Attachment 2

A Review of the California Department of Public Health's Cost-Benefit Analysis in Support of a Proposed Primary Drinking Water Standard for Hexavalent Chromium (Cr VI)[1]

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October 9, 2013

Highlights

This report analyzes the cost-benefit analysis (CBA) prepared by the California Department of Public Health (CDPH) to inform regulatory decision-making in the matter of whether to set a primary drinking water standard for hexavalent Chromium (Cr VI), and if so, what level to set. Cost-benefit analysis is relevant because economic feasibility is a statutory factor the CDPH is required to consider in making this decision.

My analysis leads to the following eight major conclusions:

1. ***<u>Economic feasibility</u>* <u>lacks a statutory definition, so the California Department of Public Health is required to devise one that has a reasoned basis</u>.** If the CDPH were to rely on economic theory and practice to provide a reasoned basis, this task is actually quite easy: A regulatory action is economically feasible if aggregate social benefits exceeding aggregate social costs.

   This is summarized in Section I.A beginning on page 7 and explained in detail in Sections II.B though II.D beginning on page 12.

2. **<u>Taken at face value, the Department's cost-benefit analysis shows that none of the regulatory alternatives analyzed is economically feasible for households served by small water systems</u>.** For each MCL analyzed, costs to small systems (and by extension, their customers) significantly exceed the monetized value of the cancer risk reductions obtained.

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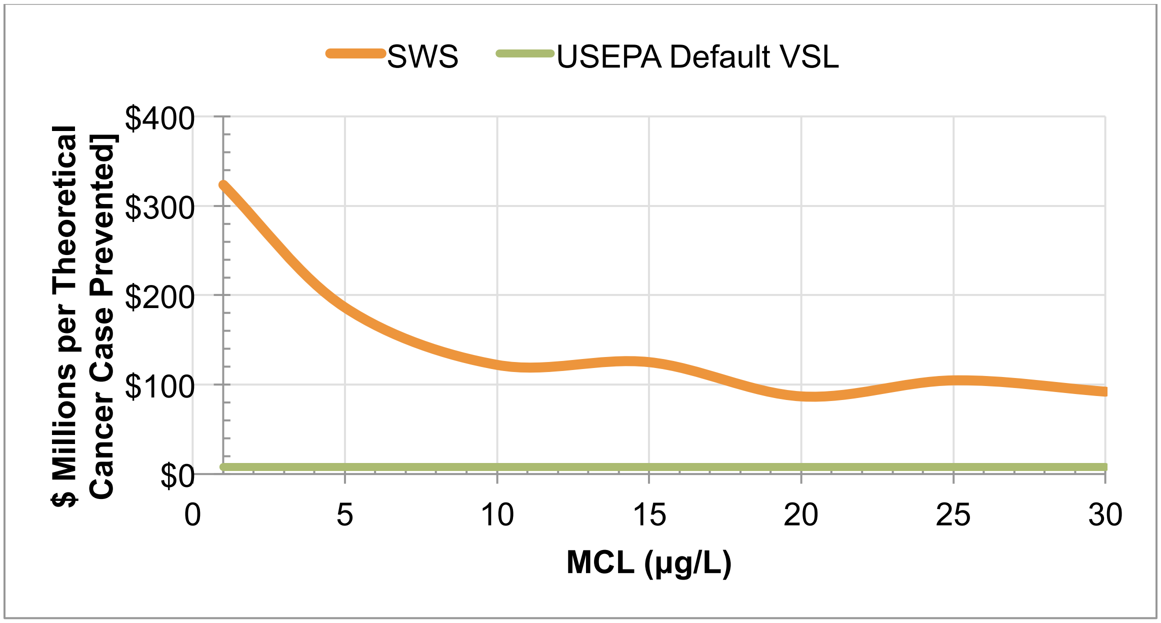
[1] This independent work was sponsored by the American Chemistry Council. The analyses presented belong to the author alone.

[2] Richard B Belzer is an independent consultant. For more information, visit rbbelzer.com.

This is illustrated in Highlights Figure i, summarized in Section I.B beginning on page 8, and explained in detail in Section II.E beginning on page 15.

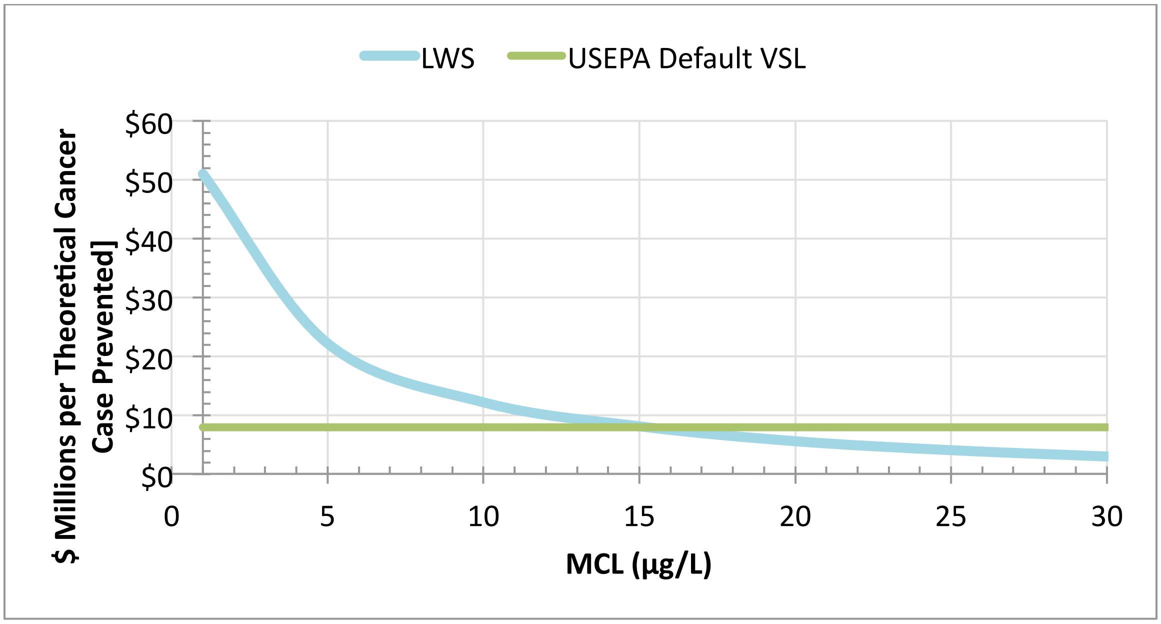
Highlights Figure i: Cost-Effectiveness Ratios for Small Water Systems Across Alternative MCLs



3. **<u>Taken at face value, the Department's cost-benefit analysis shows that drinking water standards 15 µg/L and below are not economically feasible for communities served by large water systems</u>.** The CBA shows that, for these MCLs, aggregate costs are unambiguously greater than aggregate benefits.

This is illustrated in Highlights Figure ii, summarized in Section I.C beginning on page 8, and explained in detail in Section II.E beginning on page 15.

Highlights Figure ii: Cost-Effectiveness Ratios for Large Water Systems Across Alternative MCLs



4. **The cost-benefit analysis contains a number of methodological errors that materially understate costs or overstate benefits, making the reported results highly misleading.** The practice of regulatory cost-benefit analysis requires considerable art as well as science, but theoreticians and practitioners alike have recognized a number of fundamental principles that all such analyses should meet. The CBA violates several of these fundamental principles, and these errors have material effects on the results. They make every alternative MCL appear considerably more attractive than it actually is.

   This is summarized in Section I.D beginning on page 9 and explained in detail in Section III.B report beginning on page 26.

5. **When only the most rudimentary economic error in the cost-benefit analysis are removed, every regulatory standard under consideration is shown to be economically infeasible regardless of system size.** After merely discounting future benefits the way future costs are discounted, engineering cost-effectiveness at 30 μg/L for large systems rises from $3 million to $14 million per theoretical cancer case prevented. Small system engineering cost-effectiveness rises from $92 million to $450 million per theoretical cancer case prevented. These ratios are worse for every other combination of system size and regulatory alternative.

This is shown in the table below, summarized in Section I.E beginning at page 9, and explained in Section III.D beginning on page 34.

Present Value Costs, Theoretical Benefits, and Theoretical Cost-Effectiveness

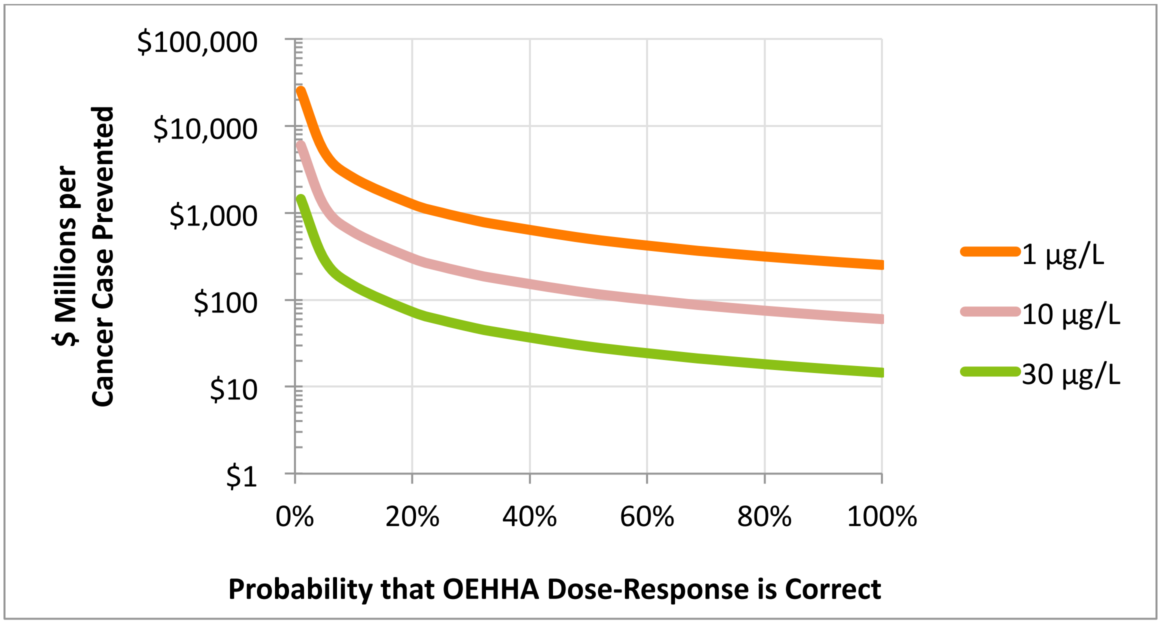
	MCL = 1 µg/L		MCL = 10 µg/L		MCL = 30 µg/L	
	SWS	LWS	SWS	LWS	SWS	LWS
PV Costs ($M)[a]	$1,257	$21,120	$183	$1,888	$5	$155
PV Theoretical Benefits (Cases)[b]	0.787	83.8	0.303	31.5	0.011	10.7
PV C-E Ratio ($M/Case)[c]	$1,600	$250	$600	$60	$450	$14
Multiple of USEPA Default [c, d]	200	32	76	7.6	58	1.8

[a] Millions; [b] 3 significant figures; [c] 2 significant figures; [d] $7.9 million.

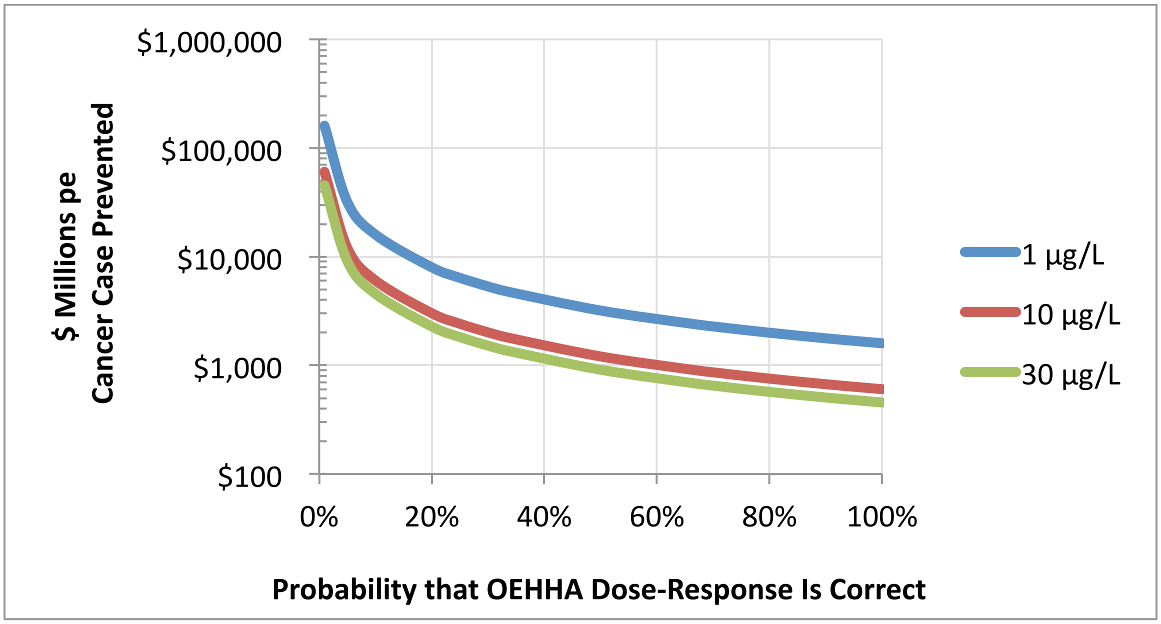
6. **<u>When uncertainty about causation in the OEHHA LNT risk model is accounted for, extremely high cost-effectiveness ratios become extraordinarily high</u>.** All previous corrections have addressed strictly economic phenomena but left intact the assumption that OEHHA's linear no-threshold risk model accurately describes low-dose human cancer risk. If the mode of action assumed by OEHHA is not correct, then this assumption is false and the cost-effectiveness of treatment is even higher.

This is illustrated in Highlights Figure iii (for large systems) and Highlights Figure iv (for small systems), summarized in Section I.E beginning on page 9, and explained in detail in Section III.D.5 beginning on page 42.

Highlights Figure iii: Cost-Effectiveness for Large Water Systems Depends on Probability that OEHHA Dose-Response Relationship is Causal



Highlights Figure iv: Cost-Effectiveness for Small Water Systems Depends on Probability that OEHHA Dose-Response Relationship is Causal



7. **<u>Promulgating any of the regulatory alternatives analyzed would expropriate 6-12% of median California household income for those served by small systems. For many households, income losses would result in a net increase in health risk</u>.** These income losses are *net of the value of cancer risk reduction.* Income is known to be inversely correlated with health risk, so California households would experience a net *increase* in health risk, including the risk of premature mortality from cancer.

   This is summarized in Section I.F beginning on page 10 and explained in detail in Section III.D beginning on page 34.

8. **<u>For "severely disadvantaged" communities served by small systems, any of the regulatory alternatives analyzed would expropriate compliance with the MCLs analyzed would consume 11-20% of household income. For virtually all low-income households, income losses would result in substantial net increases in health risk</u>.** As before, these income losses are *net of the value of cancer risk reduction.* Like California households generally, low-income households would experience net increases in health risk resulting from loss of income, including greater risk of premature mortality from cancer. For low-income households, increases in net health risk would be severe.

   This is summarized in Section I.F beginning on page 10 and explained in detail in Section III.D beginning on page 34.

I. Executive Summary

A. Economic feasibility should be defined according to objective economic criteria universally understood and practiced by the economics profession.

The California Department of Health (CDPH) has prepared a cost-benefit analysis (CBA)[3] to inform regulatory decision-making in the matter of whether to set a drinking water standard for hexavalent Chromium (Cr VI), and if so, what level to set. Cost-benefit analysis is relevant because economic feasibility is a statutory factor the CDPH is required to consider in making this decision.

The term *economic feasibility* lacks a statutory definition, however, so the Department must devise one that has a reasoned basis under the law. The first task is to rule out conceptual definitions that have no foundation in economic theory and practice. For example, it defies economic logic to interpret economic feasibility with regard to costs but not benefits. Indeed, there is no articulable support within economics for such a rule. Regulatory actions that have net social costs are not economically feasible because they make households poorer, and households would never knowingly choose to regulate themselves in ways that make them worse off.

In addition, it is textually illogical to deem a regulatory action economically feasible merely because technology exists that could achieve it. That conflates economic feasibility with technological feasibility, which is itself a statutory factor the CDPH must consider in standard-setting. While the law tries to guide what the CDPH considers to determine economic feasibility, the text is ambiguous because it attempts to define economic feasibility using other terms that are themselves not defined. In any case, the law cannot reasonably be read to imply that the Department should intentionally perform CBA in a manner that is grossly incompatible with established standards in the field.

Fortunately, economic theory and practice provide a simple, objective rule that the Department can use for positively determining what regulatory actions are economically feasible:

> A regulatory action is economically feasible if aggregate social benefits exceed aggregate social costs.

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[3] In the United States, "cost-benefit analysis" is more often called "benefit-cost analysis" by its theoreticians and practitioners. The only U.S. scholarly journal uses the latter formulation; see http://www.degruyter.com/view/j/jbca. It is the publication of the Society for Benefit-Cost Analysis. These terms are generally interchangeable, however, and the term "cost-benefit analysis" is used in this paper because it is the term used by the California Department of Public Health to refer to its own work.

This definition does not presume to dictate what regulatory alternative the CDPH should choose if more than one economically feasible regulatory alternative exists. Rather, the definition simply provides a reasoned basis that officials can use to narrow their choices to include only regulatory alternatives that are economically feasible.

B. Taken at face value, the cost-benefit analysis shows that none of the regulatory alternatives analyzed is economically feasible for households served by small water systems.

The CBA clearly shows that every regulatory alternative analyzed is economically infeasible for households served by small water systems. Engineering cost (a subset of social cost) is at least $87 million per theoretical cancer case prevented.[4] This is about 10 times the default monetary value for preventing a random premature mortality recommended for use by the U.S. Environmental Protection Agency (USEPA). The monetized value of reduced cancer incidence is less than $4 per household per year for every regulatory alternative analyzed. This is less than the price of a book of postage stamps.

Even after the monetary value of cancer reductions is subtracted, the median household served by a small water system faces an income loss that is equivalent to a new income tax ranging from $4,000 to $7,000 per year, depending on the stringency of the standard. Based on the household income figures provided in the CBA, this income loss is approximately 10% of median household income.

These income losses would be especially harmful for disadvantaged California households, and a devastating one for those disadvantaged households served by small water systems. They would expropriate as much as 20% of median household income in communities deemed severely disadvantaged (or in similar households living in more typical communities).

For a family of four living at the poverty level (income: $23,550), this loss would range from 16% of income (at 30 μg/L) to 30% of income (at 1 μg/L).

C. Taken at face value, the cost-benefit analysis shows that drinking water standards 15 μg/L and below are not economically feasible for communities served by large water systems.

The CBA clearly shows that regulatory alternatives 15 μg/L and below are not economically feasible for households served by larger water systems. Engineering cost (a

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[4] The CBA calculates cost-effectiveness by dividing the CPDH's approximation of actual engineering costs by incidence derived from OEHHA's precautionary estimate of theoretical cancer risk. This comparison is invalid in cost-benefit analysis and is therefore highly misleading. See Sections III.B.7 (explaining why) and 0.D (correcting the error).

subset of social cost) is as much as $51 million per theoretical cancer case prevented, about five times the USEPA recommended default monetary value for preventing a random premature mortality. At 15 μg/L, engineering cost-effectiveness is approximately equal to the USEPA default, and it is as low as $3 million per theoretical cancer case prevented at a 30 μg/L standard. For all regulatory alternatives analyzed, the monetized value of cancer risk reduction per household is trivially different from zero. For standards of 20 μg/L and higher, one dollar's worth of cancer reduction benefit would be spread out over 20,000 households. At a standard of 1 μg/L, one dollar's worth of cancer risk reduction benefits would be spread out over 100,000 households.

After the monetary value of cancer reductions is subtracted, median households served by large water systems face income losses as high as $278 per year. For no regulatory alternative analyzed would net benefits be greater than $25 per year for the median household.

D. The cost-benefit analysis contains a number of methodological errors that materially understate costs or overstate benefits.

The practice of regulatory cost-benefit analysis requires considerable art as well as science, but theoreticians and practitioners alike have recognized a number of fundamental principles that all such analyses should meet. These principles include such things as correctly classifying benefits, costs, and transfers; using a correct and identical baseline for counting both benefits and costs; counting costs as opportunity costs rather than expenditures; avoiding double-counting; discounting future benefits and costs; and estimating benefits and costs objectively.

The CBA violates several of these fundamental principles, and these errors have material effects on the results.

E. When obvious material errors in the cost-benefit analysis are removed, every regulatory standard under consideration is shown to be economically infeasible regardless of system size.

The analysis presented here corrects quantitatively for five major errors: (1) using a benefit model inconsistent with the risk model used to derive the public health goal set by the Office of Environmental Health Hazard Assessment (OEHHA), (2) using different time horizons for counting costs and benefits, (3) discounting future costs but not future benefits, (4) failing to account for cessation lags, and (5) failing to account for less-than-certain causation in dose-response. One of these corrections materially increases the cost estimate; the other four materially decrease the benefit estimate.

Even before corrections are made, the monetized value of cancer reductions that the CBA reports are trivially small—between a penny and $4 per household per year. Correcting the benefit assessment model reduces the 100-year sum of cancer cases prevented by a factor of one-third. Discounting reduces them by another factor exceeding 60. That means the discounted number of cancer cases prevented is about 1/200th of those reported in the CBA. This means benefits per household derived from the CBA as $4 per year are actually two cents per household per year.

After replacing the CDPH's invalid steady-state benefit model with one compatible with the OEHHA risk model, discounting benefits as well as costs, and converting benefits and costs into present values, engineering cost-effectiveness for large systems facing the least stringent regulatory alternative analyzed (30 μg/L) rises from the $3 million per theoretical cancer case reported in the CBA to $14 million per theoretical cancer case prevented. Small system cost-effectiveness rises from $92 million to $450 million per theoretical cancer case prevented. The cost-effectiveness ratio is worse for every other combination of system size and regulatory alternative.

In short, after correcting just these obvious errors, none of the regulatory alternatives analyzed is economically feasible for large systems, either. Given only the CBA, a rational decision-maker might be inclined to seek a way to exempt small systems from a uniform statewide standard. Given a corrected CBA, however, a rational decision-maker would likely abandon any such effort.

F. Promulgating any of the regulatory alternatives analyzed would impose a devastiatingly large reduction in income on households served by small systems.

The CBA implicitly acknowledges that a state drinking water standard in the range of values analyzed represents a small income loss for households served by large water systems, but a devastatingly large income loss for households served by small systems. For these households, the income loss would be 6-12% of median statewide household income. It would be 11-20% of median household income in communities designated as "severely disadvantaged."

There are errors in the CBA that, if corrected, would raise these costs even higher. For example, the CBA does not estimate the opportunity cost of lost income. For example, one of the likely opportunity costs of having to give up 10-20% of household income is a diminution of health status. Households would have to substantially cut back on their expenditures for such things as health insurance, medical care, pharmaceuticals, healthy foods, heat and air conditioning. It is easy to envision households experiencing a net increase in health risk—including cancer risk— as a consequence of this income loss.

II. Taken at Face Value, the CBA Shows that a Drinking Water Standard for Hexavalent Chromium Lacks Economic Feasibility Except for MCLs of 15 μg/L and Higher for Large Water Systems Only

The CDPH is required by law to set drinking water standards based on the Public Health Goal (PHG) set by the Office of Environmental Health Hazard Assessment (OEHHA), any national standard set by USEPA, and assessments of technological and economic feasibility.[5] USEPA has not set a national standard, so only the PHG and CDPH's assessments of technological and economic feasibility are relevant.

The CBA resolves affirmatively the question that treatment to remove Cr VI is technologically feasible at the alternative Maximum Contaminant Levels (MCLs) analyzed. However, the CBA does not address whether any member of a range of possible MCLs is economically feasible.[6]

In this section, a credible and objective definition of economic feasibility is set forth. The CBA is then examined at face value to discern whether any of the alternatives considered satisfies that definition. Based on the assumption that the CBA contains no material errors, it can be inferred that none of the alternative MCLs analyzed is economically feasible for California residents served by small public water systems (i.e., those with 200 or fewer service connections). As long as the CBA is without material error, MCLs of 15 μg/L and higher are economically feasible but only for residents served by large public water systems.

A. The CDPH cost-benefit analysis deserves credit for transparency.

The authors of the CBA have done an excellent job documenting how costs and benefits were calculated.[7] This richness is particularly on display in the calculation of engineering costs.[8]

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[5] California Health and Safety Code § 116365(b).

[6] California Department of Public Health (2013). The term "economic feasibility" does not appear anywhere in the cost-benefit analysis.

[7] Costs and benefits were not *estimated*, where that term means derived from statistical procedures whose objective it to derive the best approximation of the true but unknown relevant moment of a distribution (e.g., the arithmetic or geometric mean). Costs and benefits were *calculated* using a combination of assumptions, empirical data, and engineering equations.

[8] Costs are described in the CBA as *compliance* costs, but that description is not correct. More accurately, the methods produce values for *engineering* costs. These values are informed by knowledge of water engineering but not economic theory or economic methods, a prerequisite for cost-benefit analysis.

In Section III, material economic deficiencies in the CBA are identified and, where possible, their effects are quantified. These deficiencies are mostly in the calculation of benefits.

B. Economic feasibility determinations should account for source water and system size.

The CBA disaggregates engineering cost by source and system size. This is essential because drinking water treatment has high fixed costs and low variable costs, and each public water system must fund treatment by assessing its own customers. For large systems, the CBA calculates annualized aggregate engineering costs ranging from $11 million (30 μg/L) to $1.6 billion (1 μg/L).[9] For small systems, the CBA calculates annualized aggregate engineering costs ranging from $0.4 million (30 μg/L) to $95 million (1 μg/L).[10]

Annualized engineering costs per system vary substantially by system size, but more importantly, engineering costs per service connection vary even more by system size—and in the opposite direction. Annualized engineering costs per system range from six times greater for large systems (30 μg/L) to 16 times greater (1 μg/L).[11] However, engineering costs per service connection range from 22 times greater for small systems (1 μg/L) to 296 times greater (30 μg/L).[12] Thus, the relative burden by system size across alternative MCLs is roughly proportional when calculated in costs per system but inversely proportional when calculated in costs per service connection. That means economic feasibility cannot be determined based on cost or cost per household alone. To avoid such arbitrariness, a more sophisticated understanding of economic feasibility is required.

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[9] California Department of Public Health (2013), PDF p. 78: For 30 μg/L, 10 systems × $1,169,652 per system ≈ $11 million. For 1 μg/L, 354 systems × $4,493,794 per system ≈ $1.6 billion.

[10] California Department of Public Health (2013), PDF p. 78: For 30 μg/L, 2 systems × $196,807 per system ≈ $0.4 million; For 1 μg/L, 340 systems × $278,481$95 million.

[11] California Department of Public Health (2013), PDF p. 78.

[12] California Department of Public Health (2013), PDF p. 80.

C. Economic feasibility determinations should account for benefits as well as costs.

Considering cost alone to determine economic feasibility has another serious limitation in addition to yielding arbitrary results. In particular, a cost-only approach assumes that the health benefits of drinking water treatment are irrelevant. This approach is notably at odds with the way financial decision-makers understand feasibility and actually make decisions. When families think of what options are economically feasible, they try to consider all of the costs they expect to bear—not just a subset of costs—but they also consider the benefits they except to gain. Few, if any, household financial decisions are ever made by considering only a subset of costs and giving no attention at all to benefits. If CDPH wants its Cr VI decision to resonate with the public, it should choose a decision rule that closely approximates what families would do if the decision were theirs to make.

With this in mind, a family expenditure is understood to be economically feasible if and only if it produces total benefits greater than total costs. No family would knowingly spend $1,000 for a household appliance that was known to be worth $100. Similarly, few, if any, households would consider only the acquisition price but ignore known indirect costs. No household would simply ignore the benefits it expects to gain from having and using the appliance. A household decides that the appliance purchase is economically feasible if its total cost is less than the value the household places on the services the appliance is expected to deliver.

For the same reason, a drinking water treatment system should be considered economically feasible if its total cost of acquisition, delivery, installation, and operation is less than the monetized value of reduced health effects expected to be delivered. If total costs are greater than the expected health benefits, the system should be deemed economically infeasible because it makes the water system's customers worse off than they would be if they took no action. There is a better use of scarce resources than to spend them in ways that make households poorer.

Many financial choices are economically feasible for a family, but only one represents the best choice that a family can make. That choice is the one that produces the greatest net benefit—the highest surplus of benefits over costs. Households may not succeed in every attempt to make the best choice, especially if they lack important information concerning true costs and benefits. Still, there should be no doubt that this is what most families try to do. The decision problem for CDPH should be similar. Having first excluded regulatory standards that are economically *infeasible* because they cost more than they produce in health benefits, the Department's next task is to determine which standard is economically *optimal*—that is, which standard produces the greatest surplus of risk reduction benefits over costs.

To determine economic feasibility, decision-makes need to monetize the value of preventing a random cancer case. For all of its environmental and public health protection regulations, including primary drinking water standards set under the Safe Drinking Water Act, the USEPA recommends that $7.9 million be used as the default estimate for the value of preventing a random premature death.[13] This exceeds the value for preventing a random cancer case because not all cancers are fatal. Nonetheless, it provides a useful upper-bound benchmark for the benefits of alternative MCLs.

D. *Economic feasibility determinations should take account of disproportional impacts on the poor.*

A cost-only approach to defining economic feasibility also does not account for distributional impacts, and on this margin the CBA is seriously lacking. The CBA includes some information on median household income, both for California as a whole and for two arbitrarily defined types of "disadvantaged" communities.[14] However, having reported this information the CBA includes no analysis of whether any of the alternative MCLs would be expected to disproportionately affect these communities.

California recently established a scheme for ranking environmental justice impacts and taking them into account in public decision-making. This scheme, called the California Communities Environmental Health Screening Tool ("*CalEnviroScreen*") computes an index consisting of over dozen variables including environmental and socioeconomic factors. While the ranking system has important limitations,[15] it nevertheless represents a much more sophisticated portrayal of "disadvantage" than median household income alone. Because it is GIS-based, it is well within the capacity of CDPH to ascertain the extent to which communities that have high *CalEnviroScree*n ranks are expected to be disproportionately affected.[16] Where communities that have high *CalEnviroScreen* ranks can be expected to bear costs that exceed the monetized value of health benefits, a determination of economic infeasibility should be utterly noncontroversial.

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[13] U.S. Environmental Protection Agency (2010), p. 7-8., usually described in shorthand as the "value of a statistical life," or "VSL " USEPA's figure is in 2008 dollars. Adjusting for inflation since 2008 would increase it 8.6%, because inflation has been positive, but adjusting for income would decrease it because median income has fallen 7.2%. See Advisor Perspectives (2013). USEPA's recommended value is a function of both factors.

[14] California Department of Public Health (2013), PDF p. 82; statewide average household income: $58.553; "disadvantaged" community (defined as < 80% of the median): <$46,842, "severely" disadvantaged community (defined as < 60% of the median): < $35,132.

[15] Belzer (2012), Belzer (2013).

[16] Even constant economic effects across all California communities would have disproportionate effects on economically disadvantaged communities.

E. Taken at face value, the cost-benefit analysis shows that primary drinking water treatment to remove Cr VI is economically infeasible for all households served by small water systems, and economically feasible for households served by large water systems only if the MCL is set at 15 μg/L or higher.

The CBA reports the calculated number of theoretical cancer cases prevented and cost-effectiveness results from its analysis as shown in Table 2.

1. According to the CBA, cost-effectiveness ratios for Cr VI treatment are exceedingly high for stringent MCLs and for any MCL applied to sall systems.

Unsurprisingly, the more stringent the MCL, the greater is the number of theoretical cancer cases prevented. Nevertheless, as the MCL is made more stringent, more public water systems are expected to be required to install treatment, and aggregate engineering costs rise disproportionately faster. The result is extremely high cost-effectiveness ratios for small water systems generally and for large water systems at MCLs below 15 μg/L. Cost-effectiveness ratios less than the USEPA recommended default are in green; ratios greater than the USEPA default are in red.

Table 1: Estimated Reductions in Theoretical Cancer Cases and Estimated Cost-Effectiveness of Seven Alternative MCLs for Cr VI, as Reported by CDPH

	Estimated Reduction in Theoretical Excess Cancer Cases per Year		Estimated Cost per Theoretical Excess Cancer Case Prevented	
MCL (μg/L)	Small Water Systems	Large Water Systems	Small Water Systems	Large Water Systems
1	0.293	31.229	$323,151,751	$ 50,939,936
5	0.180	19.109	$186,383,419	$ 22,147,214
10	0.113	11.719	$122,140,461	$ 12,135,662
15	0.0575	7.938	$125,177,695	$ 8,066,685
20	0.0337	5.712	$ 86,869,733	$ 5,534,422
25	0.00566	4.669	$104,799,274	$ 4,011,427
30	0.00427	3.978	$ 92,181,248	$ 2,940,302
Source: California Department of Public Health (2013, PDF p. 83).				

Figure A and Figure B display this information graphically, separately for large and small water systems because of the wide differences in scale. For small water systems (SWS, shown in red), CDPH estimates that engineering cost generally exceeds $100 million per theoretical cancer case prevented, and falls below $90 million only for the 20 μg/L MCL alternative. For large water systems (LWS, shown in blue), CDPH estimates that engineering cost per theoretical cancer case prevented exceeds the USEPA recommended estimate of the value of preventing a random premature fatality ($7.9 million, shown in green) for the 1, 5, 10, and 15 μg/L MCLs. Only MCLs 20 μg/L and higher have engineering cost-effectiveness ratios less than the USEPA default, and only then for large water systems.[17]

2. Based on the CBA, the net benefit per household of Cr VI treatment is small for large systems at high MCLs and severely negative for small systems at all MCLs.

Another way to look at engineering costs and benefits—one that the public likely would find more intuitively appealing—is to calculate them at the household level. Although the CBA does not include annual household-level cost and benefit estimates, it does provide the necessary information to derive them.

Annual household-level benefits and net benefits derived from the CBA are summarized in Table 2, separately by system size, and presented graphically in Figure C for small systems only. Cancer risk reductions are assumed to be real, not merely theoretical, as they are correctly described by CDPH. Each cancer case is assumed to be immediately fatal; no case is ever cured. Household-level *benefits* are obtained by dividing CDPH's calculated number of theoretical cancer cases prevented by the number of service connections covered, then multiplying by the USEPA recommended estimate of the value of preventing a random premature mortality. Household-level *net benefits* are obtained by subtracting this monetized value from CDPH's estimate of engineering costs per service connection for each system size category and each alternative MCL.

For households served by large water systems, cancer risk reductions are so small that they are valued (by households themselves) at infinitesimal fractions of a penny per year, regardless of the stringency of the MCL. Households served by small water systems are calculated to benefit on average between $0.01 and $3.74 per year, depending on the

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[17] When adjustments are made to improve the accuracy of these ratios, as is performed in Section 0 beginning on page 45, MCLs 20 μg/L and above turn out not be cost-effective even for large water systems.

Figure A: Calculated Cost-Effectiveness of Alternative Cr VI MCLs, as Reported in CBA (LWS)

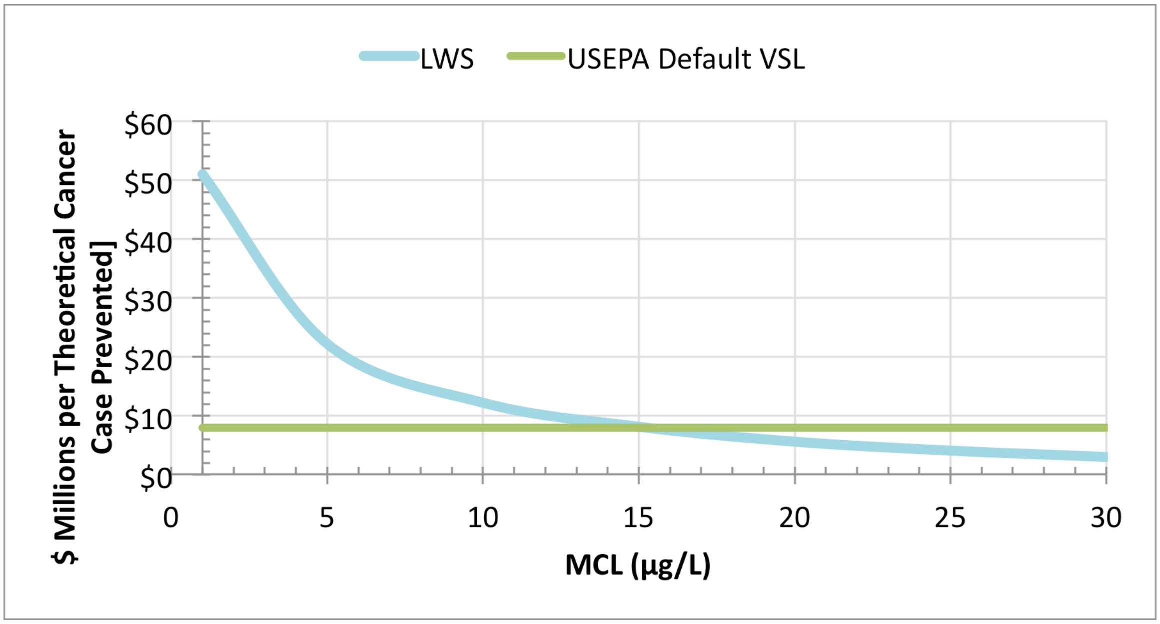

Figure B: Calculated Cost-Effectiveness of Alternative Cr VI MCLs, as Reported in CBA (SWS)

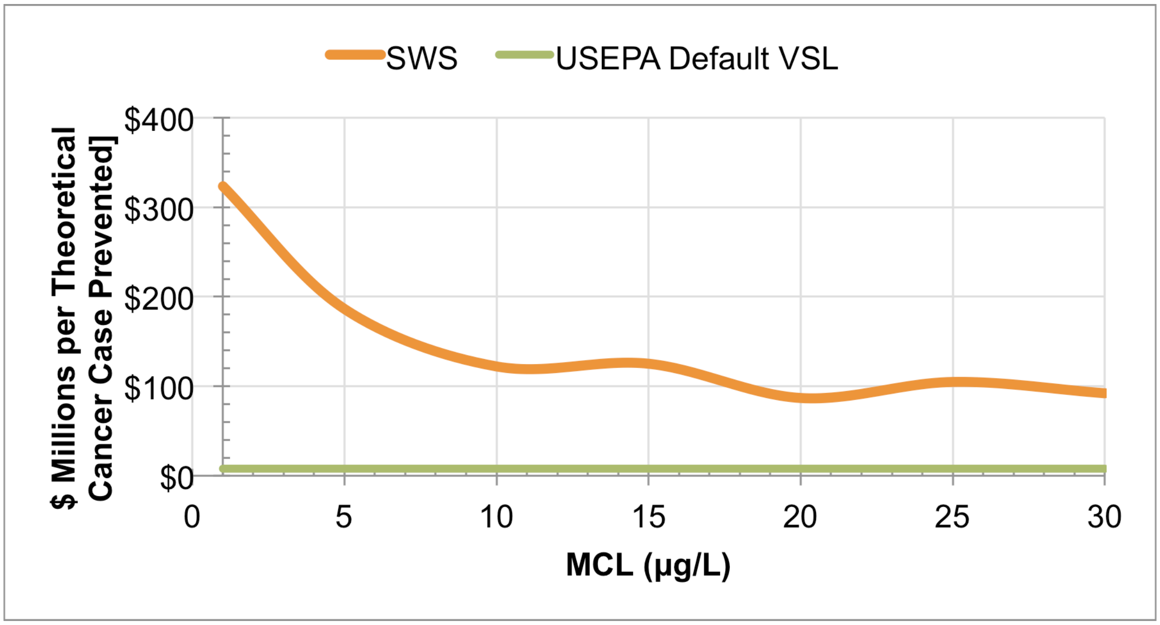

Figure C: Calculated Change in Annual Household Income for Median, Disadvantaged, and Severely Disadvantaged Communities by MCL after 70 Years of Compliance, as Derived from the CBA (SWS Only)

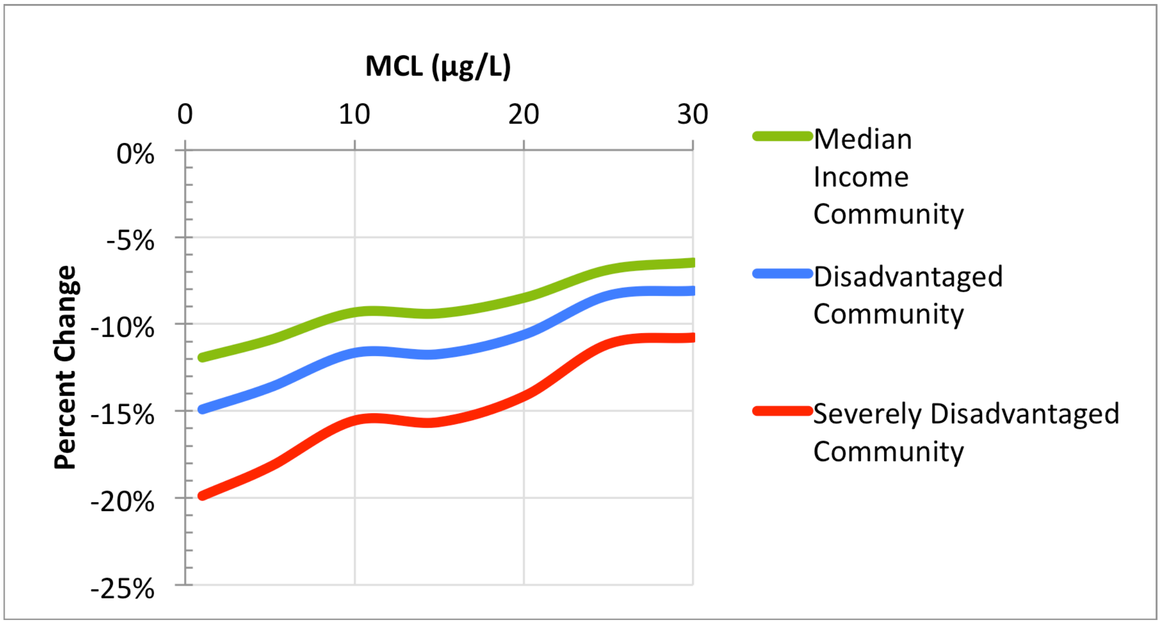

Table 2: Annual Household-Level Benefits and Net Benefits, by Water System Size

MCL (μg/L)	Small Water Systems			Large Water Systems		
	Benefits	*Net Benefits (NB)*	*NB as % of Median Income*	*Benefits*	*Net Benefits (NB)*	*NB as % of Median Income*
1	$0.01	-$6.984	-11.93%	$0.00001	-$ 270	-0.47%
5	$0.06	-$6,393	-10.92%	$0.00002	-$ 93	-0.18%
10	$0.15	-$5,463	-9.33%	$0.00003	-$ 29	-0.05%
15	$0.30	-$4,974	-9.39%	$0.00004	-$ 1	-0.00%
20	$0.93	$3,918	-8.50%	$0.00005	$ 14	0.02%
25	$2.28	-$3,818	-6.88%	$0.00005	$ 21	0.04%
30	$3.74	-$3,788	-6.47%	$0.00005	$ 25	0.04%

Sources: Derived from CBA pp. 80-83 and USEPA (2010). Median income are as reported in CBA ($58,553). All risks are assumed to be actual, not theoretical. Benefits and net benefits are realized in year 70 after the MCL is attained. See Section I.D.1.

stringency of the MCL.[18] Perhaps counterintuitively, household-level benefits decline as the MCL is made more stringent. The reason is that, as the MCL approaches zero, the number of public water systems that must install treatment rises exponentially, and so does aggregate engineering cost. However, as the MCL approaches zero the amount of cancer cases prevented rises only linearly.

The second column in each set in Table 2 provides a calculation of *net benefit*. This is the difference between household-level (engineering) costs and the monetized value of household-level (theoretical) cancer reduction benefits.[19] For large systems, net benefits are positive only for MCLs equal to or greater than 20 μg/L. For households served by small water systems, net benefits are negative at every MCL under consideration.

These negative net benefits are substantial, which can be seen by computing the percent of median household income they represent. For large water systems, net benefits represent a small fraction of median household income. Only for the 1 μg/L MCL does this fraction approach 0.5% of median income. Thus, for households served by large water systems, relatively high MCLs may meet conventional "affordability" criteria.[20] At the same time, it should be kept in mind that where *net benefits* are negative, the cost of meeting the MCL acts as a new tax on gross household income that produces nothing of value for the household. Choosing any of these MCLs would make the household poorer, and thus worse off.

For small water systems, net benefits are always negative and take away an astoundingly large fraction of household income. Treatment to remove Cr VI from drinking water is equivalent to a new tax on gross household income ranging from 6% to 12% depending on the stringency of the MCL selected. As before, this differs from a conventional income tax because yields nothing of value to the household. Recognizing that half of all households have income below the median, these percentages understate the magnitude of their income losses.

Household-level net benefits are shown graphically in Figure D (for both small and large systems) and Figure E (for large system only, to make them easier to visualize). For convenience, in Figure E a green line is drawn where net benefit equals zero—i.e., where

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[18] Actual benefits will be lower to the extent that actual risk is less than theoretical risk.

[19] As noted in Section III.B.7 beginning on page 30, this comparison is invalid and highly misleading. Cost-benefit analysis requires that net benefit be calculated as expected benefits minus expected costs. The *theoretical* number of cancer cases substantially exceeds the *expected value* number of cancer cases prevented.

[20] U.S. Environmental Protection Agency (2012). Unlike economic feasibility, "affordability" is a strictly subjective construct. There is no science underlying USEPA's decision to deem 2.5% of income as an "affordable" burden.

Figure D: Annual Net Benefit in Dollars per Household After 70 Years of MCL Engineering for Seven Alternative Cr VI MCLs

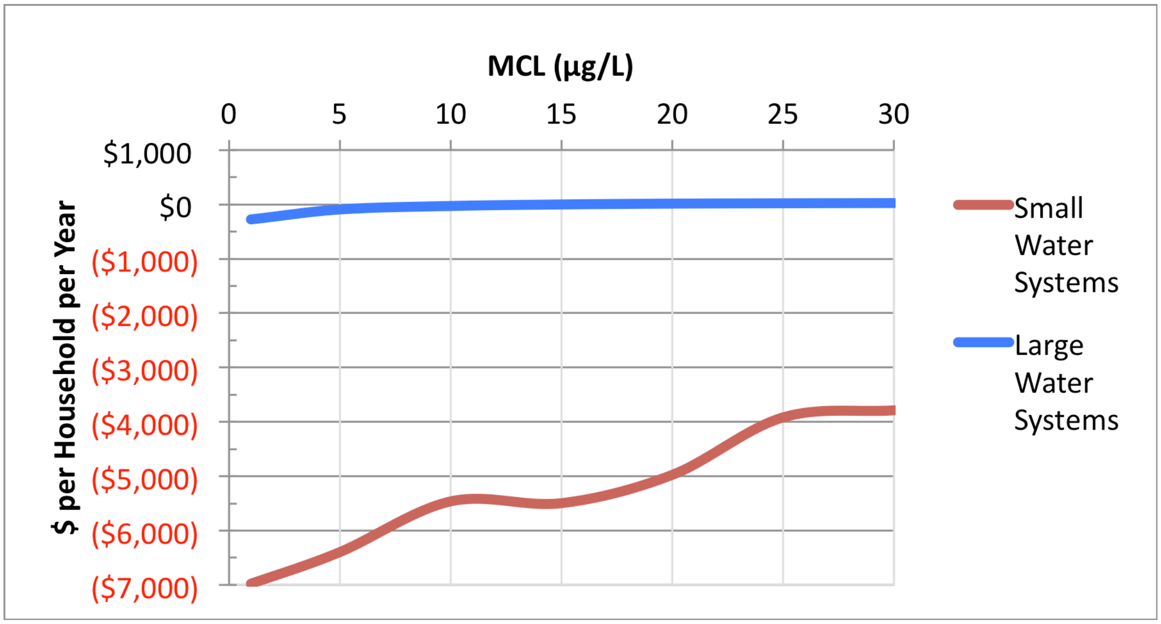
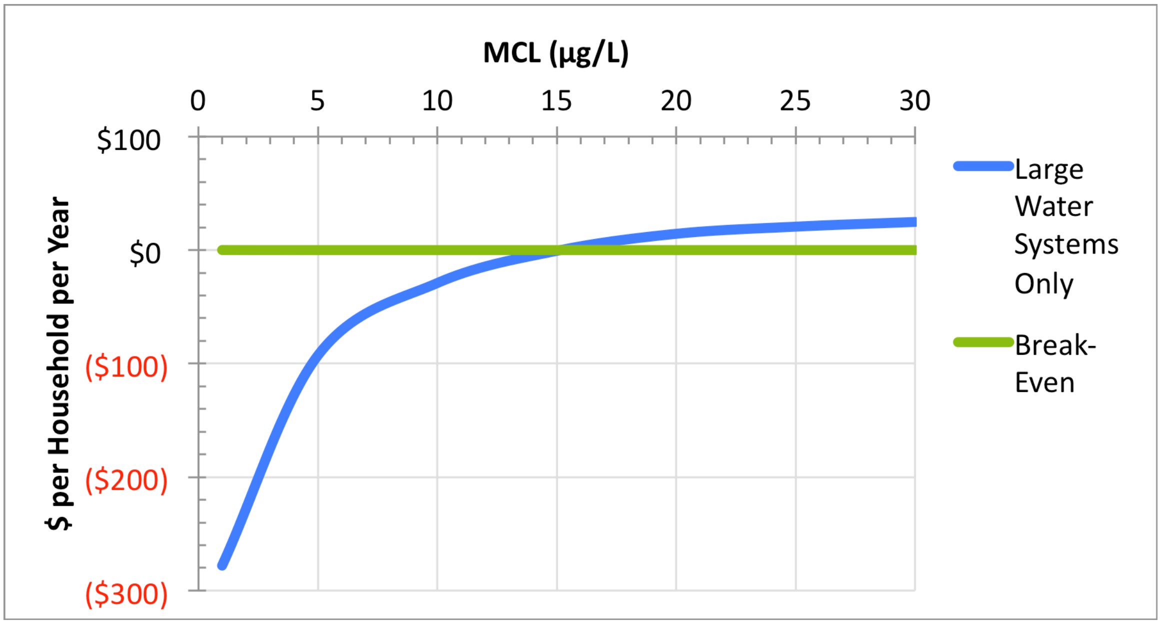


Figure E: Annual Net Benefit in Dollars per Household After 70 Years of MCL Engineering for Seven Alternative Cr VI MCLs (Large Water Systems [LWS] Only)



the household served by a large water system "breaks even." Where the curve lies above the green line, the household is made better off by purchasing treatment to remove Cr VI from drinking water. Where the curve is below the green line, however, the household is made worse off. For households served by large systems, they gain as much as $25 per year (30 μg/L) and lose as much as $278 per year (1 μg/L). For households served by small systems, however, the net benefit curve is always below the break-even line. They lose between $3,788 per year (30 μg/L) and $6,984 per year (1 μg/L).

A. Economic feasibility taking variability into account

The tables and figures shown above represent calculated midpoints across tens, hundreds, or even thousands of water systems, depending on the MCL. Unfortunately, the CBA reports only these calculated midpoints and reveals nothing about variation across systems within each system category or across MCLs. Actual engineering cost-effectiveness will differ across systems because of variability in system-level costs. Moreover, for each combination of system size and MCL, about half of all systems are expected to have engineering costs higher than the values provided in the CBA. Calculated engineering costs may significantly understate actual costs for some water systems, such as those consisting of a large number of small systems under common ownership or control, by effectively misclassifying multiple small systems as a single large system.

The absence of information and analysis about variability is a significant deficiency in the CBA. Midpoint calculations are useful first-order indicators, but they are incomplete without being accompanied by information showing how much variation lies hidden. This is especially so if the distribution from which a midpoint is drawn is skewed, for in that case the average may be a misleading indicator. For example, if costs are distributed logarithmically, the correct average is not the arithmetic mean but the geometric mean.

B. Economic feasibility taking distributional effects into account

In the preceding subsections, care was taken to distinguish between small and large water systems. The reason for doing this is obvious: drinking water treatment has high fixed costs, and small water systems have fewer customers to share them.

There is another reason to be concerned about negative net benefits, especially for small water systems. Not all communities in California have median household incomes equal (or close to) the Statewide median. The CBA reports two categories of communities of special concern: those with median household income less than 80% of the Statewide median ($46,842), termed "disadvantaged"; and those with median household income less than 60% of the Statewide median ($35,132), termed "severely disadvantaged."[21]

For households in "disadvantaged" or "severely disadvantaged" communities served by large water systems, negative net benefits consume a proportionately larger fraction of median household income. Still, for no MCL under consideration does this percentage exceed 0.8% of median household income. These households would have no basis for objecting based on conventional "affordability" criteria (less than or equal to 2.5% of income). Nonetheless, because these are *negative* net benefits—the monetized value of cancer risk reduction has already been subtracted—it remains true that these households gain nothing from enduring the an additional income tax even if it is less than 1% of gross income.

Households in "disadvantaged" or "severely disadvantaged" communities served by small water systems would endure truly punitive new financial burdens, and do so for no gain. Among "disadvantaged" communities, the implicit new income tax would claim as much as 15% of gross household income. For households in "severely disadvantaged" communities, the new income tax would claim as much as 20% of gross household income.

CDPH could easily supplement the CBA with a richer analysis of adverse distributional impacts by utilizing *CalEnviroScreen*, OEHHA's recently updated environmental justice ranking tool.[22] This tool relies on data from 11 "pollution burden" indicators and six "population characteristics" indicators to produce a weighted, relative rank ranging for approximately 1,800 Zip Codes in the State.

The choice of cut-point for deeming a community worthy of environmental justice concern is inherently arbitrary. To make the implications of alternative cut-points easier to understand, OEHHA publishes maps that divide ranks into deciles. In addition, for a previous version of *CalEnviroScreen,* OEHHA made available a spreadsheet containing each indicator's value and the composite ranking for each Zip Code.[23] Thus, a simple remedy for the dearth of distributional analysis in the CBA would be to compare each public water system in the CDPH database to the Zip Code(s) it serves, then determine whether the costs

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[21] California Department of Public Health (2013), PDF p. 82. These categories are reported in the CBA, but there is no analysis presented explaining how these communities would be affected by alternative drinking water standards.

[22] California Environmental Protection Agency Office of Environmental Health Hazard Assessment (2013a). Although OEHHA calls *CalEnviroScreen* a "scoring" tool, because it is based on relative "scores," it is actually a ranking tool. If every community's score were to be cut in half—an unambiguous improvement in environmental quality—every community's rank would remain unchanged.

[23] California Environmental Protection Agency Office of Environmental Health Hazard Assessment (2013b).

of alternative Cr VI MCLs are disproportionately greater for communities with high *CalEnviroScreen* rankings.

III. Methods Used in the Cost-Benefit Analysis Depart from Generally Accepted Economic Principles

Cost-benefit analysis sometimes requires as much art as science, but there is universal agreement among practitioners that certain principles must always be followed.[24] Unfortunately, the CBA violates several of these principles in ways that significantly undermine the ability of public officials to rely upon it for regulatory decision-making, and for the public to have confidence that the agency's decisions reflect its best interests.

Several of these universal principles are listed and described briefly in Subsection A. Examples showing how the CBA violates some of them are provided in Subsection B. How these errors are likely to mislead decision-makers is explained in Subsection C. Finally, rough corrections needed are described in Subsection D.

A. Seven generally accepted principles in cost-benefit analysis

Every cost-benefit analysis should adhere to certain generally accepted principles. CBAs that do not adhere to these principles are inherently suspect and their results are presumptively invalid and unreliable. It is the duty of the cost-benefit analyst to prove that a departure from an generally accepted principle, which may be convenient due to a variety of factors, does not materially affect the results in a particular application. Usually, making that case requires a showing by sensitivity analysis that correcting the error would have no material effect on the identity of the preferred alternative under any plausible decision rule.

1. Benefits, costs, and transfers must be correctly classified.

No CBA will produce valid and reliable results if benefits, costs, and transfers are misclassified. While it is rare for benefits to be classified as costs, and vice versa, it is not unusual for transfers to be misclassified as one or the other. Transfers do not affect net benefits because costs borne by some are benefits to others.

This does not mean transfers are unimportant. For example, to the extent that a regulatory action relies on exogenous transfers to compensate or subsidize, they must be counted as costs. In addition, to the extent that regulations require transfers and these transfers entail their own social costs, their social costs must be counted.

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[24] See, e.g., Boardman, Greenberg, Vining, et al. (2010).

2. The same regulatory baseline must be used for estimating benefits, costs, and transfers.

The choice of regulatory baseline is often a subject of controversy, and in some cases, analytical mischief. The proper baseline is the future state of the world in the absence of a regulation. To the extent that the state of the world is reasonably expected to change over time due to exogenous phenomena, those changes must be accounted for in the regulatory baseline.

In addition, it is inappropriate to divide a regulatory action into multiple parts for the purpose or effect of hiding social costs. Thus, a fair case can be made that choosing a public health goal and setting a drinking water standard are two components of the same regulatory action for analytic purposes. Treating these components separately imposes all the burden of accounting for cost on drinking water standard-setting and allows choosing the public health goal to escape the same scrutiny.

3. Costs must be estimated not as engineering costs but as opportunity cost, which is the value of benefits foregone due to bearing engineering and other compliance costs.

The proper way to estimate cost is opportunity cost, not expenditure or its cousin, engineering cost. At best, using expenditure or engineering cost captures every dollar that leaves the regulated entity's "pocket" in order to achieve compliance. Dollars that are not directly "spent" tend to be ignored, however, even if there is a substantial imputed value involved. For that reason alone, the use of expenditure or engineering cost is always and everywhere an inferior method of cost assessment.

More importantly, both expenditure and engineering cost are not appropriate ways to measure cost because they are grounded on an incorrect understanding of what "cost" means. The correct way to understand cost is *opportunity cost*, the value of benefits that must be foregone when resources are committed to comply with a regulation. Opportunity cost is always greater than expenditure or engineering cost.

When consumers decide whether to spend $1,000 on a household appliance, they compare the benefits they expect to gain from the appliance with the value they place on the next best way they could spend $1,000. If they value the appliance at, say, $1,500 but value the next best alternative expenditure at $2,000, they will choose the next best alternative in lieu of the appliance because it provides $500 more in benefits. They will spend the $1.000 on the appliance only if the value they place on the next best alternative use of that money is less than $1.500. Thus, while $1,000 may be the *price* of the appliance, its *opportunity cost* is $1,500.

Just as the opportunity cost of the $1,000 appliance must exceed its $1,000 price—otherwise, why purchase it ever?—the opportunity cost of drinking water treatment must exceed the engineering and other compliance costs of treatment.

Otherwise, the new treatment train makes households served by the water system worse off because it forces them to purchase something that reduces their welfare.

4. Future benefits, costs, and transfers must all be discounted at an appropriate interest rate.

It is generally understood that all future effects have less value than effects realized today. If costs must be borne, it is always preferred that they be borne in the future. Similarly, benefits that accrue today are more valuable than benefits whose realization is delayed. Economists may differ concerning the correct interest rate to use for accounting for future effects, but there is universal agreement that all future effects must be discounted.

5. Benefits, costs, and transfers must not be double-counted.

Not all benefits, costs, and transfers can be quantified, and not all quantifiable effects can be monetized. Nonetheless, it is essential that all reasonable effort be devoted to identifying all effects, and quantifying and monetizing them to the maximum extent practicable. In the process of attempting to be comprehensive, it is essential not to double-count any effect. Every effect should be counted, but no effect must ever be counted more than once.

6. Benefits, costs, and transfers must be estimated objectively—that is, without known biases, including those reflecting the policy preferences of either the analyst or the decision-maker.

CBA has value for helping decision-makers sort through an array of alternatives to choose the one that best suits the decision-maker's purposes. When decision-makers are public officials, it is expected that they will align their purposes with the public that they serve.

CBA is a tool for aiding decision-making, but it is not itself a decision rule. Decision-makers may choose to use well-known normative rules, such as the Kaldor-Hicks criterion, as the basis for selecting an alternatives. But that is a normative choice owned by the decision-maker or his principal (in this case, the residents of California), separate and distinct from the objective analytic procedures that animate CBA. Indeed, CBA enables public decision-makers and their constituents to understand the opportunity cost (i.e., social benefits foregone) of departing from the Kaldor-Hicks criterion.[25]

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[25] The Kaldor-Hicks criterion states that one outcome is superior to another if those who are made better off could compensate those who are made worse off and still have a surplus let over. Kaldor-Hicks dates from 1939 and is the dominant decision rule in welfare economics; cost-benefit analysis is its practical means of implementation.

For CBA to be used this way, it is imperative that all benefits, costs, and transfers be estimated as objectively as possible. Where significant uncertainties exist, they should be modeled using an appropriate tool (e.g., sensitivity analysis or Monte Carlo simulation) to determine how they affect the results. But it is improper to embed so-called "conservative" assumptions or precautionary policy preferences into CBA, irrespective of whether those preferences belong to the analyst or the decision-maker. When the analyst's policy preferences are embedded, the CBA expropriates from the decision-maker the ability to exercise statutorily delegated authority. When a decision-maker's policy preferences are embedded, the CBA becomes a façade for opaquely justifying decisions made on other grounds.

B. The CDPH cost-benefit analysis departs from several generally accepted principles.

Methods used in the CBA violate several of these foundational principles. This section focuses on violations that are obviously so large that the CBA materially misleads decision-makers and the public concerning the likely consequences of alternative drinking water standards.

1. *Cost* is estimated as engineering cost, not opportunity costs.

The CBA only considers costs from an engineering perspective. It ignores the opportunity cost to each public water system of installing and operating treatment to remove Cr VI, which consists of the value of the next best use of these resources. A public water system that is required to treat drinking water to remove Cr VI cannot use these resources for any other purpose, even if another purpose has greater value. The correct way to measure the cost of Cr VI treatment is to ascertain the next best use of these resources, estimate the social benefits that would accrue of it were chosen instead, and treat these social benefits as the opportunity cost of Cr VI treatment.

Typically, public water systems that are required to install and operate treatment to remove Cr VI will not actually bear any of these costs. Rather, costs will be shifted to some combination of stockholders, customers, suppliers, employees, and taxpayers. These shifts can be understood as indirect compliance costs because they would not exist but for the regulatory requirement to install and operate Cr VI treatment.

A reasonable first-order approximation is that engineering costs for Cr VI treatment will be shifted to water system customers. Of all the parties to whom costs could be shifted, customers are the least mobile and thus are least able to avoid them. Stockholders can avoid costs by relinquishing ownership interests. Suppliers have national markets and can choose to sell to others. Employees can change jobs. For water system customers to escape bearing the cost of treatment, they must either substitute other sources of water or physically move to a location served by a water system not subject to the regulatory standard.

Because the CBA uses engineering costs instead of opportunity costs, we can be sure that it understates the true cost of attaining any of the alternative MCLs analyzed. Without a more rigorous cost assessment, however, we cannot estimate the magnitude of this error. Nonetheless, we can know for certain that it is substantial for small water systems and their customers. It is especially substantial for low-income customers of small water systems, for whom the passed-through cost of Cr VI treatment is projected to be a large fraction of household income.

2. The cost-benefit analysis does not include all water systems that would be required to install treatment.

Each public water system is assumed to have a fixed concentration of Cr VI, and the category to which it is assigned is determined based on the results of preliminary monitoring. Depending on this concentration, water systems must adhere to a specified monitoring schedule.

For reasons that are not explained in the CBA, it is apparently assumed that subsequent monitoring never changes a water system's assignment. That is, if preliminary monitoring indicates that the sources on which a system relies contains, say, 17 μg/L Cr VI, that system is covered by a treatment requirement if the MCL is set at 15 μg/L or less, but never covered if the MCL is set at 20 μg/L or more. That is, monitoring is assumed to never result in any water system that begins exempt from treatment being subsequently required to install and operate a treatment train. In other words, preliminary monitoring is assumed to fully determine regulatory applicability, and subsequent monitoring has no effect.

In fact, subsequent monitoring may result in water systems shifting from an exempt to a nonexempt category. Similarly, subsequent monitoring could show that a water system had been misclassified as nonexempt based on preliminary monitoring results that were outliers.

The proposed rule cannot treat differences in subsequent monitoring results in a manner that is economically symmetrical. If subsequent data show that a system is distributing drinking water above the MCL, it will be required to install treatment. But if subsequent data show that a system's influent water is consistently below the MCL, that system will not be able to recover its prior investments in treatment because these costs are sunk.[26]

This feature of the CBA thus understates the number of water systems that will be required to install treatment, and therefore it understates total engineering costs. The amount of understatement depends on the variability of monitoring results, about which

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[26] The proposed rule also makes it harder for water systems to exit a treatment requirement than to be captured by it.

there appear to be few data. The CBA implicitly assumes that this variability is less than the differences in alternative MCLs. No evidence is supplied to support this assumption, and monitoring data published by CDPH indicate that it is not valid.

3. Engineering costs are estimated over a 20-year horizon, but benefits are estimated with an infinitely long horizon.

The CBA reasonably assumes that expenditure on water treatment is largely borne up front and paid for in 20 annualized payments reflecting either the interest rate on debt financing or consumers' preference for delayed cost-bearing (the CBA does not say which). Implied in this analysis, however, is the idea that capital investments in drinking water treatment have a finite lifetime. However, the CBA does not account for the cost of replacing treatment technology in the 21st year, or at the end of any multiple of 20 years after initial installation. Instead, costs are assumed to vanish after 20 years.

At the 7% discount rate used in the CBA, the value today of a dollar's expenditure in Year 21 is about $0.26. If the capital were to be replaced in years 21, 41, 61, and 81 as estimated for Year 1, present value engineering costs would be about 35% higher than presented in the CBA. Still, there is nothing intrinsically incorrect about terminating the engineering cost analysis at 20 years, provided that benefits also are terminated after 20 years.

In the CBA, however, benefits are assumed to accrue annually each year indefinitely. To be compatible with the engineering cost estimate, benefits would have to be terminated at the end of Year 20.[27] This asymmetry cannot be dismissed on the ground that engineering costs and cancer risk reduction benefits are both annualized because costs are annualized and benefits are not.

4. Cancer risk reductions are calculated in a manner inconsistent with the OEHHA LNT risk model.

The CBA correctly recognizes that OEHHA's unit cancer risk estimate reflects risk reductions accruing over a 70-year lifetime.[28] To obtain an estimate of the number of cancer cases per year that would be prevented through drinking water treatment, CDPH divides OEHHA's 70-year lifetime risk estimate by 70 and multiplies it by the product of the amount of Cr VI calculated to be removed and the number of persons whose exposure would be thereby reduced. This constant number of cases is then assumed to be prevented each year.

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[27] More accurately, they would have to be phased out. Exposure reductions obtained for Year 1 through Year 20 would fall to zero in Year 21.

[28] California Department of Public Health (2013), PDF p. 82.

If the CDPH methodology works, is does so only under steady-state conditions that are obtained after an MCL has been attained for 70 years and there is no one left in the population who has been exposed to a combination of pre- and post-treatment Cr VI concentrations. Taking at face value the OEHHA linear no-threshold (LNT) cancer risk model, however, requires us to count only one year's exposure reduction for each year the MCL has been attained. Thus, in Year 1 after treatment has been installed and the MCL has been met, water consumers capture only 1/70th of these annual risk reduction benefits. That is, to obtain the correct value for cancer cases prevented in Year 1, one must divide the 70-year lifetime cancer incidence by 70 (reflecting the number of cases prevented per year under steady-state conditions), then divide by 70 again (reflecting the fact that consumers have only experienced 1/70th of the exposure reductions that are necessary to obtain 1/70th of the lifetime risk reduction). For each year from Year 2 through Year 70, the number of cancer cases prevented rises by an amount equal to the number of cases reduced in the first year.

When Year 71 begins, the number of cases prevented each year will finally be in steady-state and equal the annual figure used in the CBA. Until then, however, the steady-state figure will be wrong if the OEHHA LNT risk model is correct.

Figure F and Figure G show for large and small water systems, respectively, how the CDPH steady-state benefits model differs from the OEHHA LNT risk model, using the 1 μg/L MCL for illustrative purposes. Under the CDPH model, the full reduction of cancer cases is realized in the first year after the MCL is achieved. But this is incompatible with the OEHHA LNT risk model, which requires a full 70 years of annual exposure reductions to occur before lifetime risk reductions are realized. Linearity in the OEHHA model means that each year's exposure reduction makes the same contribution to risk reduction; hence, the green line representing the OEHHA LNT model slopes upward at a constant rate for 70 years. In Year 71, and for every year thereafter, the CDPH model is the same as the OEHHA risk model.

This methodological inconsistency has significant ramifications for the calculation of engineering cost per theoretical cancer case prevented. The cost-effectiveness ratios presented in the CBA apply, if at all, only under steady-state conditions—i.e., in Year 71 after the MCL is attained, and every year thereafter.

Figure F: Theoretical Cancer Cases Prevented per Year; CDPH Steady-State Model vs. OEHHA LNT Risk Model (LWS, 1 μg/L)

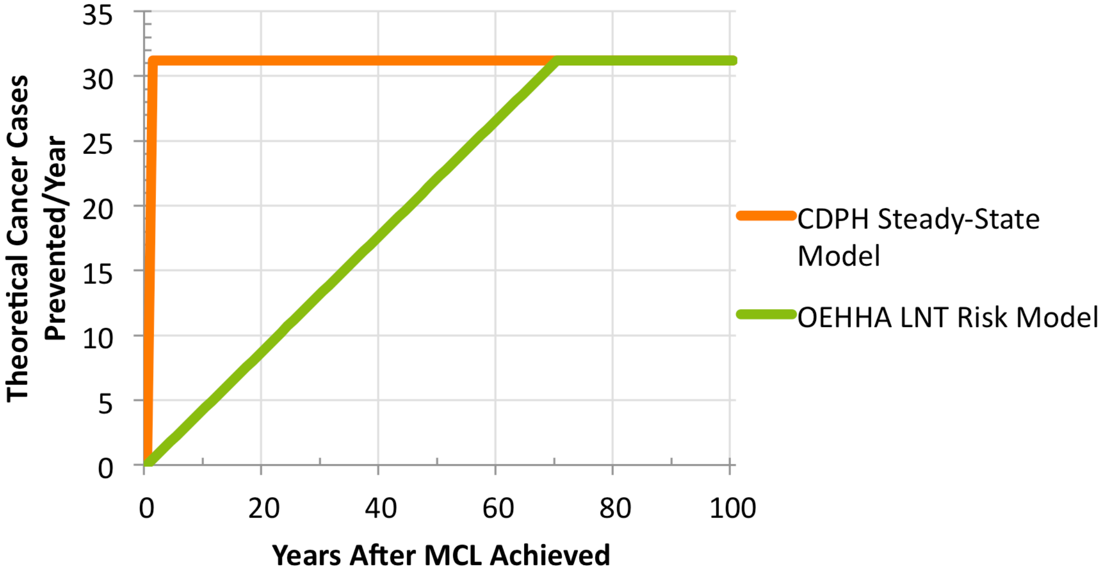
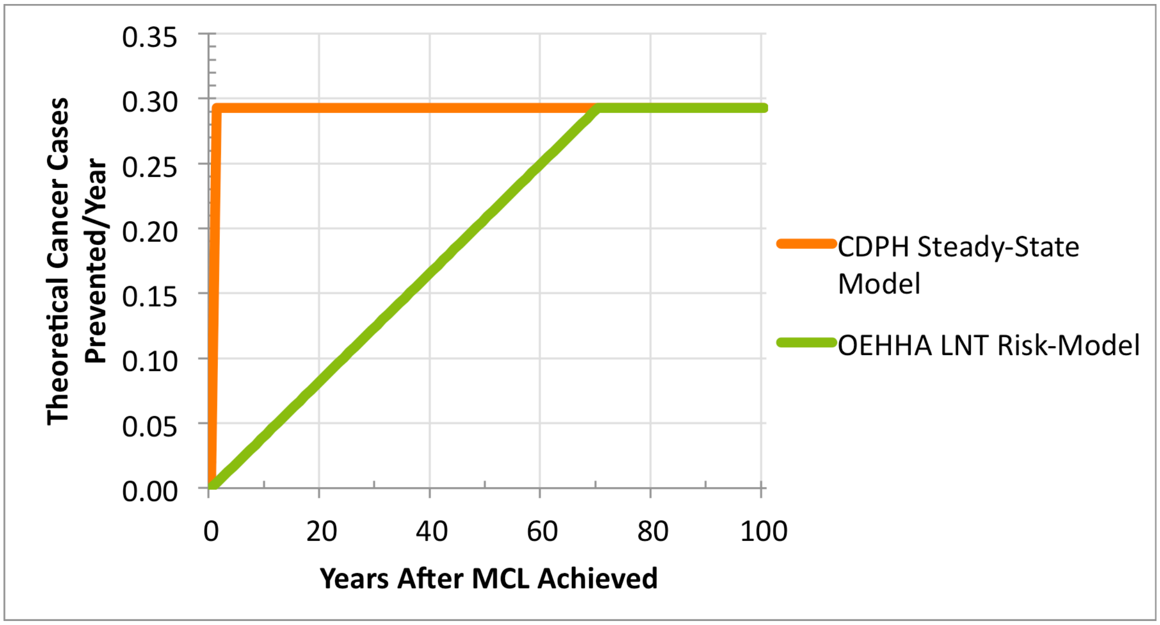


Figure G: Theoretical Cancer Cases Prevented per Year; CDPH Steady-State Model vs. OEHHA LNT Risk Model (SWS, 1 μg/L)



5. The cost-benefit analysis does not account for the cessation lags required to maintain consistency with the biology behind the OEHHA LNT risk model.

Every mode of action hypothesis for cancer anticipates a lag between the relevant biological insult and the physical manifestation as cancer. The length of this lag is both uncertain (i.e., it may not be known, or even knowable) and variable across the population (i.e., there may be a distribution of lags). Even though the length of this lag is uncertain, it is known with certainty to be greater than zero. If the ingestion of Cr VI in drinking water at low concentrations does cause cancer in humans, there will be a positive lag between exposure and its manifestation. No cancer case is expected to occur today from Cr VI ingestion today.

Conversely, cancer risk is not eliminated immediately after exposure stops. Rather, there is a *cessation l*ag defined as the biological interval between the reduction in exposure and the reduction in risk.[29] An accurate portrayal of the risk reductions associated with reducing Cr VI ingestion via drinking water requires an honest attempt to estimate the cessation lag.[30] If science does not permit estimation, then a sensitivity analysis is required to explore the effects of alternative hypothesized lags.

When a cessation lag is accounted for, benefits are delayed. Therefore, the number of cancer cases prevented is lower in every future year than they appear to be when the cessation lag is ignored.

6. Future costs are discounted but future benefits are not.

Discounting is the standard method in benefit-cost analysis for accounting for positive rates of time preference. Future streams of costs and benefits must be discounted to obtain present-value equivalents.

In the CBA, the capital cost of installing treatment technology is amortized using a 7% discount rate. This allows the expenditure to be borne in the first year but its repayment to be spread out over the lifetime of the capital. However, the CBA does not discount future benefits at all. The prevention of a cancer case is assumed to have the same value in Year 20 as it does on Year 1.

This is a serious methodological error. People prefer benefits that are realized earlier rather than later. If they were given a choice between spending any specific amount to reduce a fixed level of cancer risk in 2015 (i.e., Year 1) or 2084 (i.e., Year 70), consumers will prefer to reduce the fixed level of risk in 2015. Looked at from the other side, if

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[29] U.S. Environmental Protection Agency (2010), p. xi.

[30] U.S. Environmental Protection Agency (2010), p. 7-5.

consumers have the choice to reduce a fixed level of cancer risk in Year 1 or Year 20, they will be willing to pay less to reduce cancer risk in year 20. A similar story can be told about exposure reductions; consumers are willing to pay more to avoid an exposure in Year 1 than in Year 20.[31]

No reason is given in the CBA explaining why costs were discounted but benefits were not. An explanation would have been useful given that this practice has no support in economic theory and thus violates generally accepted practice in cost-benefit analysis. For decision-makers and the public, the implications of this error are clear. The true cost-effectiveness calculations are much higher than those reported in the CBA.

7. Likely engineering costs are improperly compared with theoretical benefits.

The CBA may do a credible job of approximating typical engineering costs; a proper review of that must be done by a qualified water treatment engineer. As noted above, the failure to account for opportunity cost imparts a material downward bias in the estimate of true compliance costs even of the engineering cost estimate is objectively estimated. However, the CBA does not even attempt to estimate benefits objectively.[32] Instead, the CBA takes at face value OEHHA's unit cancer risk estimate without conducting any review to ensure that it meets the objectivity requirements of CBA.

Had CDPH performed such a review, it would have quickly discerned that OEHHA's unit cancer risk value does not meet the required objectivity standard for the simple reason that OEHHA did not intend for or design it to do so. OEHHA's cancer risk estimate is not objective but intentionally precautionary. It is very unlikely to understate the true (but unknown) low-dose human cancer risk from ingesting Cr VI in drinking water and it is highly likely to overstate it. This bias in favor of precaution reflects OEHHA's established risk assessment practice. While the merits of that practice are beyond the scope of this review, its incompatibility with CBA is not in dispute.[33]

CDPH should be commended for correctly describing the OEHHA unit risk value as yielding estimates of *theoretical* cancer cases.[34] However, theoretical risk is not the same as actual or statistically expected risk, so any derivation of the *theoretical* number of cancer

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[31] Approximately 25% of the population does not expect to live 20 more years. For these individuals, cancer risk reductions in Year 20 have no value at all.

[32] This is true independently of the lack of discounting noted in Section I.B.5 above.

[33] This is not an original observation or a new problem. See, e.g., Nichols and Zeckhauser (1986), Office of Management and Budget (1990), and Kopp, Krupnick and Toman (1996).

[34] California Department of Public Health (2013), PDF pp. 80-84.

cases is not the same thing as a derivation of the statistically *expected* number of cancer cases. It is not even a plausible upper-bound estimate of the *expected* number; it is an extrapolation to a disproportionately exposed population of the number of cases that would be expected if the precautionary unit risk estimate, and the LNT model used to extrapolate it, were both true.

It is a material violation of generally accepted principles of cost-benefit analysis to compare calculated "typical" costs with calculated "theoretical" benefits. For this reason, the benefits assessment and calculations of cost-effectiveness in the CBA are invalid, unreliable, and misleading. For every alternative MCL and water system category, the reported number of cancer cases prevented materially overstates the true number. That, in turn, means that the CBA understates the cost per cancer case prevented for every MCL.

C. Departures from generally accepted principles of cost-benefit analysis are likely to cause decision-makers and the public to misunderstand the costs and benefits of the proposed drinking water standard.

These major methodological defects do not have random effects on the CBA, nor do their effects in any way "wash out." Each methodological defect results in either the

Table 3: Major Methodological Defects in the Cost-Benefit Analysis Systematically Understate Costs or Overstate Benefits

Methodological Choices that Systematically Understate Costs	*Methodological Choices that Systematically Overstate Benefits*
Large systems that consist of multiple small systems under common ownership are misclassified as large systems.	Engineering costs are estimated over a 20-year horizon, but benefits are estimated with an infinite horizon.
Cost is estimated as engineering cost, not opportunity costs.	Benefits are estimated using a steady-state model inconsistent with the OEHHA linear no-threshold risk model.
Engineering costs do not include water systems that would be required to install treatment as a result of future monitoring.	Cancer risk reductions are calculated in a manner inconsistent with the OEHHA linear no-threshold risk model.
	Future benefits are not discounted.
	Likely engineering costs are improperly compared with theoretical benefits.

systematic understatement of cost or the systematic overstatement of benefits. This is summarized in Table 3 below.

Without an accurate portrayal of costs, benefits, and cost-effectiveness, legislators, executive decision-makers, and the public are all misled concerning the likely consequences of regulatory action. In particular, given the systematic nature of the major methodological errors in the CBA, legislators, executive decision-makers, and the public will all garner a much more sanguine view of the consequences than they would have gained from an unbiased portrayal.

Economically disadvantaged households, whether or not they reside in communities designated as "disadvantaged" or "severely disadvantaged," will be especially harmed by the systematic biases in the CBA's cost and benefit estimates. To be sure, if they are served by a small water system, they need no error correction to understand how deeply harmed they would be by any of the MCLs under consideration. However, they cannot gain that understanding without error correction if they are served by a large water system.

D. Correcting the largest methodological errors on the benefit side of the cost-benefit analysis gives a substantially different picture of the economic feasibility of alternative drinking water standards.

Among the methodological errors discussed in Section III.B above, it is not easy to tell *a priori* which errors have the largest effects on the CBA. Of the three major methodological errors on the cost side, the largest is likely to be the use of engineering cost instead of opportunity cost. No data appear to be readily available that could be used to roughly approximate the magnitude of this error. Hints about its magnitude are apparent when one considers that the CBA estimates engineering cost per service connection for small systems at 6-12% of median community household income, and up to 20% of median community household income for communities classified as severely disadvantaged. Half of all households in any community have income below the median, so there is no question that most households served by small systems face a devastating loss of income. The difference between engineering cost and opportunity cost could be extremely large given the magnitude of effects, and opportunity cost likely includes new health risks.

Some of the major errors on the benefits side are easier to correct. This section is devoted to providing those corrections.

1. Accounting for the linear no-threshold model used by OEHHA to estimate cancer risk reduces benefits.

As previously noted, the CBA relies on a steady-state model for counting theoretical cancer cases prevented, an approach that is inconsistent with OEHHA's LNT cancer risk model. OEHHA's model assumes that cancer risk is a linear function of lifetime exposure, and that there is no exposure below which cancer risk is indistinguishable from zero. Thus, every microgram of hexavalent chromium ingested per unit of body weight is assumed to pose the same cancer risk regardless of when it is ingested.

CDPH's error can be corrected by replacing the steady-state benefits model with one consistent with the OEHHA LNT risk model. When this is done, cost-effectiveness ratios rise exponentially over those reported in the CBA. Figure H and Figure I show for large and small water systems, respectively, that the cost-effectiveness ratios reported in the CBA are valid, if ever, only beginning in Year 71 and beyond. The green curves represent cost-effectiveness ratios based on the OEHHA LNT risk model. The vertical axis is shown logarithmically to make these revised ratios easier to read. The USEPA's recommended default value for preventing a random premature mortality is plotted in blue.

For large systems (Figure H), the corrected engineering cost-effectiveness ratio grows infinitely high as the number of years considered approaches zero. It takes about 40 years of compliance with a 1 μg/L MCL before engineering cost declines below $100 million per theoretical cancer case prevented. During the first two years of compliance, the engineering cost-effectiveness ratio exceeds $1 billion per theoretical cancer case prevented. For small systems (Figure I), the corrected engineering cost-effectiveness ratio exceeds $1 billion per theoretical cancer case for the first 20 years of compliance, which coincidentally is the assumed lifetime of the capital.

These examples use the 1 μg/L MCL for illustrative purposes, but the lessons apply generally. Figure J and Figure K show for large and small systems, respectively, how engineering cost per theoretical cancer case prevented, properly calculated, varies across the full range of MCLs analyzed in the CBA. As before, a $\log_{10}$ scale is used on the vertical axis to make these differences easier to see.

For large systems, engineering cost-effectiveness after 20 years of MCL compliance (when the lifetime of the capital investment in treatment expires) ranges from $10 million to $178 million per theoretical cancer case prevented. For small systems, engineering cost-effectiveness after 20 years of MCL compliance ranges from $323 million to $1,031 million per theoretical cancer case prevented. All values in these ranges greatly exceed the USEPA recommended default value for preventing a random premature mortality.

Figure H: Engineering Cost per Theoretical Cancer Case Prevented; CDPH Steady-State Model vs. OEHHA LNT Risk Model (LWS, 1 μg/L)

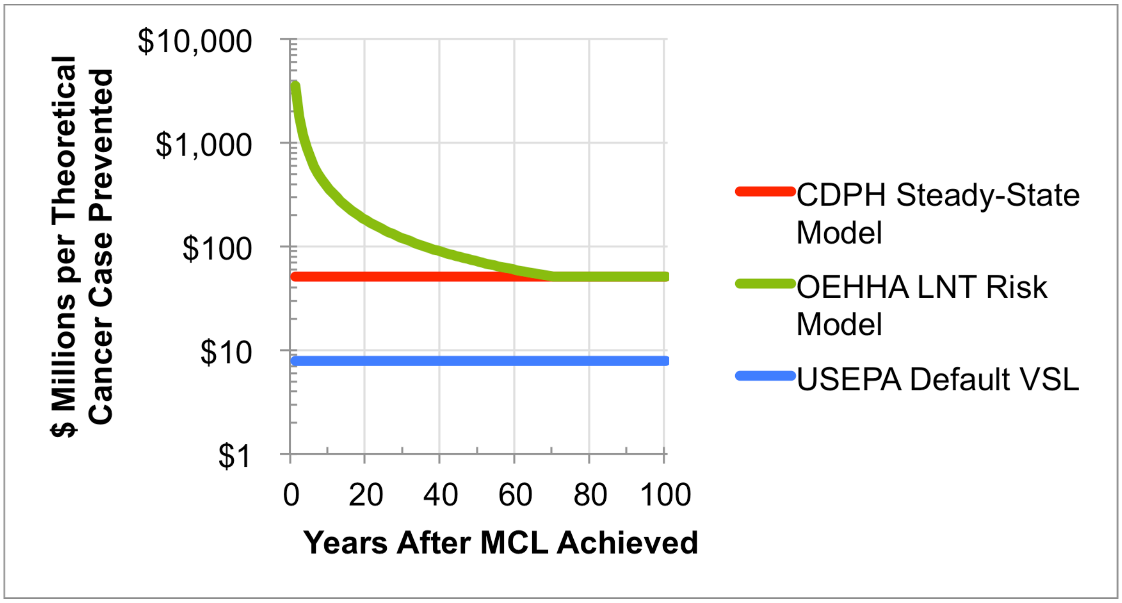


Figure I: Engineering Cost per Theoretical Cancer Case Prevented; CDPH Steady-State Model vs. OEHHA LNT Risk Model (SWS, 1 μg/L)

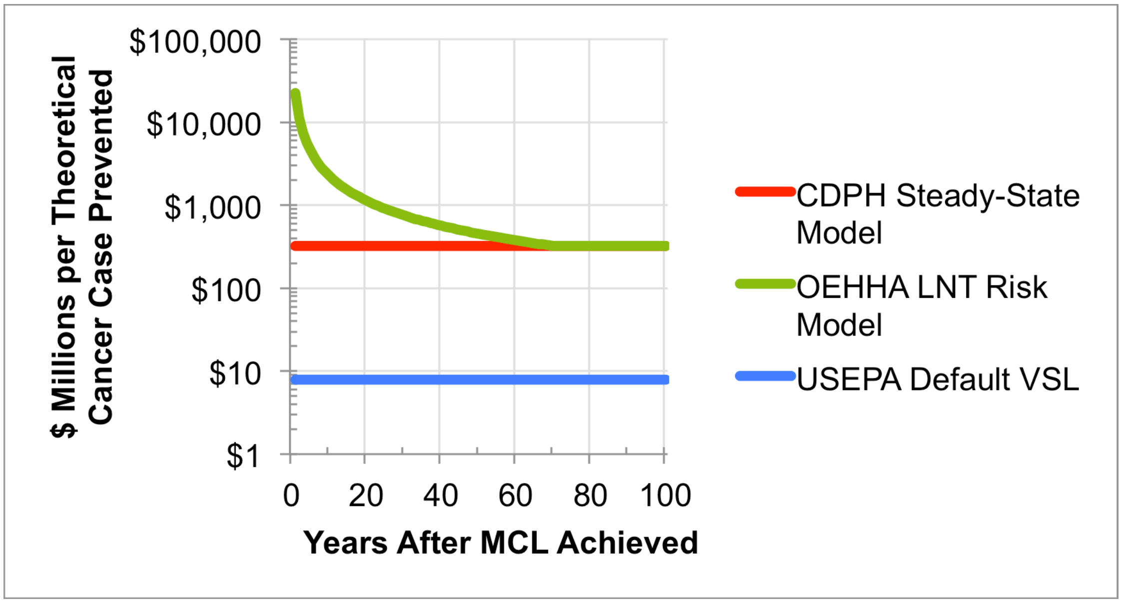


Figure J: Engineering Cost per Theoretical Cancer Case Prevented; CDPH Steady-State Model vs. OEHHA LNT Risk Model (LWS, Range of MCLs)

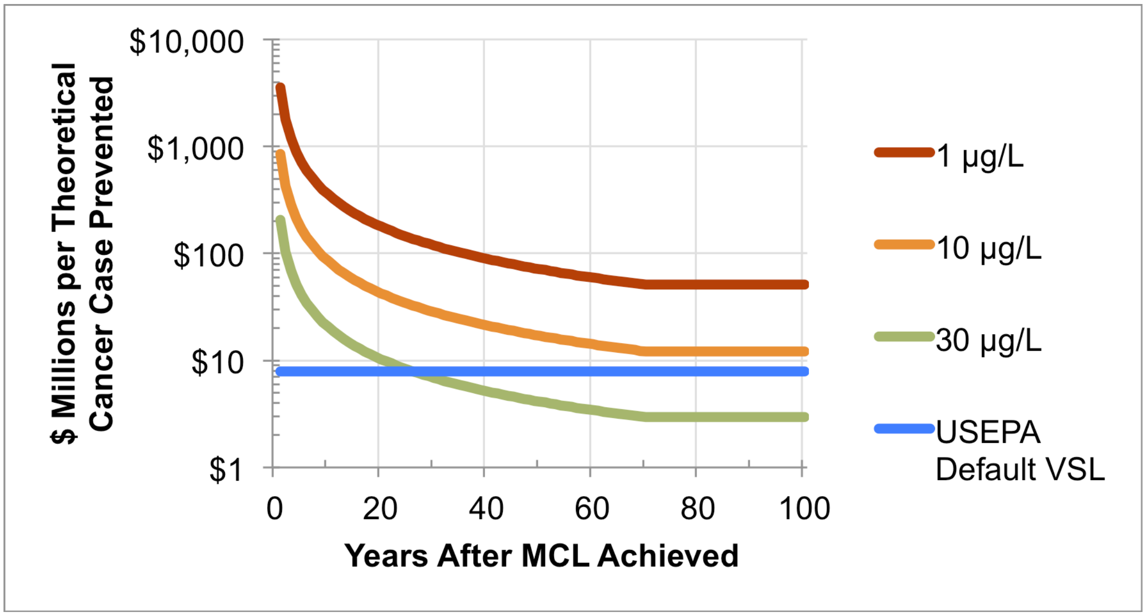
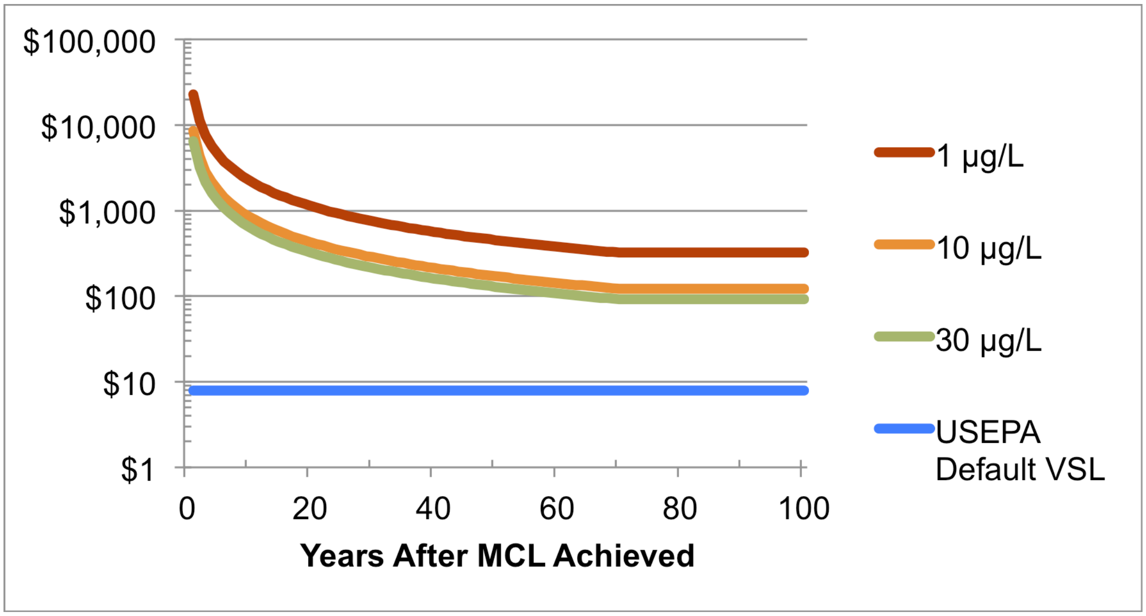


Figure K: Engineering Cost per Theoretical Cancer Case Prevented; CDPH Steady-State Model vs. OEHHA LNT Risk Model (SWS, Range of MCLs)



2. Making the cost horizon consistent with the benefits horizon increases costs.

As noted previously, the CBA uses a 20-year time horizon for counting engineering costs but assumes an infinitely long time horizon for counting theoretical cancer risk reductions. To restore a proper comparison, a choice must be made concerning what time horizon to use, and both costs and benefits must be counted for the same time horizon.

Because the OEHHA LNT risk model requires 70 years of MCL compliance before steady-state conditions apply, it makes sense to use a time horizon of at least that length. For purposes of this analysis a time horizon of 100 years is used, ensuring that there is an extended period (30 years) during which steady-state conditions actually apply.

A 100-year time horizon requires that treatment costs be replicated in years 21, 41, 61, and 81. For purpose of this analysis, the estimated annualized costs in the CBA for three alternative MCLs (1 μg/L, 10 μg/L, and 30 μg/L) are replicated for all 100 years. To be clear, this means that in every year from Year 1 through Year 100, large public water systems in California must bear additional real (i.e., inflation-adjusted) engineering costs ranging from $12 million (30 μg/L) to $1.6 billion (1 μg/L). Small systems must bear additional engineering costs each year of $0.4 million to $95 million.[35]

In the following section, all costs and benefits for the same 100-year time horizon are discounted to present value equivalents. This is the preferred way to compare costs and benefits.

3. Discounting benefits the way costs are discounted reduces benefits.

As noted previously, future costs are discounted (at 7%) in the CBA but benefits are not discounted at all. This materially overstates benefits and leads to an apples-to-oranges comparison of discounted costs and undiscounted benefits. Such a comparison is inherently improper, rendering all cost-effectiveness ratios in the CBA invalid and misleading.

Figure L and Figure M show for large and small systems, respectively, how undiscounted and discounted benefits differ. Undiscounted benefits using the OEHHA LNT risk model are shown in green and are the same as previously displayed in Figure F (large systems) and Figure G (small systems). The orange curves are the same future benefits discounted at the 7% rate used in the CBA to discount future costs. Undiscounted benefits rise until Year 70 and remain constant every year thereafter. Discounted benefits rise for

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[35] A better approach, but one that cannot be done under the time constraint for public comments, is to reconstruct actual, undiscounted costs for each year, then replicate these costs for the 20-year periods beginning in Years 21, 41, 61, and 81.

Figure L: Theoretical Cancer Cases Prevented per Year; Undiscounted and Discounted at 7% (LWS, MCL = 1 μg/L)

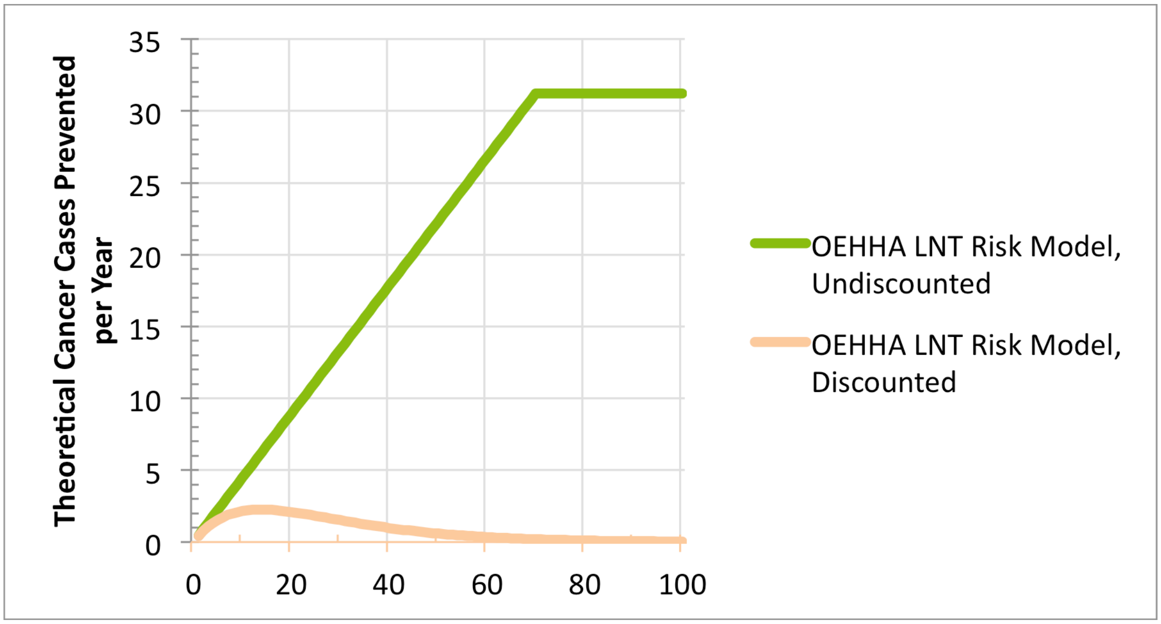
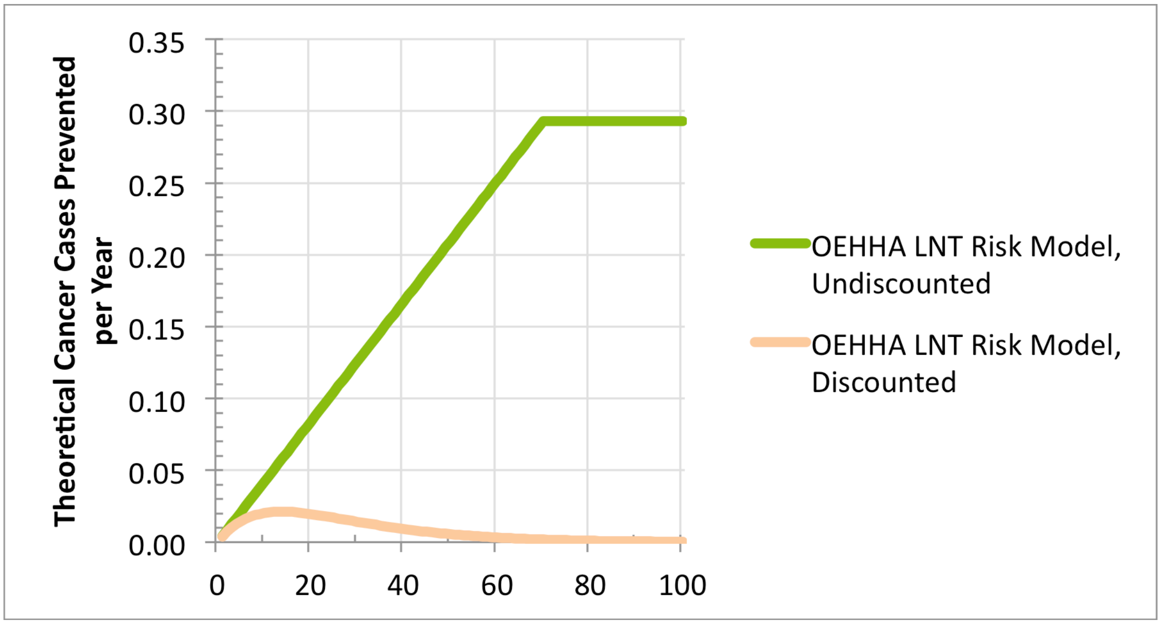


Figure M: Theoretical Cancer Cases Prevented per Year; Undiscounted and Discounted at 7% (SWS, MCL = 1 μg/L)



the first several years but decline asymptotically thereafter. This is both mathematically necessary and intuitively sensible: reductions in cancer risk that are realized early are more valuable than reductions in cancer risk that are delayed for decades.

When benefits are discounted along with costs, the cost-effectiveness of drinking water treatment changes dramatically. Obtaining present value cost-effectiveness requires calculating the present value of both costs and benefits before dividing costs by benefits. Present values are obtained by discounting future costs and benefits for the same number of years at the same interest rate. Costs are adjusted to account for capital replacement in years 21, 41, 61 and 81. This is approximated by using the annualized cost estimates in the CBA for each year from Year 1 to Year 100.

Table 4 summarizes, for the range of MCLs analyzed in the CBA, results when future benefits are discounted as well as future costs, and both benefits and costs are counted for

Table 4: Present Value Costs (in $ Millions), Theoretical Benefits (in Cancer Cases), and Cost-Effectiveness (in $ Millions per Theoretical Cancer Case)

	MCL = 1 µg/L		**MCL = 10 µg/L**		**MCL = 30 µg/L**	
	SWS	LWS	SWS	LWS	SWS	LWS
Costs ($M) [a]	$ 1,257	$21,120	$ 183	$ 1,888	$ 5	$ 155
Theoretical Benefits (Cases) [b]	0.787	83.8	0.303	31.5	0.011	10.7
C-E Ratio ($M/Case) [c]	$ 1,600	$ 250	$ 600	$ 60	$ 450	$ 14
Multiple of USEPA Default [c, d]	200	32	76	7.6	58	1.8

[a] Millions; [b] 3 significant figures; [c] 2 significant figures; [d] $7.9 million.

100 years. Note that present value engineering costs are reported in millions of dollars. Present value large system engineering costs range from about $150 million (30 μg/L) to about $21,000 million (i.e., $21 billion), and small system engineering costs range from about $5 million (30 μg/L) to about $1,300 million (i.e., $1.3 billion) (1 μg/L).

Present value benefits (i.e., the present value number of theoretical cancer cases prevented over 100 years) can be understood as equivalent to the number of theoretical cancer cases prevented in Year 0 that is equivalent to the 100-year stream of theoretical cancer cases prevented based on the OEHHA LNT model. For large systems, present value benefits range from about 11 theoretical cancer cases (30 μg/L) to about 84 theoretical cancer cases (1 μg/L). For small systems, present value benefits range from about 1/100th of one theoretical cancer case (30 μg/L) to about 80/100ths of one theoretical cancer case (1 μg/L).[36]

Present-value engineering cost-effectiveness ratios are calculated by dividing present value engineering costs by present value cancer cases prevented. This calculation is equivalent to a one-time expenditure today for a one-time reduction in theoretical cancer cases today. For large systems, the engineering cost-effectiveness ratio ranges from about $14 million per theoretical cancer case (30 μg/L) to about $250 million per theoretical cancer case (1 μg/L). For small systems, the engineering cost-effectiveness ratio ranges from about $450 per million per theoretical cancer case (30 μg/L) to about $1,600 million (i.e., $1.6 billion) per theoretical cancer case (1 μg/L).

All of these cost-effectiveness ratios exceed the USEPA's recommended default value for the prevention of a random premature mortality, in most cases by very large multiples. Even the lowest cost-effectiveness ratio (large systems at 30 μg/L) is 1.8 times the USEPA's recommended default estimate of the value of preventing a random premature mortality (not a random cancer case).

4. Accounting for cessation lags in cancer risk reduction reduces benefits.

As noted previously, the CBA does not incorporate any cessation lag in its estimates of the numbers of theoretical cancer cases prevented. More specifically, reductions in Cr VI exposure are assumed to be converted instantaneously into reduced cancer incidence. This assumption is incompatible with every biological mode of action hypotheses that animates cancer risk assessment, including the OEHHA LNT risk model.

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[36] These figures are aggregated across all covered water systems. They are not cancer cases prevented per water system.

Table 5: Present Value Cost-Effectiveness for Three Alternative Cessation Lags Across the Range of Alternative MCLs Analyzed

Lag in Years	MCL = 1 µg/L		MCL = 10 µg/L		MCL = 30 µg/L	
	SWS	LWS	SWS	LWS	SWS	LWS
0	$1,600	$250	$600	$60	$450	$14
5	$2,300	$363	$869	$86	$656	$21
10	$3,314	$522	$1,252	$124	$945	$30
20	$6,900	$1,088	$2,608	$259	$1,968	$63

$ Millions per theoretical cancer case prevented.

Without reliable scientific information concerning the length of the average cessation lag, the best that can be done to account for it is to perform a sensitivity analysis capturing its plausible magnitude. Table 5 shows the effect on present value engineering cost-effectiveness of three alternative cessation lags: 5, 10, and 20 years. Every cessation lag reduces the benefits of treatment and has no effect on engineering costs. These benefit reductions are manifest in higher cost-effectiveness ratios across the board. A 5-year lag increases the engineering cost-effectiveness ratio by about half; a 10-year lag doubles it; and a 20-year lag doubles it again.

5. Accounting for less-than-certain causality in dose-response

The corrections made so far fix errors CDPH committed in converting OEHHA's LNT risk estimates into theoretical cancer cases prevented. There is another error the magnitude of which may exceed all others: the assumption that OEHHA's unit cancer risk value captures a true causal relationship between low-level human ingestion of Cr VI in drinking water and cancer.

OEHHA's Public Health Goal (PHG) does not resolve the question of biological causation.[37] Rather, it manages this uncertainty by simply assuming that the relationship is casual. The unit risk estimate in the PHG is therefore not a statistical "best" estimate of a one-in-one-million cancer risk, nor is it an estimate that minimizes the weighted sum of errors that result from under- and overestimation. As noted above, these attributes of the PHG make it incompatible with use as an input to cost-benefit analysis. CBA requires that unbiased estimates of human health risk be used to estimate both baseline risks and the benefits of reducing them.

Uncertainty about causation combined with statutory anti-backsliding provisions mean that precautionary standard-setting is essentially impervious to any advance in scientific knowledge that reduces the estimate of human cancer risk at environmentally relevant doses. For example, if new research were published tomorrow proving that there is a threshold below which the ingestion of Cr VI in drinking water poses no cancer risk, it would be extremely difficult (and perhaps impossible) for CDPH to revise or rescind a promulgated MCL.

This statutory constraint on regulatory decision-making has substantial social costs. First, it commits California residents to bear the costs of water treatment for which today's estimated cancer risk reduction benefits may be rendered invalid by tomorrow's science. Second, it strongly disincentivizes research targeted at resolving scientific uncertainty that could reduce the risk estimate. If the results of such research cannot affect regulatory decision-making, then the practical value of this information is nil. When research has no practical value, there is no incentive to fund it. These social costs are not counted in the CBA. It counterfactually assumes that the unit cancer risk estimate derived by OEHHA is scientifically certain.

While it is possible that the dose-response relationship assembled by OEHHA is exactly correct and causal, there are important reasons for thinking that it is not. For example, observable effects in rats and mice at very high doses must be proportional to unobservable effects at very low doses. In addition, humans must respond to ingested Cr VI in exactly the same way that rats and mice do. Both of these aspects of the OEHHA LNT risk model are assumptions that have always been scientifically controversial.

When faced with untestable assumptions that drive analytic results and sufficient data for simulation are not available, the correct thing to do is to perform a sensitivity analysis that captures alternative plausible assumptions. That would have been quite simple for the CDPH to perform, but the CBA contains no evidence that it did so.

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[37] California Environmental Protection Agency Office of Environmental Health Hazard Assessment (2011).

Figure N and Figure O show for large and small systems, respectively, the results of a simple sensitivity analysis on present value cost-effectiveness of relaxing the assumption of certainty in dose-response. The right-hand edge of each curve is the cost-effectiveness ratio in the special case where OEHHA's dose-response function is true (i.e., OEHHA's dose-response relationship is causal with a probability equal to 1). The cost-effectiveness ratio cannot decline any further, and it must be higher if there is any doubt about causation. As doubt rises, and the probability of causation approaches zero, the cost-effectiveness ratio rises exponentially.

Consider the example in which there is a 50% likelihood that OEHHA's dose-response relationship is true and a 50% likelihood that Cr VI does not cause cancer at drinking water concentrations. The engineering cost-effectiveness ratio for large water systems at a 30 μg/L MCL is not $14 million per cancer case prevented, but $29 million. The same scenario doubles the engineering cost-effectiveness ratio for small systems from about $450 million to over $900 million per cancer case prevented. As the causation probability declines toward zero, present value cost-effectiveness ratios rise into the billions of dollars per cancer case prevented.

The correct causation probability is, of course, unknown, and reputable biologists likely have different views. Figure N and Figure O provide a handy way for decision-makers and the public to understand the implications of alternative scientifically-grounded expert opinions. They can choose any probability value on the horizontal axis and obtain an approximate cost-effectiveness ratio by drawing a vertical line to the MCL of interest.[38]

6. Accounting for opportunity costs would increase costs.

As noted previously, the correct way to measure *cost* is in terms of opportunity cost. This is the value of benefits foregone resulting from devoting scarce resources for the installation and operation of drinking water treatment. Opportunity cost is the monetized value of the benefits households must give up in order to pay hundreds or thousands of dollars more per year for drinking water.

Unfortunately, the CBA uses direct engineering cost instead of opportunity cost as its proxy for compliance cost. This imparts a substantial downward bias to the CBA's compliance cost estimates. The magnitude of this bias is unknown, though for specific situations it could be estimated. For example, a specific public water systems can identify what it must give up if it has to install and operate treatment for Cr VI removal and cannot shift engineering costs to its customers. The value of what it must give up is the

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[38] The mental exercise proposed here relies on qualified and independent expert opinion concerning the most likely probability of causation. It would not be informed by the results of a public opinion poll or the opinions of activists and policy-makers.

Figure N: Present Value Cost-Effectiveness by Probability of Causation in Dose-Response (LWS)

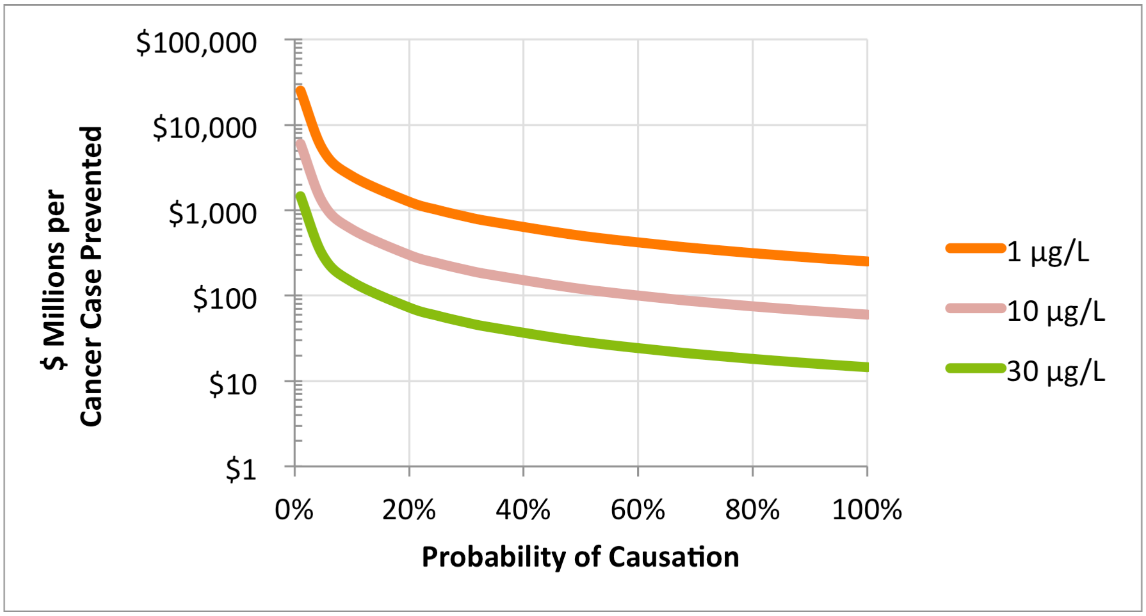
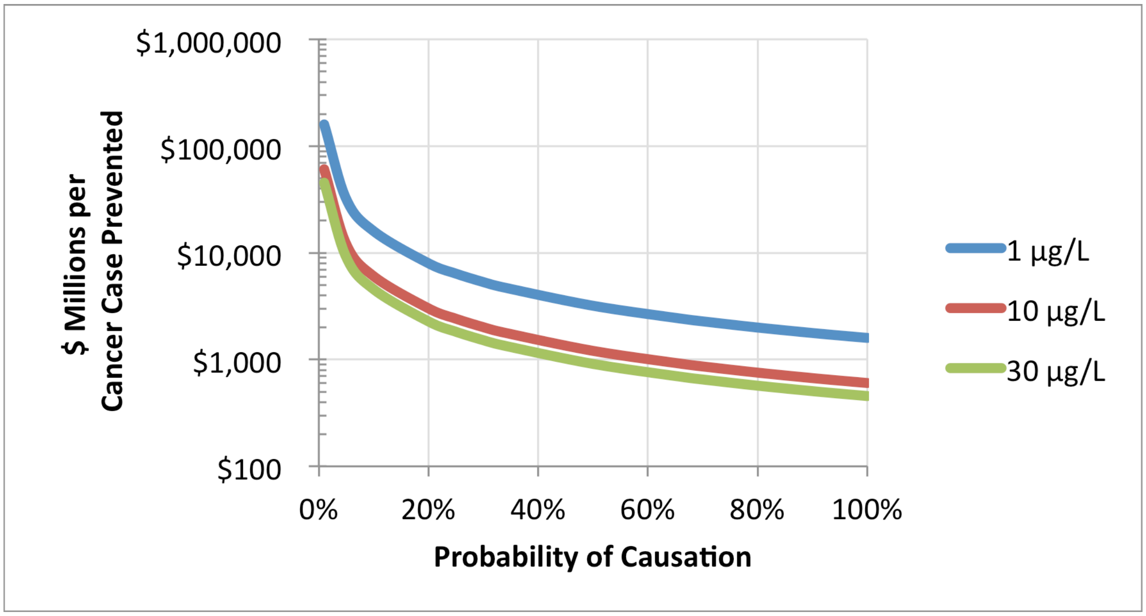


Figure O: Present Value Cost-Effectiveness by Probability of Causation Dose-Response (SWS)



opportunity cost to that water system of installing and operating drinking water treatment to remove Cr VI. If a water system can pass on these costs to its customers, then the correct approach is to ascertain what its customers must give up in order to pay the higher water bills that Cr VI treatment requires. The sum of the values they place on what they must give up is the opportunity cost of Cr VI treatment.

IV. Corrections Demonstrate that Every MCL Analyzed Is Economically Infeasible

This analysis of the CDPH cost-benefit analysis reveals a number of important lessons.

A. Taken at face value, the CBA shows that each of the MCLs under consideration Is economically infeasible for communities and households served by small water systems.

The CBA clearly shows that every alternative MCL analyzed is economically infeasible. Engineering cost is at least $87 million per theoretical cancer case prevented, about 10 times the USEPA recommended default monetary value for preventing a random premature mortality.

The monetized value of reduced cancer incidence is less than $4 per household per year at every MCL analyzed. This is less than the value of a package of 12 postage stamps.

Even after the monetary value of cancer reductions is subtracted, households served by small water systems face an implicit new tax on gross income ranging from $4,000 to $7,000 per year, depending on the stringency of the standard. Based on the income figures provided in the CBA, this represents 6-12% of median household income statewide, and as much as 20% of median household income in communities deemed severely disadvantaged.

B. Taken at face value, the CBA shows that MCLs of 15 μg/L and below are economically infeasible for communities served by large water systems.

The CBA clearly shows that MCLs below 15 μg/L are economically infeasible for large water systems. Engineering cost is as much as $51 million per theoretical cancer case prevented, about five times the USEPA recommended default monetary value for preventing a random premature mortality. At 15 μg/L, engineering cost-effectiveness is approximately equal to the USEPA recommended value for preventing a random premature mortality. It is as low as $3 million per theoretical cancer case prevented at 30 μg/L. After the monetary value of cancer reductions is subtracted, households served by large water systems face implicit new taxes as high as $278 per year. For no MCL would household net benefits be greater than $25 per year.

For households served by large systems, the value of cancer reduction benefits is trivially different from zero. At a 20 μg/L MCL and above, the CBA projects that one dollar's worth of benefit would be obtained per year for every 20,000 households. At a 1 μg/L MCL, one dollar's worth of benefit would be obtained per year for every 100,000 households.

C. The CBA contains a number of methodological errors that materially understate costs or overstate benefits.

As detrimental as they are for establishing economic feasibility, the results presented in the CBA make that case look more attractive than it really is. Several methodological errors were committed that either understate costs or overstate benefits. These errors are clearly material; a rational decision-maker who might be inclined to devise a way to exempt small systems from a Statewide MCL would not do so after considering the cost-effectiveness of treatment after these errors are corrected.

D. When these errors are removed, every MCL under consideration is shown to be not economically feasible regardless of system size.

The analysis presented here corrects quantitatively for five major errors: (1) using a benefit model inconsistent with the OEHHA LNT risk model, (2) using the same time horizon for counting both costs and benefits, (3) discounting future costs but not future benefits, (4) failing to account for cessation lags, and (5) failing to account for less-than-certain causation. One of these corrections materially increases the cost estimate; the other four materially decrease the benefit estimate.

After replacing the CDPH's invalid steady-state benefit model with one compatible with the OEHHA risk model, discounting benefits as well as costs, and converting benefits and costs into present values, cost per theoretical cancer case prevented for large systems facing the least stringent MCL rises from the $3 million per case reported in the CBA to $14 million. Small system cost-effectiveness rises from $92 million to $450 million per theoretical cancer case prevented. The cost-effectiveness ratio is worse for every other combination of system size and MCL.

Even before corrections are made to the CBA methodology, the monetized value of cancer reduction benefits is reported in the CBA to be minor—between a fraction of a penny and $4 per household per year. Correcting the benefit assessment model reduces the 100-year sum of cancer cases prevented by a factor of one-third. Discounting reduces them by another factor exceeding 60. That means the monetized value of cancer risk reductions is about 1/200th of the values derived from the CBA: the $4 per household per year valuation derived from the CBA is really just two cents per household per year.

E. Promulgating any of the MCLs under consideration would impose a new gross income tax on virtually all Californians.

For households served by large water systems, a state Cr VI drinking water standard in the range of values analyzed represents at worst a small loss of income. For households served by small water systems, however, the net income loss would be exceptionally large—6-12% of median household income.[39] These income losses are net of the value of cancer risk reduction, so they are equivalent to a new income tax that simply expropriates 6-12% of gross household income.

F. These new income taxes would be devastatingly harmful for disadvantaged California households served by small water systems.

Because drinking water treatment costs are paid by water system customers, and customers' water use is not much affected by income, the most severe net income losses would be imposed on those least able to afford it. The CBA calculates small system engineering costs in the range of $4,000 to $7,000 per household, depending on the MCL selected. That's up to 20% of median household income in communities defined by CDPH as "severely disadvantaged."

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[39] Recall that the CBA reports statewide median household income as $58,553. See California Department of Public Health (2013), PDF p. 82.

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