound toxicological footing that is consistent with other exposure assessments that have been used for mixtures of compounds having identical biochemical modes of action.

Table 1. Illustration of the calculation of chlorpyrifos toxic equivalent residues for determining exceedance of the loading capacity when diazinon and chlorpyrifos co-occur.

Date	Time	Agency	Diazinon	Chlorpyrifos	Total Chlorpyrifos Toxic Equiv.	Exceed 1 hr?	Exceed 4 day?
08/04/93	10:15	USGS NAWQA-4	0.09	0.012	0.026	Yes	Yes
08/25/93	19:00	USGS NAWQA-4	0.026	0.008	0.012	No	No
02/13/00	5:30	USGS-CVRWQCB	0.075	0.012	0.024	No	Yes
02/13/00	12:00	USGS-CVRWQCB	0.03	0.012	0.017	No	Yes
02/13/00	23:00	USGS-CVRWQCB	0.036	0.005	0.011	No	No

Table 2. Illustration of the S (sums of ratios method proposed by Karkoski et al. 2004 to determine exceedance of the loading capacity when diazinon and chlorpyrifos co-occur.

Date	Time	Agency	Diazinon $\mu\text{g/L}$	Chlorpyrifos $\mu\text{g/L}$	1-hr Sum of Ratios (S)		4-day Sum of Ratios (S)	
					1 hr	Exceed?	4 day	Exceed?
08/04/93	10:15	USGS NAWQA-4	0.09	0.012	1.0	No	1.8	Yes
08/25/93	19:00	USGS NAWQA-9	0.026	0.008	0.5	No	0.8	No
02/13/00	5:30	USGS-CVRWQCB	0.075	0.012	0.9	No	1.6	Yes
02/13/00	12:00	USGS-CVRWQCB	0.03	0.012	0.7	No	1.2	Yes
02/13/00	23:00	USGS-CVRWQCB	0.036	0.005	0.4	No	0.7	No

Issue No. 3

Use of different diazinon water quality targets to account for invertebrate versus salmonid impacts.

Karkoski et al. (2004) point to the paper by Scholz et al. as evidence that the proposed diazinon water quality criteria may not be adequately protective of salmon, especially when sublethal effects are considered. At one point in the amendment report, Karkoski et al. (2004) seem to treat the lowest diazinon concentration tested by Scholz et al. (0.1 $\mu\text{g/L}$) as a LOAEC (p. 55, “...effects of diazinon have been observed at levels as low as 0.1 $\mu\text{g/L}$; p. 55, “The work by Scholz et al....suggests a possible lowest observed adverse

effect level to Chinook salmon of 0.100 µg/L.”). Elsewhere, 0.1 µg/L is viewed as a NOAEC (p. 56, “ The study by Scholz [2000] suggests that if diazinon levels do not exceed 0.100 µg/L, then salmon should be protected; p. 56, “...the 0.100 µg/L should be considered an instantaneous maximum...””). Critical examination of Scholz et al. 2000 shows the 0.1 µg/L to be an empirical concentration wherein no statistical difference (at $p \leq 0.05$) in behavioral endpoints (food strikes or swimming activity) was observed in exposed fish compared to the non-dosed control fish. Thus, the concentration 0.1 µg/L was an empirical NOAEC in the Scholz et al. (2000) study. However, the value of using 0.1 µg/L as a guide is dubious for the following reasons.

First, examination of the highest dose tested (10 µg/L) by Scholz et al. (2000) suggested that the behavioral endpoint of swimming activity cessation in response to an alarm pheromone did not differ significantly between unexposed and exposed fish. Thus, there was no linear dose-response effect, as would be expected from an anticholinesterase mechanism of action. For the other behavioral endpoint, food strikes per unit time, the concentrations of 1 and 10 µg/L did not differ from one another, again illustrating an endpoint that showed no clear dose-response effect. Indeed, on average, fewer food strikes were observed in fish exposed to 10 µg/L than in fish exposed to 1 µg/L.

Second, the ecological relevance of the endpoints chosen remains unclear. The behavioral endpoints are not an all-or-none-effect but a gradient of possible responses when the test salmon were exposed to their alarm pheromone. Indeed, unexposed fish also made “wrong” behavioral responses (i.e., swimming activity and food strikes continued, albeit at a lower rate), and the quantitative responses of exposed fish were not largely different from the responses of the unexposed fish. The magnitude of the observed endpoints should be examined relative to the normal response in the absence of alarm pheromone. In all cases, the expected response by all exposed fish was significantly different after alarm pheromone presentation in comparison to before pheromone presentation. In other words, both exposed and unexposed fish stopped swimming and engaging in food strikes in response to the pheromone. The behavior exhibited by the middle dosed group was at best modestly different than the behavior of the lower and higher dosed group. In sum, using behavioral toxicity based on the Scholz et al. (2000) study as the basis of a water quality criterion is presently inappropriate owing to a poor quantitative separation of responses.

Third, if there is indeed a behavioral effect associated with 1 µg/L diazinon, it is not clear that the mode of physiological action is due to inhibition of acetylcholinesterase. The possible binding of diazinon to a key olfactory receptor cannot be discounted. Sandhal et al. (2002) alluded to the hypothesis of a biochemical mechanism different than acetylcholinesterase inhibition. Thus, at this time, the observations of Scholz et al. (2000) cannot be put into the context of indirect effects owing to toxicity at lower trophic levels of the food web nor analogized to toxicity due to inhibition of acetylcholinesterase. More research is needed before attempting to incorporate behavioral toxicity into water quality objectives.

Fourth, an earlier study (Moore and Waring 1996) on the effects of diazinon on impairment of response of the olfactory rosette in Atlantic salmon suggests a NOAEC of 5 µg/L when overlapping 95% confidence intervals are examined (Felsot 2001). The study by Moore and Waring (1996) can be related to the Scholz et al. (2002) study because sex and alarm pheromones are first sensed by the nervous tissue of the rosette. Moore and Waring measured electrical recordings in the rosette in vivo while stimulating it with female sex pheromone. In summary, evidence from a second study suggests that not enough information is known about sublethal effects on pheromone-mediated behaviors to set a criterion specifically for salmon.

In conclusion, reliance on the Scholz et al. (2000) paper for a “special” water quality standard that protects salmon lacks validity. Reliance on behavioral toxicity endpoints based on the Scholz et al. (2000) study is presently inappropriate owing to a poor quantitative separation of responses. Minimal absolute differences occurred between diazinon treatments, and no clear dose-response relationship existed among all diazinon exposure concentrations. Furthermore, compared to normal behavioral responses in the absence of an alarm pheromone the exposed salmon at all diazinon concentrations reacted as expected by significantly ceasing swimming and food strike behavior. Finally, the behavioral endpoint seems to have ambiguous ecological relevance to protecting water quality in the SJR Basin. Instead, the most sensitive endpoint for protecting water quality in the SJR remains invertebrate toxicity. Invertebrate toxicity is based on many species responses, the final criteria are probabilistic in nature, and mortality is a definitive endpoint and an appropriate benchmark for rapidly reproducing species at the lower trophic levels.

Comments on Overarching Questions

The published literature contains several studies wherein toxicity of field-collected water from the SJR Basin were determined against *Ceriodaphnia* and split samples were used to assess either diazinon or chlorpyrifos concentrations. These kinds of studies are useful for validating the likelihood that the proposed WQ criteria (i.e., daily loads) are likely to be protective enough. However, the published studies cover water samples collected circa 1993-1995. Are there any current studies of combined toxicity assessment and residue analysis? Given that the levels of chlorpyrifos and diazinon seem to be declining and detected less frequently during the 2000’s than during the 1990’s, a current perspective on toxicity would be very useful.

The Karkoski et al (2004) report lacked any perspective on the current status of fish and invertebrate populations in the SJR and tributaries. Some published studies from the 1990’s have stated in their introductions that invertebrate abundance (and by implication diversity) has declined in the Basin waterways. However, it would be helpful to see some actual ecological monitoring data. Is there any credible evidence that invertebrate populations post 2000 are still deficient compared to pre-2000?

On a related issue, how important (or relevant) are the invertebrates that were used to determine the FAV by the CFDG? The report contains no discussion of the ecological relevance of the invertebrates chosen to derive the WQ criteria.

Regarding a related issue of ecological relevance, Karkoski et al. state that portions of the SJR are completely dry (p. 5). The whole rationale for promulgation of TMDLs for the insecticides was to mitigate adverse effects. But what could be more adverse to aquatic organisms than a portion of their habitat drying out? From the viewpoint of a thorough (and sound) scientific analysis, Karkoski et al. may want to communicate why focusing on the two insecticides for mitigation would be important in the light of the drastic habitat change on the SJR during the growing season or after.

The analysis of management practices focusing on pesticide use alternatives seems to have overlooked new classes of commercially available reduced risk insecticides/acaricides that are effective for dormant and in-season orchard pest control practices. For example, several chloronicotinyl (a.k.a. neonicotinoid) insecticides are now available and have excellent activity against Homoptera (scale insects, psylla) and some Lepidoptera. The general biochemical class known as METI (mitochondrial electron transport chain inhibitors) have excellent selectivity and are highly active as acaricides. Both of these classes of compounds are used at lower rates although they will be more expensive than the “conventional” treatments on a per acre basis.

The above issues notwithstanding, I conclude that Karkoski et al. (2004) have used the best available scientific methods for proposing the water quality criteria of the individual OP insecticides diazinon and chlorpyrifos. My main concern is the use of the formula for determining the potential exceedance of WQ criteria. I don't believe the formula actually represents a toxicologically based method. Instead, adoption of a relative potency factor approach whereby all residues can be transformed to chlorpyrifos equivalences would allow authentic cumulation of exposure. Finally, it is commendable that Karkoski et al. intend to protect endangered species like salmon by considering new toxicological endpoints. However, they have relied too heavily on the research presented in essentially one paper (i.e., Scholz et al. 2000) without a skeptical examination of its quantitative aspects. Thus, at this time a separate criterion for salmon cannot be supported by the state of the science. In short, if invertebrates are protected, all other species will be protected.

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