NATIONAL ENERGY TECHNOLOGY LABORATORY

TECHNO-ECONOMIC ANALYSIS OF CHEMICAL PRECIPITATION FOLLOWED BY LOW HYDRAULIC RESIDENCE TIME BIOLOGICAL TREATMENT, INCLUDING ULTRAFILTRATION

CHAD ABLE

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Disclaimer

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ACRONYMS AND ABBREVIATIONS

ACI	Activated carbon injection
Ar	Argon
BACT	Best Available Control Technology
BAT	Best Available Technology
Btu	British thermal unit
CapEx	Capital expenditure
CO_2	Carbon dioxide
COE	Cost of electricity
CP	Chemical precipitation
DOE	Department of Energy
DSI	Dry sorbent injection
ELG	Effluent limitation guideline
EPA	Environmental Protection Agency
EPRI	Electric Power Research Institute
FGD	Flue gas desulfurization
ft	Feet
gpd, GPD	Gallons per day
gpm, GPM	Gallons per minute
h, hr	Hour
H_2O	Water
H_2S	Hydrogen sulfide
HCl	Hydrochloric acid
Hg	Mercury
HHV	Higher heating value
HRTR	High residence time reduction
ID	Induced draft
IOU	Investor-owned utility
ISO	International Organization for Standardization
kg	Kilogram
kJ	Kilojoule
kW, kWe	Kilowatt electric
LAER	Lowest Achievable Emission Rate
lb	Pound
LCOE	Levelized cost of electricity
LHV	Lower heating value
LRTR	Low residence time reduction
mg/L	Milligrams per liter
MMBtu	Million Btu
MW, MWe	Megawatt electric
MWh	Megawatt-hour
N/A	Not applicable/available
N_2	Nitrogen
NETL	National Energy Technology Laboratory
NOx	Nitrogen oxides
NPDES	National Pollutant Discharge Elimination System
NS	Not Specified
NSPS	New Source Performance Standards
NSR	New Source Review
NTU	Nephelometric turbidity unit
O&M	Operation and maintenance
O_2	Oxygen
ORP	Oxidation reduction potential
PC	Pulverized coal
PM	Particulate matter
POTW	Publicly owned treatment works
ppb	Parts per billion
ppm	Parts per million
ppmw	Parts per million, weight
ppt	Parts per trillion
PSD	Prevention of Significant Deterioration
PSES	Pretreatment Standards for Existing Sources
psia	Pound per square inch absolute
psig	Pound per square inch gauge
PSNS	Pretreatment Standards for New Sources
QGESS	Quality Guidelines for Energy System Studies
SCR	Selective catalytic reduction
SE^{+4}	Selenite
SE^{+6}	Selenate
SO_2	Sulfur dioxide
SO_3	Sulfur trioxide
TASC	Total as-spent cost
TDS	Total dissolved solids
TOC	Total overnight cost

TPC	Total plant cost
tpd, TPD	Tons per day
U.S.	United States
wt%	Weight percent
yr	Year
µS/cm	MicroSiemens per centimeter
°F	Degrees Fahrenheit

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EXECUTIVE SUMMARY

The United States Environmental Protection Agency (EPA) has issued a new ruling to update the Effluent Limitation Guidelines (ELGs) originally finalized in November 2015. [1] [2] [3] The goal of this proposed ruling was to produce more reasonable constraints on ELG components [4] and alleviate cost concerns [2] with the 2015 Best Available Technology (BAT) for flue gas desulfurization (FGD) wastewater treatment by suggesting a new BAT: chemical precipitation (CP) followed by low hydraulic residence time reduction (LRTR) biological treatment.

Public comments—regarding the proposed ruling—from industry leaders discussed concerns regarding EPA's test stream concentrations versus the expected stream concentrations of influent ELG components reflective of industry. [5] [6] Discussion centered around whether or not an LRTR system would provide a sufficient residence time for selenium removal; additional concerns suggested that EPA's cost basis was an underestimation of the required costs for procuring and operating a treat-and-discharge system of this type.

This study focused on evaluating the performance and cost of the CP/LRTR system versus the corresponding CP + high residence time reduction (HRTR) biological treatment system. In doing so, this study employed a combination of EPA's own evaluation of the two technologies [4], comments from industry leaders [5] [6], and vendor data. [7]

The performance results were based on EPA's own test data [8] at five plants used to evaluate the performance of the LRTR process. The influent and effluent concentrations were combined into a reduction ratio and compared to the necessary reduction ratio used in the design basis. Exhibit ES-1 displays the result of that effort—based on these results, the residence times may be insufficient for the reduction of high selenium FGD wastewater flows. However, the LRTR system appears to be more than sufficient for the necessary reduction of nitrates/nitrites in FGD wastewater.

Exhibit ES-1. Reduction ratios from EPA plant data [8] as compared to the selenium reduction required from the design basis

EPA Plant #	Average Selenium Reduction		Maximum Selenium Reduction		Required Selenium Reduction	
	Ratio	%	Ratio	%	Ratio	%
2019	10.3	90.3	58.7	98.3	301.2	99.7
2027	13.8	92.8	44.9	97.8		
2066	9.3	89.2	55.5	98.2		
2090	9.3	89.8	37.4	97.3		
2097	42.2	97.6	354.7	99.7		

Although the LRTR system is expected to be cheaper than the corresponding HRTR system, the levelized cost of electricity (LCOE) results across a number of sensitivity parameters (shown in

Exhibit ES-2 and Exhibit ES-3) suggest that the CP/LRTR treatment system will not have a significant cost impact relative to the CP/HRTR system—the difference in the LCOE values for the two base cases amounts to $0.1/MWh. The CP/LRTR and CP/HRTR systems are similarly impacted by parameters such as FGD flow rate, capital cost, non-fuel operation and maintenance (O&M) costs, retrofit difficulty factor, nitrate levels, and capital expenditure (CapEx) period.

Exhibit ES-2. CP/LRTR LCOE results along with sensitivity analyses

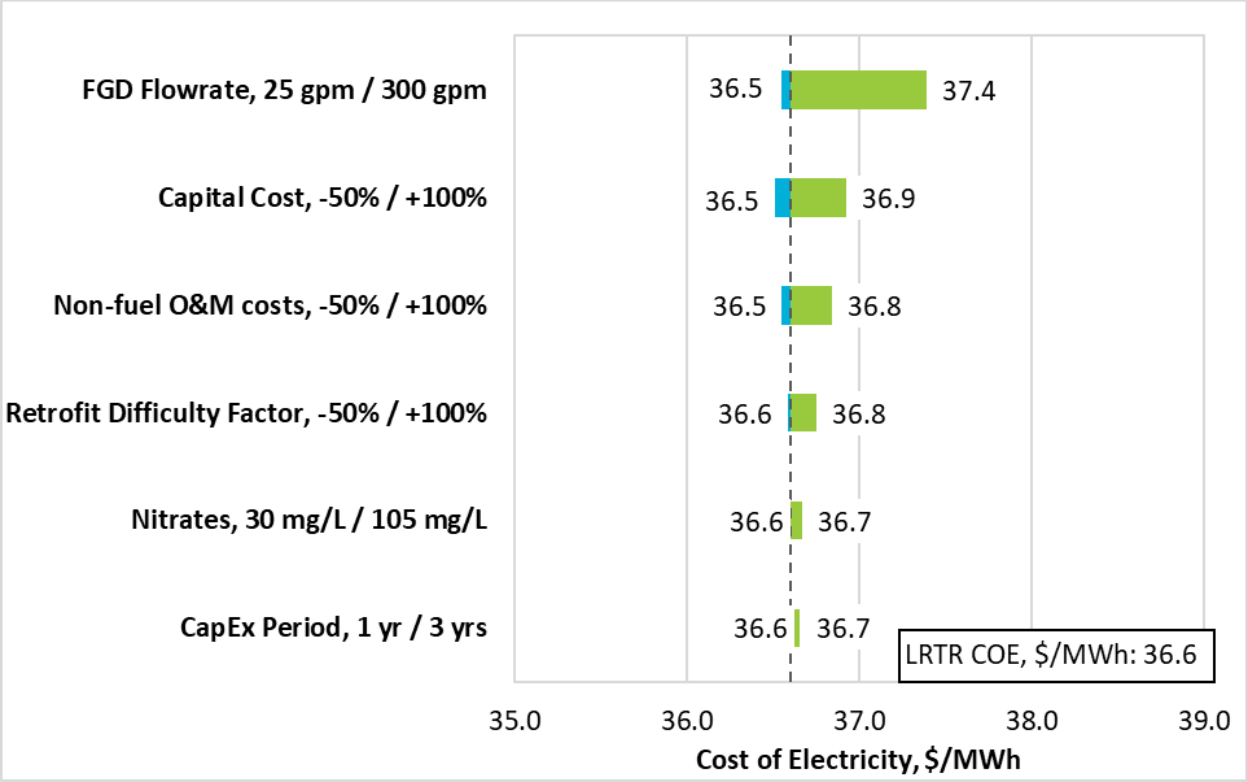


Exhibit ES-3. CP/HRTR LCOE results along with sensitivity analyses

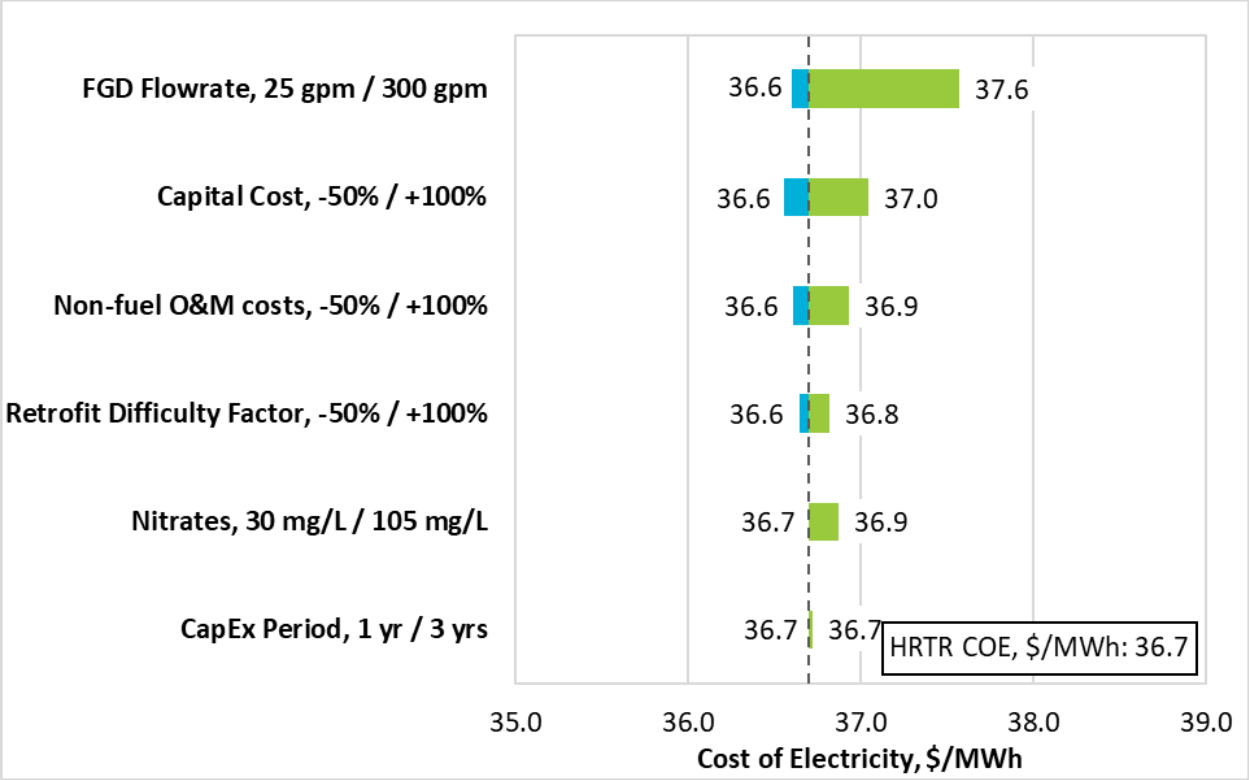


Exhibit ES-4. Advantages (in bold) and disadvantages (in italics) of LRTR and HRTR systems

Low Residence Time Reduction (LRTR)	High Residence Time Reduction (HRTR)
Designated BAT for meeting 2019/2020 ELGs for selenium, nitrate/nitrite	**Former BAT for 2015 ELGs for selenium and nitrate/nitrite, which are more stringent**
Lower capital cost, footprint than high residence time systems	**Demonstrated success with high Se flows, FGD wastewater**
Currently only in mining applications, though FGD applications have been pilot tested and full-scale operations are in progress	*Requires larger reactors in order to meet desired selenium effluent values*
May struggle with high selenium flows based on kinetic analyses, comments from industry leaders	

Operational issues have been discussed for biological treatment systems in general, whether low or high residence time, by industry leaders—increased chloride or total dissolved solids levels can eventually kill a bacteria colony within a biological treatment system, and the treatment system will not handle off-design conditions well. An equalization tank will smooth out sudden spikes, but chronic increases that lead to concentrations that exceed the tolerance of the bacteria colony will lead to failure. Future research into biological treatment systems should seek to quantify the impact of residence time on selenium removal, identify the toxicity

limits of the bacteria colony, as well as to improve the overall robustness of the biological treatment system under off-design conditions.

1 INTRODUCTION

In November 2019, the United States (U.S.) Environmental Protection Agency (EPA) proposed an update to the effluent limitation guidelines (ELGs) and standards for the steam electric power generation point source category, as an update to the prior November 2015 ruling. [1] [2] This new ruling includes separate requirements for high flue gas desulfurization (FGD) flow plants, electric generating units that will cease coal combustion by 2028, and low utilization electric generating units. In particular, a central motivation for this ruling was to reduce the expected costs of the Best Available Technology (BAT) for coal-fired power plants relative to the 2015 ruling, in the wake of a surge in natural gas and renewable technologies and the subsequent suppression of coal-fired and nuclear-powered generation. [2]

In support of this cost reduction, EPA introduced a relaxation in the long-term average discharge limitations for selenium and nitrates/nitrites—these components are removed via biological treatment in the BAT for both the 2015 and 2019 rulings. [1] [2] The increase in the required levels for these contaminants has led to the description of a new BAT—chemical precipitation (CP) followed by low residence time reduction (LRTR) biological treatment. EPA notes that there will be a significant cost savings in using a lower residence time for treatment of selenium and nitrates/nitrites as opposed to the high residence time reduction (HRTR) technology used in the 2015 ruling. [1] However, comments from industry leaders [5] [6] note that there are potential issues involved with the implementation of LRTR systems—namely, that they may not be able to handle high total dissolved solids (TDS) wastewater flows in FGD wastewater, that the residence time may be insufficient for reduction of high selenium flows, and that the samples collected by EPA are not representative of the industry at large. Moreover, they note that the systems implemented in the EPA test sites may not have been designed to meet ELGs for selenium and were not designed for coal-fired power plants. [5]

This study is, therefore, an addition to the original work conducted by the National Energy Technology Laboratory (NETL) on the BAT for FGD wastewater treatment from the 2015 ruling: "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants." [7] The aim of this work is to evaluate the performance and compare the capital and operation and maintenance (O&M) costs published by EPA for LRTR and HRTR systems to the costs prepared by NETL for HRTR systems for the 2015 ruling, as well as to evaluate the concerns raised by industry leaders. [5] Sensitivities to these capital and O&M costs are prepared based on the flow rates of FGD wastewater, the capital costs and O&M costs of the treatment systems themselves, the difficulty of retrofit, the level of nitrates/nitrites present, the capital expenditure (CapEx) period and the equipment type (whether or not labor, installation, contingency, controls costs, and associated costs are included with EPA estimates). In addition, the impact of coal type, stream composition beyond nitrate level, and transient plant operation are qualitatively discussed based on EPA documentation [2] [4], industry comments [5] [6], and vendor discussions. Finally, these costs are combined in a levelized cost of electricity (LCOE) estimate for a pulverized coal (PC) power plant in the midwestern United States, based on the Bituminous Baseline, revision 4 method. [9]

2 DESIGN BASIS

This section discusses the design basis common to all technologies, as well as environmental targets and cost assumptions used in this study. Technology-specific design criteria are covered in subsequent chapters.

2.1 SITE CHARACTERISTICS

All plants in this study are assumed to be located at a generic site in the midwestern United States with site characteristics and ambient conditions as presented in Exhibit 2-1 and Exhibit 2-2. The ambient conditions are the same as International Organization for Standardization (ISO) conditions.

Exhibit 2-1. Site characteristics

Parameter	Value
Location	Midwestern U.S.
Topography	Level
Size, acres	300
Transportation	Rail or Highway
Ash Disposal	Off-Site
Water	50% Municipal and 50% Ground Water

Exhibit 2-2. Site ambient conditions

Parameter	Value
Elevation, ft	0
Barometric Pressure, psia	14.696
Average Ambient Dry Bulb Temperature, °F	59
Average Ambient Wet Bulb Temperature, °F	51.5
Design Ambient Relative Humidity, %	60
Cooling Water Temperature, °F[A]	60
Air composition based on published psychrometric data, mass %	
N_2	75.055
O_2	22.998
Ar	1.280
H_2O	0.616
CO_2	0.050
Total	100.00

[A]The cooling water temperature is the cooling tower cooling water exit temperature, which is set to 8.5°F above ambient wet bulb conditions in ISO cases

The quality of plant makeup water will vary dramatically from source to source (municipal versus ground water), as well as from site to site, and can be expected to vary significantly throughout any given site, particularly if ground water is utilized. In this study, 50 percent of the makeup water to the plants is sourced from a publicly owned treatment works (POTW), with the balance of the makeup water sourced from ground water. The assumed design makeup water composition is provided in Exhibit 2-3.

Exhibit 2-3. Design makeup water quality

Parameter	Ground Water (Range)	POTW Water (Range)	Makeup Water (Design Basis)
pH	6.6 – 7.9	7.1 – 8.0	7.4
Specific Conductance, µS/cm	1,096 – 1,484	1,150 – 1,629	1312
Turbidity, NTU		<50	<50
Total Dissolved Solids, ppm			906
M-Alkalinity as $CaCO_3$, ppm[A]	200 – 325	184 – 596	278
Sodium as Na, ppm	102 – 150	172 – 336	168
Chloride as Cl, ppm	73 – 100	205 – 275	157
Sulfate as SO_4, ppm	100 – 292	73 – 122	153
Calcium as Ca, ppm	106 – 160	71 – 117	106
Magnesium as Mg, ppm	39 – 75	19 – 33	40
Potassium as K, ppm	15 – 41	11 – 21	18
Silica as SiO_2, ppm	5 – 12	21 – 26	16
Nitrate as N, ppm	0.1 – 0.8	18 – 34	12
Total Phosphate as PO_4, ppm	0.1 – 0.2	1.3 – 6.1	1.6
Strontium as Sr, ppm	2.48 – 2.97	0.319 – 0.415	1.5
Fluoride as F, ppm	0.5 – 1.21	0.5 – 0.9	0.8
Boron as B, ppm	0.7 – 0.77		0.37
Iron as Fe, ppm	0.099 – 0.629	0.1	0.249
Barium as Ba, ppm	0.011 – 0.52	0.092 – 0.248	0.169
Aluminum as Al, ppm	0.068 – 0.1	0.1 – 0.107	0.098
Selenium as Se, ppm	0.02 – 0.15	0.0008	0.043
Lead as Pb, ppm	0.002 – 0.1		0.026
Arsenic as As, ppm	0.005 – 0.08		0.023
Copper as Cu, ppm	0.004 – 0.03	0.012 – 0.055	0.018
Nickel as Ni, ppm	0.02 – 0.05		0.018
Manganese as Mn, ppm	0.007 – 0.015	0.005 – 0.016	0.009
Zinc as Zn, ppm	0.005 – 0.024		0.009
Chromium as Cr, ppm	0.01 – 0.02		0.008
Cadmium as Cd, ppm	0.002 – 0.02		0.006
Silver as Ag, ppm	0.002 – 0.02		0.006
Mercury as Hg, ppm	0.0002 – 0.001		3E-04

[A]Alkalinity is reported as $CaCO_3$ equivalent, rather than the concentration of HCO_3; the concentration of HCO_3 can be obtained by dividing the alkalinity by 0.82

2.2 COAL CHARACTERISTICS

The characteristics of the design coal (Illinois No. 6) are presented in Exhibit 2-4 and the associated composition of the coal ash and coal trace elements are provided in Exhibit 2-5 and Exhibit 2-6, respectively. The coal properties are from the April 2016 revision of "Quality Guidelines for Energy System Studies (QGESS): Detailed Coal Specifications." [10]

Exhibit 2-4. Design coal characteristics

Rank	Bituminous	
Seam	**Illinois No. 6**	
Source	Old Ben Mine	
Proximate Analysis (weight %)[A]		
	As Received	**Dry**
Moisture	11.12	0.00
Ash	9.70	10.91
Volatile Matter	34.99	39.37
Fixed Carbon	44.19	49.72
Total	100.00	100.00
Sulfur	2.51	2.82
HHV, kJ/kg (Btu/lb)	27,113 (11,666)	30,506 (13,126)
LHV, kJ/kg (Btu/lb)	26,151 (11,252)	29,544 (12,712)
Ultimate Analysis (weight %)		
	As Received	**Dry**
Moisture	11.12	0.00
Carbon	63.75	71.72
Hydrogen	4.50	5.06
Nitrogen	1.25	1.41
Chlorine	0.15	0.17
Sulfur	2.51	2.82
Ash	9.70	10.91
Oxygen[B]	7.02	7.91
Total	100.00	100.00

[A]The proximate analysis assumes sulfur as volatile matter
[B]By difference

Exhibit 2-5. Coal ash composition

Name	Formula	Concentration, %
Silica	SiO_2	45.0
Aluminum Oxide	Al_2O_3	18.0
Titanium Dioxide	TiO_2	1.0
Iron Oxide	Fe_2O_3	20.0
Calcium Oxide	CaO	7.0
Magnesium Oxide	MgO	1.0
Sodium Oxide	Na_2O	0.6
Potassium Oxide	K_2O	1.9
Phosphorus Pentoxide	P_2O_5	0.2
Undetermined	N/A	1.8

Exhibit 2-6. Coal trace elements composition

Name	Formula	Concentration, ppm (dry basis)
Arsenic	As	7.5
Boron	B	90
Beryllium	Be	1.2
Cadmium	Cd	0.5
Chlorine	Cl	1,671
Cobalt	Co	3.5
Chromium	Cr	14
Copper	Cu	9.2
Fluorine	F	93
Mercury	Hg	0.09
Lithium	Li	9.4
Manganese	Mn	38
Molybdenum	Mo	8.4
Nickel	Ni	14
Phosphorus	P	87
Lead	Pb	24
Tin	Sn	0.9
Selenium	Se	1.9
Thorium	Th	1.5
Uranium	U	2.2
Vanadium	V	31
Zinc	Zn	84.4

2.3 LIMESTONE CHARACTERISTICS

The characteristics of the limestone used in the wet FGD are presented in Exhibit 2-7. The limestone properties are from the April 2016 revision of "QGESS: Specification for Selected Feedstocks." [11]

Exhibit 2-7. Limestone composition

Name	Formula	Concentration
Calcium Carbonate	$CaCO_3$	80.40
Magnesium Carbonate	$MgCO_3$	3.50
Silica	SiO_2	10.32
Aluminum Oxide	Al_2O_3	3.16
Iron Oxide	Fe_2O_3	1.24
Sodium Oxide	Na_2O	0.23
Potassium Oxide	K_2O	0.72
Undetermined	N/A	0.43

2.4 ENVIRONMENTAL TARGETS

2.4.1 Air Emissions

Environmental targets were established for each of the technologies as follows:

- Sulfur dioxide (SO_2), filterable particulate matter (PM), mercury (Hg), and hydrochloric acid (HCl) limits were set according to the emission limits for existing coal-fired units by the April 2016 update to the Utility Mercury and Air Toxics Standards for PC technologies. [12]
- The nitrogen oxides (NOx) emission limit was set equal to the air quality permit for Duke Energy's Cliffside Steam Station. [13]

The regulations differentiate between low rank and non-low rank coal types based on their heating value. Coals with a higher heating value (HHV) of greater than 8,300 Btu/lb on a moist, mineral-matter free basis are considered non-low rank. Therefore, Illinois No. 6 coal, with an HHV (moist, mineral-matter free) of 12,900 Btu/lb, is considered a non-low rank coal.

The air emission limits established as applicable to this study are provided in Exhibit 2-8.

Exhibit 2-8. Emission limits for SO_2, NOx, PM, Hg, and HCl at existing plants [12], [13]

Pollutant	(lb/MWh-gross)
SO_2	1.50
NOx	1.00
PM (Filterable)	0.30
Hg	$1.3x10^{-5}$
HCl	0.02

Other regulations that could affect emissions limits from an existing plant include the New Source Review (NSR) permitting process and Prevention of Significant Deterioration (PSD). The NSR process requires installation of emission control technology, meeting either the Best Available Control Technology (BACT) determinations for sources being in areas meeting ambient air quality standards (attainment areas), or Lowest Achievable Emission Rate (LAER) technology for sources being in areas not meeting ambient air quality standards (non-attainment areas). Environmental area designation varies by county and can be established only for a specific site location. Based on EPA's Green Book Non-attainment Area Map, relatively few areas in the midwestern United States are classified as "non-attainment," so the plant site for this study was assumed to be in an attainment area. [14]

It was assumed that low-NOx burners and staged overfire air would limit NOx production to 0.35 lb/MMBtu and that selective catalytic reduction (SCR) technology would be approximately 66 percent efficient. By adjusting the ammonia flow rate and/or catalyst bed depth in the SCR unit, the NOx emissions limit was able to be met exactly.

To meet the SO_2 emission limit exactly, the wet FGD was assumed to be 96 percent efficient. Current technology allows wet FGD removal efficiencies in excess of 99 percent. However, based on the assumed existing source air emission limits, such high removal efficiency is not necessary.

While fabric filters are commonly capable of achieving removal efficiencies of greater than 99.9 percent, the efficiency was varied to meet the PM emission limit exactly, which required a removal rate of 99.5 percent.

It was assumed that the total mercury removal rate resulting from the combination of pollution control technologies used (SCR, dry sorbent injection [DSI], activated carbon injection [ACI], fabric filters, and wet FGD) would meet the applicable limit. DSI is required to limit the effects of sulfur trioxide (SO_3) on Hg capture due to the high sulfur content of the coal in this study.

2.4.2 Water Emissions

The ELGs are national technology-based standards derived from data collected from industry. [1] They are intended to provide flexibility in implementation through use of technologies already installed and operating in the power industry. The federal standards established are the minimum discharge standards. As the guidelines are enforced under the National Pollutant

Discharge Elimination System (NPDES) [15], more stringent water quality-based standards may be established by the local permitting authority. These additional requirements were not considered in this assessment.

Intermittent discharges (e.g., chemical metal cleaning wastewater), coal pile runoff, low volume waste (e.g., boiler blowdown), and cooling tower blowdown were assumed to be compliant with all applicable regulations with no additional treatment beyond conventional considerations. Exhibit 2-9 provides the technologies that served as the basis for EPA's determination of the effluent discharge limits in the 2015 ruling—these have been changed in the 2019 ruling for specific subcategories in Exhibit 2-10. For existing sources with direct discharge—the focus of this study—the final rule establishes effluent limitations based on BAT.

Exhibit 2-9. ELG 2015 Final Rule: steam electric regulatory options

Waste Stream	Technology basis for the main BAT/NSPS/PSES/PSNS regulatory options					
	A	B	C	D	E	F
FGD Wastewater	CP	CP + Biological Treatment	CP + Biological Treatment	CP + Biological Treatment	CP + Biological Treatment	Evaporation
Fly Ash Transport Water	Dry Handling	Dry Handling	Dry Handling	Dry Handling	Dry Handling	Dry Handling
Bottom Ash Transport Water	Impoundment	Impoundment	Dry Handling/ Closed Loop (for units >400 MW); Impoundment (for units ≤ 400 MW)	Dry Handling/ Closed Loop	Dry Handling/ Closed Loop	Dry Handling/ Closed Loop
Flue Gas Mercury Control Wastewater	Dry Handling	Dry Handling	Dry Handling	Dry Handling	Dry Handling	Dry Handling
Gasification Wastewater	Evaporation	Evaporation	Evaporation	Evaporation	Evaporation	Evaporation
Combustion Residual Leachate	Impoundment	Impoundment	Impoundment	Impoundment	CP	CP

Exhibit 2-10. ELG 2019 Proposed Rule: Steam Electric Regulatory Options

Waste Stream	Subcategory	Technology basis for the main BAT/PSES regulatory options			
		1	2	3	4
FGD Wastewater	N/A	CP	CP + LRTR Biological Treatment	CP + LRTR Biological Treatment	Membrane Filtration
	High Flow Facilities	NS[A]	CP	CP	CP
	Low Utilization Boilers	NS[A]	CP	NS	NS
	Boilers Retiring by 2028	Impoundment	Impoundment	Impoundment	Impoundment
	Voluntary Incentive Program (direct dischargers only)	Membrane Filtration	Membrane Filtration	Membrane Filtration	N/A
Bottom Ash Transport Water	N/A	Dry Handling/ High Recycle Rate Systems	Dry Handling/ High Recycle Rate Systems	Dry Handling/ High Recycle Rate Systems	Dry Handling/ High Recycle Rate Systems
	Low Utilization Boilers	NS	Impoundment + Best Management Practices	NS	NS
	Boilers Retiring by 2028	Impoundment	Impoundment	Impoundment	Impoundment

[A] Not subcategorized

In 2015, EPA decided to establish BAT effluent limitations based on certain technologies (described as Option D in Exhibit 2-9). After considering the technologies available to treat and discharge, EPA concluded that (existing source) BAT for wet FGD wastewater is CP plus biological treatment. The new proposed ruling suggested an additional four options (shown in Exhibit 2-10). Option 1 is specific to mercury and arsenic limitations only. Options 2 and 3 include selenium and nitrate/nitrite; Option 2 subcategorizes boilers producing less than 876,000 MWh per year. Option 4 is the most stringent, including bromide and TDS. The proposed BAT falls under Option 2 in Exhibit 2-10. The proposed applicable wet FGD wastewater discharge limits for an existing source are shown in Exhibit 2-11, provided they do not fall under high flow or low utilization subcategories.

Exhibit 2-11. Proposed discharge limits for wet FGD wastewater from existing sources [2]

Effluent Characteristic	Long-Term Average	Daily Maximum Limit	Monthly Average Limit[A]
Arsenic, ppb	5.1	18	9
Mercury, ppt	13.5	85	31
Selenium, ppb	16.6	76	31
Nitrate/nitrite as N, ppm	2.6	4.6	3.2

[A]Monthly Average Limit refers to the highest allowable average of daily discharges over 30 consecutive days

For the FGD wastewater treatment systems, the limits are applied at the discharge, prior to mixing with other plant water systems. As compared to 2015 limits [1], the arsenic and mercury limits have been decreased while selenium and nitrate limits have been increased.

Although the existing source limits are based on BAT, a plant is not required to use BAT as the control technology to comply. The technologies available to achieve compliance for existing plants have been discussed in further detail in the NETL report, "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants" [7] and are not reiterated here, save for the comparisons in BAT between 2015 and 2019 rulings. [1] [2]

2.5 Existing Plant Basis

For the purposes of this study, the existing fleet is represented by a modified version of the non-capture, subcritical PC plant from the Bituminous Baseline. [9] The modifications consisted of removing the spray dryer evaporator to allow the wet FGD wastewater to be discharged to surface water and updating the air and water emission standards to better reflect emission limits for existing PC power plants. This plant is the reference case (shown in Exhibit 2-12) from the NETL study titled "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants." [7]

Exhibit 2-12. Summary description of Case 0 [7]

General Characteristics	
Net Capacity, MWe	650
Steam Cycle	Subcritical
Steam Conditions, psig/°F/°F	2,400/1,050/1,050
Capacity Factor, %	85
Wastewater characteristics	
FGD Wastewater Flow Rate, gpm	57
Wet FGD Chloride Concentration, ppmw	20,000
As Concentration, ppb	1,400
Hg Concentration, ppb	700
Se Concentration, ppb	5,000
Nitrate/Nitrite Concentration, ppb	30,000

The steam generator for Case 0 is a drum-type, wall-fired, balanced draft, natural circulation, totally enclosed dry bottom furnace, with superheater, reheater, economizer, and air preheater. It is assumed for the purposes of this study that the power plant is designed for operation as a base-load unit.

While the capacity factor assumed for this study is achievable for a subcritical PC plant, the average capacity factor for a plant in the existing coal fleet was 52.0 percent in 2019. [16] It was assumed that plants with relatively high capacity factors, compared to the fleet average, would be more likely to undertake a retrofit to achieve ELG compliance. Therefore, the capacity factor was assumed to be 85 percent in all cases.

It is assumed that the capacity factor would not be negatively impacted by the addition of a wet FGD wastewater treatment system.

2.5.1 Process Water Sources[a]

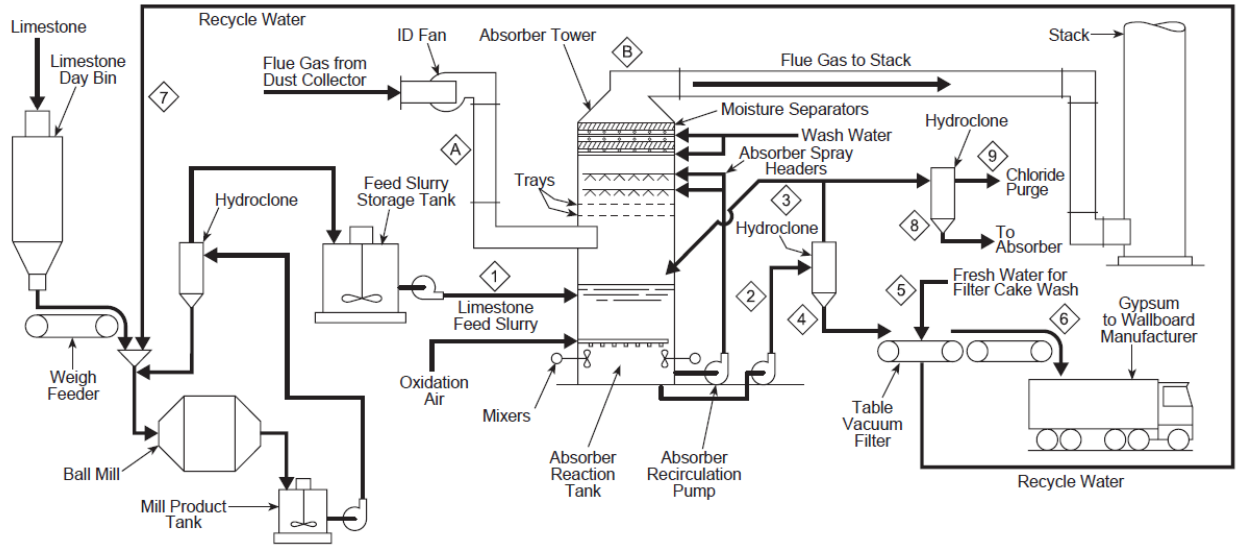
As discussed in Section 2.4.2, the only system in this study that produces a wastewater stream that must be treated for ELG compliance is the wet FGD (combustion residual leachate is assumed to be outside the scope of this study). A detailed process description of the wet FGD is provided in the following subsection.

2.5.1.1 Wet Flue Gas Desulfurization

Exhibit 2-13 provides a diagram of a wet limestone, forced oxidation, positive pressure, non-reheat absorber unit, with wet-stack and gypsum production. The function of the wet FGD system is to scrub the boiler exhaust gases to remove SO_2 prior to release to the environment. The wet FGD's sulfur removal efficiency is 96 percent in all cases. The FGD system is assumed to operate at a scrubber liquor chloride level of 20,000 ppm.

[a] This section taken from the NETL report, "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants" [7]

Exhibit 2-13. Example diagram of a wet FGD unit



Used with permission from The Babcock and Wilcox Company [17]

2.5.1.1.1 FGD Scrubber

Flue gas enters the absorber and bubbles through the tray, which provides enhanced contact between gas and reagent. Multiple spray elevations with header piping and nozzles maintain a consistent reagent concentration in the spray zone. Continuing upward, the scrubbed flue gas passes through several levels of moisture separators. These consist of chevron-shaped vanes that direct the gas flow through several abrupt changes in direction, separating the entrained droplets of liquid by inertial effects. The scrubbed flue gas exits at the top of the absorber vessel and is routed to the plant stack.

The scrubbing slurry falls to the lower portion of the absorber vessel, which contains a large inventory of liquid. Oxidation air is added to promote the oxidation of calcium sulfite contained in the slurry to calcium sulfate (gypsum). Multiple agitators operate continuously to prevent settling of solids and enhance mixture of the oxidation air and the slurry. Recirculation pumps recirculate the slurry from the lower portion of the absorber vessel to the spray level. Spare recirculation pumps are provided to ensure availability of the absorber.

The absorber chemical equilibrium is maintained by continuous makeup of fresh reagent, and blowdown of byproduct solids via the bleed pumps. A spare bleed pump is provided to ensure availability of the absorber. The byproduct solids are routed to the byproduct dewatering system. The circulating slurry is monitored for pH and density.

2.5.1.1.2 Byproduct Dewatering

The function of the byproduct dewatering system is to dewater the bleed slurry from the FGD absorber module. The dewatering system selected for this plant is intended to produce wallboard-grade gypsum. The scope of the system is from the bleed pump discharge connections to the gypsum storage pile.

Gypsum (calcium sulfate) is produced by the reaction of calcium sulfite with limestone reagent in the absorber tower recycle tank. The bleed from the absorber contains approximately 20 weight percent (wt%) gypsum. The absorber slurry is pumped by an absorber bleed pump to a primary dewatering hydrocyclone cluster. The primary hydrocyclone performs two process functions. The first function is to dewater the slurry from 20 wt% to 50 wt% solids. The second function is to perform a $CaCO_3$ and $CaSO_4 \bullet 2H_2O$ separation. This process ensures a limestone stoichiometry in the absorber vessel of 1.10 and an overall limestone stoichiometry of 1.05. This system reduces the overall operating cost of the FGD system. The underflow from the hydrocyclone flows into the filter feed tank, from which it is pumped to a horizontal belt vacuum filter. Two 100 percent filter systems are provided for redundant capacity.

2.5.1.1.2.1 Hydrocyclones

The hydrocyclone is a simple and reliable device (no moving parts) designed to increase the slurry concentration in one step to approximately 50 wt%. This high slurry concentration is necessary to optimize operation of the vacuum belt filter.

The hydrocyclone feed enters tangentially and experiences centrifugal motion so that the heavy particles move toward the wall and flow out the bottom. Some of the lighter particles collect at the center of the cyclone and flow out the top. The underflow is thus concentrated from 20 wt% at the feed to 50 wt%.

Multiple hydrocyclones are used to process the bleed stream from the absorber. The hydrocyclones are configured in a cluster with a common feed header. The system has two hydrocyclone clusters, each with five six-inch diameter units. Four cyclones are used to continuously process the bleed stream at design conditions, and one cyclone is spare.

Cyclone overflow and underflow are collected in separate launders. The overflow from the primary hydrocyclones still contains about 5 wt% solids, consisting of gypsum, fly ash, and limestone residues; a portion is sent back to the absorber. The remainder of the overflow is fed to a secondary hydrocyclone, where the resulting underflow is returned to the absorber and the overflow is blown down to the process water treatment system, for chloride control. The flow to the secondary hydrocyclones is controlled to maintain a chloride concentration of 20,000 ppmw in the blowdown.

The underflow of the primary hydrocyclones flows into the filter feed tank from where it is pumped to the horizontal belt vacuum filters.

2.5.1.1.2.2 Horizontal Vacuum Belt Filters

The secondary dewatering system consists of horizontal vacuum belt filters. The pre-concentrated gypsum slurry (50 wt%) is pumped to an overflow pan through which the slurry flows onto the vacuum belt. As the vacuum is pulled, a layer of cake is formed. The cake is dewatered to approximately 90 wt% solids as the belt travels to the discharge. At the discharge end of the filter, the filter cloth is turned over a roller where the solids are dislodged from the filter cloth. This cake falls through a chute onto the pile prior to the final byproduct uses. The required vacuum is provided by a vacuum pump. The filtrate is collected in a filtrate tank that

provides surge volume for use of the filtrate in grinding the limestone. Filtrate that is not used for limestone slurry preparation is returned to the absorber.

2.5.1.1.3 FGD Wastewater Quality

The design wastewater composition for the wet FGD blowdown is provided in Exhibit 2-14. The design water quality is based on a survey of plants burning bituminous, high-sulfur coal [18] [19] and on internal information from Black & Veatch projects. Exhibit 2-14 includes a range of values, an average, and the design basis.

Exhibit 2-14. Design basis FGD wastewater quality

Parameter	Range	Average	Design Basis
pH	5.5 – 7.4	6.6	7.2
Chemical Oxygen Demand, ppm	304 – 1,060	682	350
Biological Oxygen Demand, ppm	21 – 1,370	422	500
Specific Conductance, µS/cm	5,990 – 32,000	9,595	32,000
Ammonia as N, ppm	1.5 – 31.5	8.4	10
Suspended Solids, ppm	4,970 – 25,300	13,888	15,000
Total Dissolved Solids, ppm	4,740 – 44,600	21,310	43,494
Chloride as Cl, ppm	832 – 28,800	9,966	20,000
Sulfate as SO_4, ppm	1,290 – 11,900	4,212	7,600
Calcium as Ca, ppm	751 – 5,370	2,791	5,370
Magnesium as Mg, ppm	176 – 7,000	2,728	6,000
Sodium as Na, ppm	59 – 5,340	998	2900
Boron (total), ppm	3.0 – 626	220	430
Potassium as K, ppm	35 – 684	226	250
M-Alkalinity as $CaCO_3$, ppm[A]	131 – 625	275	200
Iron (total), ppm	3.4 – 824	200	290
Aluminum (total), ppm	1.0 – 289	93	150
Silica as SiO_2, ppm	1 – 91	33	100
Manganese (total), ppm	1.58 – 225	32.1	60
Nitrate/Nitrite as N, ppm	1.0 – 54.5	20.5	30
Total Kjeldahl Nitrogen, ppm	6.2 – 51.6	19.2	20
Carbon, ppm			8
Phosphorus, ppm	0.05 – 10.5	4.61	7
Nickel (total), ppm	0.447 – 6.0	2.05	5
Selenium (total), ppm	0.651 – 8.66	2.75	5

Parameter	Range	Average	Design Basis
Zinc (total), ppm	0.31 – 9.04	3.23	6
Barium (total), ppm	0.588 – 11.900	3.330	5
Titanium (total), ppm	0.377 – 8.18	2.57	4
Vanadium (total), ppm	0.078 – 1.58	0.67	1.3
Fluorine, ppm			1
Arsenic (total), ppm	0.0599 – 3.000	0.799	1.4
Copper (total), ppm	0.0376 – 2.130	0.788	1.4
Lead (total), ppm	0.0312 – 4.000	0.896	1.3
Molybdenum (total), ppm	0.065 – 1.340	0.59	0.9
Mercury (total), ppm	0.0164 – 1.070	0.255	0.7
Chromium, ppm	0.176 – 1.380	0.777	1
Cobalt, ppm			0.1
Lithium, ppm			0.1
Beryllium (total), ppm	0.0036 – 3.000	0.438	0.140
Cadmium (total), ppm	0.00484 – 0.238	0.0728	0.140
Thallium (total), ppm	0.00633 – 0.300	0.0864	0.140
Antimony (total), ppm	0.00923 – 0.0518	0.0269	0.040
Uranium, ppm			0.03
Thorium, ppm			0.02
Tin, ppm			0.01

[A]Alkalinity is reported as $CaCO_3$ equivalent, rather than the concentration of HCO_3; the concentration of HCO_3 can be obtained by dividing the alkalinity by 0.82

2.6 STUDY ASSUMPTIONS

In addition to those noted above, the following assumptions were also made:

- There are no impacts to plant emissions from installing any of the wet FGD wastewater treatment configurations assessed in this study.
- The wet FGD, SCR, and fabric filter of an existing plant would be less efficient and, therefore, achieve lower removal rates for equivalent energy demands to those of the non-capture, subcritical PC Baseline plant. Therefore, no adjustments to the performance or operation were made beyond reducing the air quality control equipment's removal efficiencies.
- No specific considerations were made for deconstruction, modification, disposal, or other specific costs associated with retrofitting an existing plant with wet FGD wastewater treatment processes.

- All capital costs of the existing plant are paid off, such that capital costs reflect only the cost of the wastewater treatment system and plant upgrades required to accommodate the retrofit.

2.7 Cost Estimating Methodology

In this study, it was assumed that all capital costs of the existing plant were paid off, such that capital costs reflect only the cost of the wastewater treatment system and plant upgrades required to accommodate the retrofit. The reference subcritical PC plant is utilized as a reference for all costs not associated with the wastewater treatment system. [9] The methodology used in developing the costs for the non-capture, subcritical PC plant is consistent with "QGESS: Cost Estimation Methodology for NETL Assessments of Power Plant Performance." [20]

EPA estimates [4] provide cost curves for CP and LRTR and HRTR systems depending on whether or not transport and disposal are handled on- or off-site, as well as the level of influent nitrates to the FGD treatment system. These costs are compared with the NETL FGD study, "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants" for the HRTR case (Case 1). [7] Additional factors are applied to the cost estimates provided by EPA based on comments left by industry leaders [5] on the proposed ruling—these are additionally discussed in the Appendix: EPA Cost Curves and Implementation.

2.7.1 Retrofit Difficulty Factors

Retrofit difficulty factors were applied to the equivalent capital and labor costs at the sub-account level according to "QGESS: Estimating Plant Costs Using Retrofit Difficulty Factors." [21] Equipment retrofit factors varied 5–10 percent and labor retrofit factors varied 15–30 percent; the retrofit factors applied at the sub-account level in each case are available in Exhibit 2-15.

Exhibit 2-15. Retrofit difficulty factors

Sub-Account[A]	Description	Retrofit Difficulty Factor, %[B]	
		Equipment	Direct Labor
3.8	Wet FGD Wastewater Treatment Processes	5	15
11.2	Station Service Equipment	5	15
11.3	Switchgear & Motor Control	5	15
11.4	Conduit & Cable Tray	10	15
11.5	Wire & Cable	10	15
11.6	Protective Equipment	5	15
12.5	Signal Processing Equipment	0	0
12.6	Control Boards, Panels & Racks	5	15
12.7	Distributed Control System Equipment	5	30
12.8	Instrument Wiring & Tubing	5	20
12.9	Other Instrumentation & Controls Equipment	5	20

Sub-Account[A]	Description	Retrofit Difficulty Factor, %[B]	
		Equipment	Direct Labor
13.1	Site Preparation	5	20
13.2	Site Improvements	5	20
13.3	Site Facilities	5	20

[A]Sub-accounts correspond to the capital cost code of accounts provided in the NETL FGD study on the costs of wastewater treatment [7]
[B]Retrofit costs are calculated by multiplying the estimated equipment and direct labor greenfield costs by 1 plus the associated retrofit difficulty factor for each sub-account

Exhibit 2-15 represents all sub-accounts impacted by the installation of a wet FGD wastewater treatment system. Since there is no specific guidance in the retrofit QGESS documentation for wastewater treatment systems, the retrofit difficulty factors applied to this sub-account were assumed to be the same as the retrofit difficulty factors applied to a CO_2 removal system. For the EPA estimates themselves (applied to sub-account 3.8), it was assumed that the retrofit difficulty factor was implicit in the cost estimates, as the ease of retrofit was a minor factor in the choice for BAT—therefore, the default values provided by the cost curves are referenced to 5 percent for equipment and 15 percent for direct labor.

2.7.2 Financial Parameters

Typical values assumed for investor-owned utility (IOU) power plant retrofit projects implementing commercially available technologies are listed in Exhibit 2-16, along with the estimated fixed charge rates and total as-spent cost (TASC)/total overnight cost (TOC) ratios used to calculate the LCOE according to the Bituminous Baseline, revision 4 method. [9] The fixed charge rate (FCR) is calculated for the 30-year life of the plant with 20 years of depreciation.

Exhibit 2-16. Financial parameters for retrofit projects

Parameter	Values
Financial Structure Designation	IOU Conventional Financing
Tax Rate	26% Effective (21% Federal, 6% State)
Capital Cost Escalation During Capital Expenditure Period	3.0%
Escalation of Revenue, O&M, and Fuel Costs	3%
Debt Interest Rate, %	5%
Debt % of Total	55%
Equity % of Total	45%
Required Return on Equity	10%
Capital Expenditure Period	1 Year
Fixed Charge Rate	0.0707
Total as-Spent Cost/Total Overnight Cost	1.051

2.8 CHEMICAL PRECIPITATION

All wet FGD wastewater treatment processes evaluated in this study require pretreatment with a CP process. Mercury and arsenic removal are achieved in treat and discharge configurations by the addition of organosulfide, which converts soluble forms of mercury and arsenic to insoluble sulfide precipitates, resulting in high removal rates (over 99 percent).

For this study, it is assumed that mercury and arsenic are reduced through the following reactions:

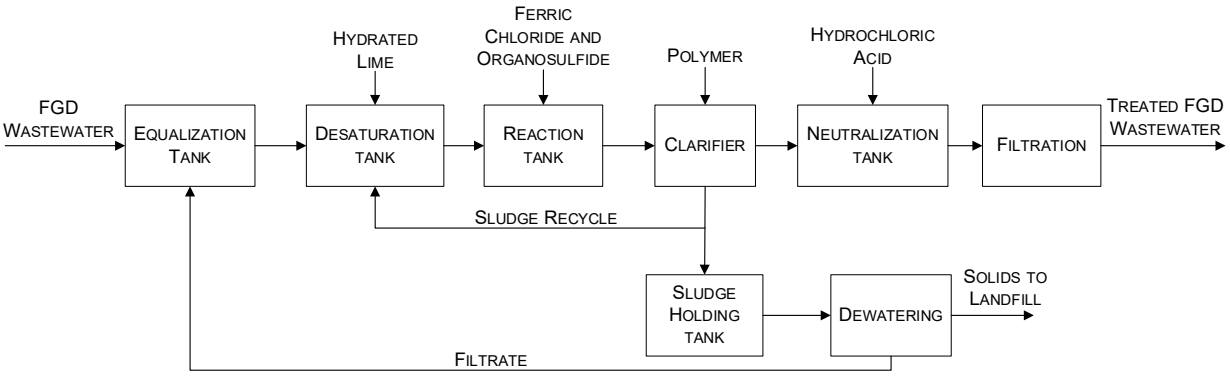
$$2As^{3+} + 3S^{2-} \xrightarrow{yields} As_2S_3$$

$$Hg^{2+} + S^{2-} \xrightarrow{yields} HgS$$

There may be other reactions involving mercury and arsenic within the CP process, but the considerations of this study were limited to the two provided above. The feed rate is controlled so that organosulfide is present in significant excess, ensuring the desired removal rates are consistently achieved.

As shown in Exhibit 2-17, the wet FGD wastewater is pumped to a continuously mixed equalization tank, which buffers the treatment system against variations in wastewater quality and flow. The wastewater flows to a desaturation tank where lime is added in order to precipitate gypsum, which decreases the potential for scaling from sulfates in downstream equipment and incidentally increases the pH to about 8.5. [22] One continuously mixed reaction tank is located downstream of the desaturation tank to allow for addition of ferric chloride (a coagulant) and organosulfide, which is sized to allow sufficient reaction time for the CP reactions to occur. [22] [23] The reaction tank feeds a clarifier where polymer is added to increase the particle size of the insoluble particles to allow settling within the clarifier. Suspended solids from the wet FGD system such as gypsum, inerts, unreacted limestone, heavy metals, and fly ash are removed with traditional clarification techniques. Settled solids from clarification are directed to dewatering equipment.

Exhibit 2-17. CP process configuration



The pH of the clarified water is adjusted with hydrochloric acid to approximately 6.5 to prevent scaling in the downstream filters used to polish the clarifier effluent to achieve low residual carbonate solids. Filtration can reduce effluent suspended solids to <10 ppm and significantly

reduce the concentration of heavy metals such as mercury. The most common type of filter used for wet FGD wastewater treatment is a continuously backwashed sand filter, which has a continuous backwash stream recycled to the equalization tank.

The sludge produced in the clarifying process contains between 10 and 15 wt% solids and will be dewatered to approximately 50 wt% solids using plate and frame filter presses prior to being sent to the landfill for disposal. The filtrate is collected and recycled back to the inlet of the CP process. A portion of the sludge is recycled to the desaturation tank at a rate of 5–10 percent of the FGD blowdown flow rate.

There is typically up to 12 hours of residence time in the equalization tank and a few additional hours of residence time through the reaction tanks, clarifiers, and filters. As a result, the effluent from the CP system will be close to ambient temperature and pressure conditions.

2.8.1 Limitations and Additional Considerations

The primary concern regarding treating wet FGD wastewater to comply with existing source ELG standards is the removal of mercury, arsenic, and selenium, which can exist in either soluble or insoluble forms. These metals are normally precipitated as hydroxides, carbonates, sulfides, or some combination of these salts according to the following general reactions:

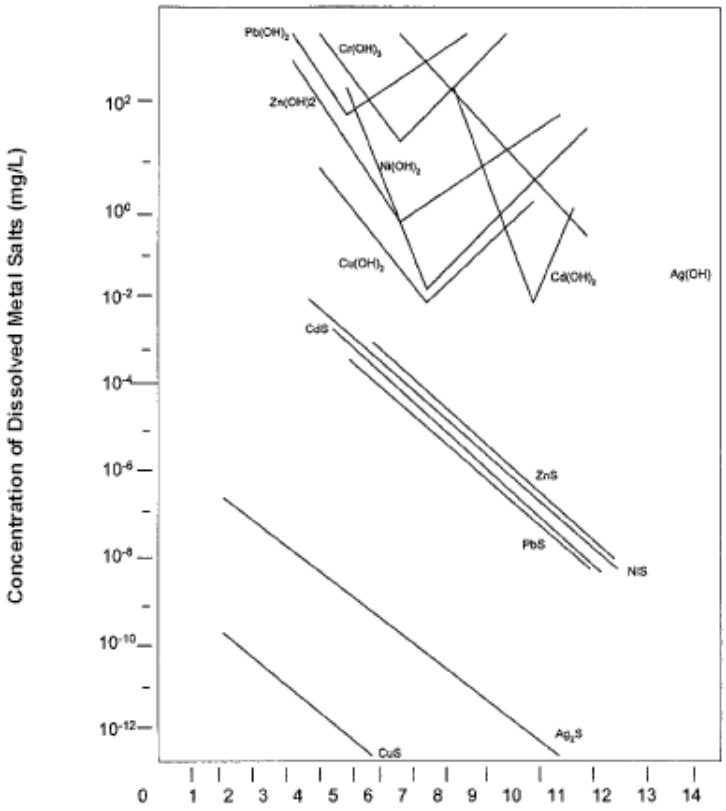
$$M^{+n} + nOH^{-} \rightarrow M(OH)_n$$

$$M^{+2} + S^{-2} \rightarrow MS$$

Co-precipitation of metal salts can enhance the performance of the precipitation process and reduce some heavy metals below their theoretical solubility limits by the adsorption of metals to precipitated metal salts. Generally, this is done with an iron salt and, thus, the iron co-precipitation process. Additional filtration of suspended solids following the clarification step is required to meet very low metals removal requirements or to address concerns related to clarifier upsets.

Many metal hydroxides are amphoteric, meaning their solubility reaches a minimum at a specific pH. This pH value is different for each metal hydroxide, which can make simultaneous removal of multiple metals more difficult and possibly require multiple reaction steps for each constituent being removed or coordination of the solubility of specific metals to the performance needed. Metal sulfides generally do not have this amphoteric concern, making sulfide precipitation advantageous. Exhibit 2-18 is a general summary of theoretical solubility for various metal hydroxides and metal sulfides as a function of pH. [24] As can be seen, metal sulfide solubilities are much lower, making sulfide precipitation the preferred process when extremely low limits are required.

Exhibit 2-18. Solubility of metal hydroxides as a function of pH



Source: EPA [24]

While the design of the CP process in this study reduces suspended solids and heavy metals (including mercury and arsenic) at or below existing source ELG limits, additional treatment downstream of the CP process is required to meet selenium and nitrate/nitrite standards. [19] The biological treatment process (whether LRTR or HRTR) is necessary because while selenite (Se^{+4}) can typically be removed from wastewater through CP, selenate (Se^{+6}) is more soluble and requires reduction to a less soluble form for removal. Selenium in wet FGD wastewater typically exists in both forms and the concentration of each depends on plant operation and the type of coal being combusted.

2.9 BIOLOGICAL TREATMENT

The purpose of biological treatment is to reduce nitrate/nitrite and selenium, which are not removed by the CP system described in Section 2.8. Anoxic/anaerobic biological treatment is an industry-proven technology for selenium and nitrate/nitrite removal from wet FGD wastewater. Biological treatment involves the growth of two types of naturally occurring microorganisms: denitrifiers and selenium-reducing bacteria. A food source (nutrient) is oxidized by both microorganisms, which, in turn, reduces the different oxyanions. The denitrifiers reduce nitrate/nitrite to nitrogenous gas and water, and the selenium-reducing bacteria reduce both selenate and selenite and precipitate solid elemental selenium. The order of oxyanions favored

in biological reaction kinetics is dissolved oxygen, followed by nitrate and selenate. Biological treatment typically consists of a series of reaction vessels to accomplish the desired removals. A controlled environment is required for the proper reactions to occur. The denitrification process requires an anoxic (absent of dissolved oxygen) environment, and the selenium is then reduced in an anaerobic environment (absent of oxygen and nitrate/nitrite).

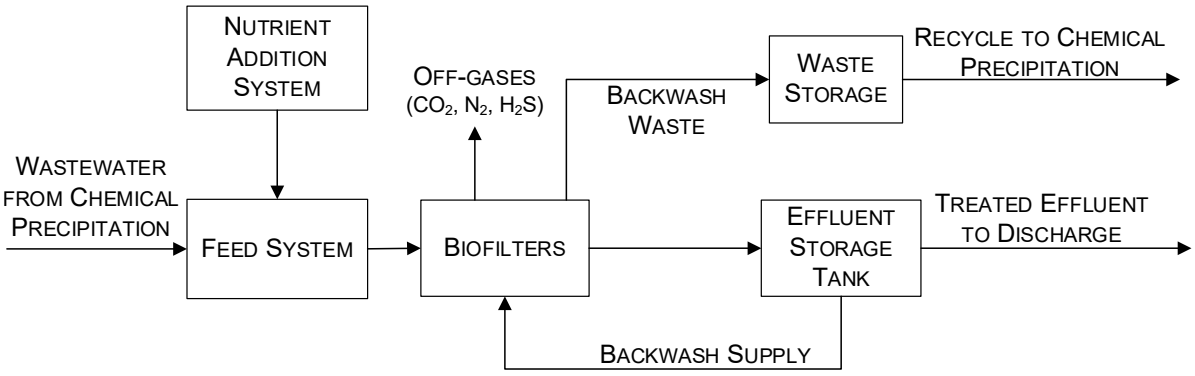
Biomass and elemental selenium are either backwashed from the reactors or removed utilizing downstream filtration. Ultrafiltration is typically used for downstream filtration in order to increase removal efficiency. The periodic backwash waste stream from either the reactors or filters requires further thickening and dewatering prior to disposal. This is accomplished by sending the backwash to the equalization tank of the upstream pretreatment system.

The two types of biological reactors are attached growth and suspended growth (activated sludge type) systems. Attached growth is composed of a biofilm, or layer of microorganisms, that grows on the surface of a solid phase media such as rock, granular activated carbon, sand, or other substrate. Attached growth technologies include Suez's ABMet® process, Envirogen Technologies' fluidized bed reactor system and Frontier's SeHAWK® process that have been developed specifically for removal of selenium and other oxyanions from wastewaters. Suspended growth systems are not being used to treat wet FGD wastewater and were not further considered in this study.

Both biological treatment systems selected for consideration in this study consist of a series of fixed film biofilters in a packed bed reactor design and dual-stage arrangement. Each packed bed reactor contains a bed of granular activated carbon, on which specifically selected microbes grow, develop a biofilm, and are retained.

As shown in Exhibit 2-19, wet FGD wastewater from the CP pretreatment system is fed to the first stage storage tank and is pumped to the first stage of biofilters. Wastewater flows downward through the biofilters to a second stage feed tank by gravity. Wastewater from the tank is pumped to the second stage of biofilters for additional treatment and collected in an effluent tank.

Exhibit 2-19. Biological treatment process configuration



Wet FGD wastewater entering the system will typically have a positive redox level.[b] As the wastewater passes downward through the biofilter, the redox potential gradually decreases as a result of microbial activity. A nutrient (carbon source) solution is added to the system to control the degree of redox gradient of each biofilter within the range that is ideal for nitrate/nitrite and selenium removal. Within the first stage packed bed reactor, nitrate and nitrite are reduced to nitrogen gas (denitrification) followed by the second stage packed bed where selenate and selenite are reduced to elemental selenium. If a membrane bioreactor is included for nitrate/nitrite reduction, the first stage packed bed reactor of the biological treatment process is not necessary.

Entrained solids and gases need to be periodically removed with a backwash process, which may occur 1–4 times per month. To backwash, treated effluent is used as a counter-flow wash to remove entrained solids and gases from the biofilter substrate. Backwash wastewater is allowed to degas and is then recycled to the equalization tank of the CP treatment system where the solids are settled in the clarifier and dewatered with the pretreatment sludge. Waste testing of existing installations has shown that the solids produced can be disposed of as non-hazardous waste per the Toxicity Characteristic Leaching Procedure test.

2.9.1 Limitations and Additional Considerations

There are multiple full-scale HRTR biological treatment installations operating on wet FGD wastewater in the United States—the scale of implementation of LRTR is currently unknown.[c] Because wet FGD wastewater has been tested significantly with biological systems, some equipment suppliers can guarantee a system without the need for pilot testing, although it is still recommended as every wastewater has different characteristics that could cause specific operational challenges.

Data for HRTR biological systems is largely limited to wet FGD wastewaters from plants burning bituminous coals [25, 26], and is entirely missing for LRTR in terms of coal type. Variation in coal type (among various other factors) may lead to changes in the FGD wastewater quality, including the concentration of chlorides, nitrate/nitrite, and other ELG-regulated constituents. The impact of other coal types on wet FGD wastewater quality and downstream wastewater treatment equipment has not been quantified. Additionally, some wet FGD wastewater may contain compounds inhibitory or toxic to microorganisms present in biological systems; the identity and control of these compounds requires further investigation. [25]

Biological reactor sizing is based on a required hydraulic residence time—the average length of time that the wastewater remains in the bioreactor—which varies with each specific vendor's design and is based on the flow rate and water quality of the wastewater. The concentration of nitrate/nitrite and selenium are particularly impactful on the reactor size—this is discussed further in Section 2.9.2 and Section 3.1.

[b] Redox level refers to the oxidation-reduction potential (ORP) of the wastewater. As the ORP reaches 0, denitrification occurs, and as ORP decreases further into the negative range, selenate and/or selenite is reduced to an elemental state.

[c] HRTR Installations include Duke Allen Station, Duke Belews Creek, Duke Roxboro, AEP Mountaineer, AEP Flint Creek, Duke Crystal River, and Duke Marshall Stations.

The reduction and subsequent removal of nitrate, nitrite, selenate, and selenite depends on the oxidation reduction potential (ORP) of the system, which is adjusted by the amount of nutrient fed to the process. At higher ORPs, nitrate and nitrite reduction occurs. As the ORP decreases, selenate and selenite reduction begins. However, at lower ORPs, sulfide production begins to occur. Therefore, ORP must be closely monitored and maintained at desired levels throughout the process.

Removing selenium and nitrate/nitrite using biological systems relies on microorganisms that are sensitive to co-contaminants in wet FGD wastewater, most notably chlorides. For example, permissible chloride levels for some commercial systems may be less than 25,000 mg/L; [26] however, wet FGD wastewater can reach chloride levels of up to 40,000 mg/L. [22]

The design basis of this study is a chloride concentration of 20,000 ppm, which is near the limit of the biological system—this is true for LRTR [5] and HRTR [7] systems. Treatment of wet FGD wastewater to comply with existing source ELG standards at higher chloride concentrations hasn't been well established and it's likely that wet FGD wastewater above this concentration will need to be diluted.

In addition to a chloride limit, at very high nitrate/nitrite concentrations, it is more efficient to denitrify the wastewater with a separate technology, prior to feeding the biological system. Typically, when nitrate/nitrite concentrations in wastewater are approximately 80–100 ppm or above, a membrane bioreactor will be used for denitrification after the CP process and upstream of the biological treatment process. [4]

Inconsistent selenium removal has been documented at several facilities, making consistent ELG compliance difficult to achieve without adding an additional polishing step. [25] Installing polishing technologies (e.g., oxide sorption or [22] ion exchange [27] systems) further increases the costs and complexity of compliance.

Biological reactors require a greater level of communication between the power block and wastewater treatment facilities. Increasing or decreasing load and changes in fuel need to be communicated in advance, and multiple changes to firing conditions should not be made simultaneously. Biological reactors also require several weeks from seeding to being fully operational. They do not operate well at low loads. However, short periods of up to a month of outage or very low load can be accommodated by recycling water and adding nutrient without adverse effects to biological activity. Training needs to be provided to ensure proper operation of the facility.

2.9.2 Residence Time Discrepancies

A central motivator for the new EPA proposed ruling is to reduce costs via the reduction of residence time in biological treatment systems. These LRTR systems (such as Frontier's SeHAWK process and Envirogen's fluidized bed process) are expected by EPA to reach selenium and nitrate/nitrite constraints in a much shorter residence time than HRTR systems—a typical residence time for LRTR systems is around 1–4 hours, versus 10–16 hours for an HRTR system. An LRTR system is designed with two stages like an HRTR system; however, the systems are in an upflow-downflow configuration (as opposed to two downflow stages for an HRTR process).

Vendor conversations have noted that a downstream ultrafiltration module is necessary to prevent the backwashing of ELG contaminants—mercury, specifically. [4] Discussions with Suez suggests that the stark decrease in mercury limitations between the 2015 and 2019 ELG ruling requires an ultrafiltration module for HRTR systems as well; however, this cost is not factored into initial EPA estimates on HRTR systems.

Conversations with LRTR vendors suggest that requirements for selenium reduction set the required residence time for the biological treatment process—with a roughly twofold increase in the selenium limit between the 2015 ruling and 2019 proposal, it is assumed that the required residence time for biological systems would tend to decrease. However, industry leaders [5] have made their concerns clear regarding the efficacy of the LRTR process even in the face of a relaxed selenium constraint; they point out that EPA's sampling data [8] is not representative of influent selenium concentrations and may privilege LRTR systems in these arbitrarily low selenium environments. EPA does not report the residence times used in determining the effectiveness of an LRTR process—thus, it is not possible to quantitatively determine what a residence time would need to be for a given technology and given selenium reduction ratio. However, EPA sampling data can still be used for a qualitative estimate of the success of the LRTR process in a high-selenium environment—this is discussed in greater detail in the performance estimates in the following section.

In addition to the overall removal of selenium, the altered residence time of an LRTR system is expected to impact a number of other parameters, including the amount of nutrient required to maintain ORP and chemical oxygen demand (COD), and the amount of biomass solids that are produced and must be handled in the backwash. Envirogen's fixed-film bioreactor system [28] is reported to produce a smaller amount of biomass solids as a result of its design, including a lower hydraulic residence time—however, this technology still requires full-scale FGD wastewater testing. EPA does not quantitatively discuss these parameters [4]; thus, this study assumes that both the influent nutrient and solids production are solely a function of flow rate and are identical between the LRTR and HRTR systems.

3 RESULTS, ANALYSIS, AND DISCUSSION

This section discusses the results of this study, in comparison between the two different technologies used for biological treatment. Qualitative discussion follows the cost estimates in Section 3.2.

3.1 PERFORMANCE RESULTS

A comparison between the surface discharge plant and the CP + biological treatment system can be seen in the Case 0 and Case 1 comparison in the NETL study, "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants." [7] The 2019 ELG proposal does not change this performance comparison, and there is insufficient information to quantitatively discuss the LRTR and HRTR differences in terms of net plant efficiency, water consumption or auxiliary loads.

In lieu of a quantitative performance comparison, five EPA plants were examined in terms of their influent and effluent concentrations of chloride and ELG contaminants, in order to determine if the technology employed at these plants (CP + LRTR) would be sufficient as-is to reach the required reduction in order to meet the 2019 proposal. [2] Based on discussions with LRTR vendors, the reduction in selenium and nitrate/nitrite should follow a first-order reaction rate:

$$r = -\frac{dC_A}{dt} = kC_A$$

Where:

r is the reaction rate (or the reduction of the concentration of reactant A with time)

k is the reaction rate constant

C_A is the concentration of reactant A

Using a first-order reactor design equation (and assuming that the final concentration of a reactant is much smaller than its initial concentration), the following relationship can be used for residence time:

$$\frac{V}{\dot{v}} = \tau = \frac{C_{A0}}{kC_A}$$

Where:

V is the reactor volume

$\dot{v}$ is the volumetric flow rate

τ is the residence time for the reaction

C_{A0} is the initial concentration of reactant A

Using this relationship for the removal of selenium and nitrate/nitrite requires the following assumptions:

- For both selenium and nitrate/nitrite, the reactions must be of a strict A → B form; side reactions and inhibitors to the overall reaction are not accounted for.
- The reactor vessel is assumed to be a continuously-stirred tank reactor (the actual design is usually a packed bed that requires more information; however, this is sufficient to compare residence times in a qualitative sense if an individual reactor does not need designing).
- The initial concentration is much larger than the final concentration (the top term is normally C_{A0}–C_A; however, C_{A0} can be used for a rough estimate).

Without knowing the reaction rate constant k, the residence time cannot be quantitatively discussed; however, the reduction in pollutant in the five EPA plants can be compared to the required reduction assumed in the design basis. The reduction ratio used to qualitatively compare the two is calculated by the following:

$$reduction\ ratio = \frac{C_{A0}}{C_A}$$

If the reduction ratio from the five EPA plants is larger than the required reduction ratio from the design basis, it can be assumed that the residence time employed at the EPA plants is sufficient for selenium or nitrate/nitrite removal. The selenium and nitrate/nitrite reduction ratios are displayed in Exhibit 3-2 and Exhibit 3-3, and the design basis requirements are shown in Exhibit 3-1.

Exhibit 3-1. Reduction ratios using the design basis and ELGs from the 2019 proposal

Parameter (units)	Selenium Value	Nitrate Value
Assumed influent concentration (ppb)	5,000	30,000
ELG limit (ppb)	16.6	2,600
Required reduction ratio, percent	301.2, 99.7%	11.5, 91.3%

Exhibit 3-2. Selenium concentrations and reduction ratios from the five EPA plants [8]

EPA Plant #	Average Selenium Reduction		Maximum Selenium Reduction		Required Selenium Reduction	
	Ratio	%	Ratio	%	Ratio	%
2019	10.3	90.3	58.7	98.3	301.2	99.7
2027	13.8	92.8	44.9	97.8		
2066	9.3	89.2	55.5	98.2		
2090	9.3	89.8	37.4	97.3		
2097	42.2	97.6	354.7	99.7		

Exhibit 3-3. Nitrate/nitrite concentrations and reduction ratios from the five EPA plants [8]

EPA Plant #	Average Nitrate/ Nitrite Reduction		Maximum Nitrate/ Nitrite Reduction		Required Nitrate/ Nitrite Reduction	
	Ratio	%	Ratio	%	Ratio	%
2019	1.43	30.1	10.8	90.7	11.5	91.3
2027	525.8	99.8	1916.7	99.9		
2066	3.7	73.0	15.7	93.6		
2090	1.6	37.5	4.5	77.8		
2097	1.0	0	1.0	0		

The required reduction ratios from the design basis and ELGs exceed most of the selenium reduction ratios shown in the five EPA plants, with one data point from Plant 2097 as the sole exception. Based on vendor communications, selenium is expected to determine the required residence time for the biological treatment system; therefore, this data suggests that the residence times for the LRTR systems tested by EPA would be insufficient to meet the long-term reductions required in the design basis. However, the influent selenium concentrations are also much lower than the design basis—this could also suggest that more data are needed at very high selenium concentrations in order to determine if LRTR can handle these high selenium flows.

3.2 Cost Results

A summary of the cost results from this study is given in Exhibit 3-4. The results shown are estimates given from EPA cost curves for both LRTR and HRTR systems and compared with earlier results from the NETL study, "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants." [7] These costs have been updated to the Bituminous Baseline, revision 4 method. [9]

Exhibit 3-4. Summary of cost results

Case	0[A]	1[B]	CP + LRTR (EPA)	CP + HRTR (EPA)
Total Plant Cost, $/kW	**0**	**81**	**32**	**37**
Total Plant Cost Wastewater Treatment, $/kW	0	69	23	27
Total Plant Cost, $/1,000	**0**	**$52,575**	**$21,023**	**$23,838**
Total Plant Cost Wastewater Treatment, $/1,000	0	$45,040	$14,936	$17,604
TOC, $/kW	0	98	39	44
TASC, $/kW	0	103	41	47
Levelized Cost of Electricity,[F] $/MWh	36.1	37.5	36.6	36.7
Capital, $/MWh	0.0	1.0	0.4	0.4
Fixed O&M, $/MWh	8.9	9.3	9.1	9.1

Case	0[A]	1[B]	CP + LRTR (EPA)	CP + HRTR (EPA)
Variable O&M, $/MWh	7.6	7.7	7.6	7.7
Fuel, $/MWh	19.5	19.5	19.5	19.5

[A]No treatment – Surface water discharge
[B]CP + HRTR (Case 1 from [7])

EPA estimates (for both LRTR and HRTR systems) are substantially lower than the original estimates used in the aforementioned prior NETL study—however, the difference between the two EPA estimates is fairly small ($14.9 million vs. $17.6 million) and amounts to a negligible difference in LCOE ($36.6 vs. $36.7/MWh). Further estimates from industry leaders suggest that EPA grossly underpredicts the cost of both CP and biological treatment (LRTR and HRTR) systems. Electric Power Research Institute (EPRI) corrections [5] are given as a range based on the multiplicative factors found between vendor estimates and EPA's cost estimates in Exhibit 3-5 and Exhibit 3-6. The calculations for these corrections are discussed in the Appendix: EPA Cost Curves and Implementation.

Exhibit 3-5. Summary of LRTR cost corrections

Case	1	CP + LRTR (EPA)	CP + LRTR (EPRI corrections)	CP + LRTR (ratio w/ 1)
Total Plant Cost, $/kW	**81**	**32**	**44 – 67**	**70**
Total Plant Cost Wastewater Treatment, $/kW	69	23	34 – 56	59
Total Plant Cost, $/1,000	**$52,575**	**$21,023**	**$28,399 – $43,467**	**45,453**
Total Plant Cost Wastewater Treatment, $/1,000	$45,040	$14,936	$21,937 – $36,314	38,214
TOC, $/kW	98	39	53 – 81	85
TASC, $/kW	103	41	56 – 85	89
Levelized Cost of Electricity,[F] $/MWh	37.5	36.6	36.9 – 37.4	37.3
Capital, $/MWh	1.0	0.4	0.5 – 0.8	0.8
Fixed O&M, $/MWh	9.3	9.1	9.1 – 9.3	9.3
Variable O&M, $/MWh	7.7	7.6	7.7	7.7
Fuel, $/MWh	19.5	19.5	19.5	19.5

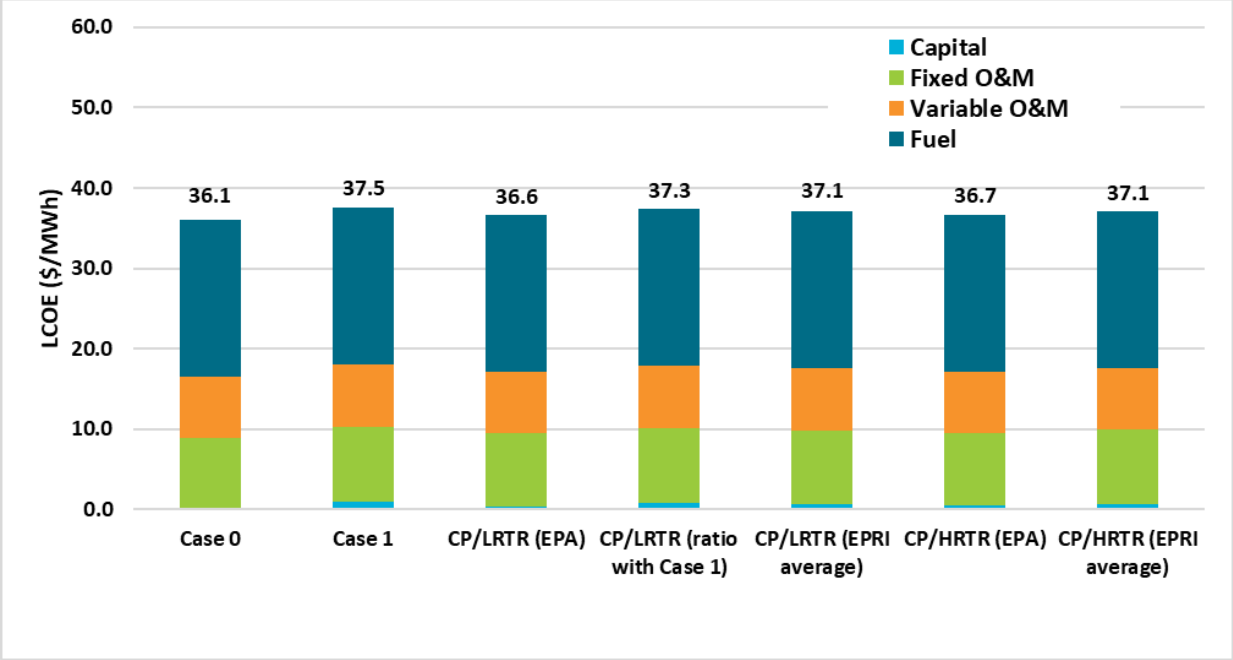
Exhibit 3-6. Summary of HRTR cost corrections

Case	1	CP + HRTR (EPA)	CP + HRTR (EPRI corrections)
Total Plant Cost, $/kW	**81**	**37**	**50 – 65**
Total Plant Cost Wastewater Treatment, $/kW	69	27	40 – 54
Total Plant Cost, $/1,000	**$52,575**	**$23,838**	**$32,323 – $41,997**
Total Plant Cost Wastewater Treatment, $/1,000	$45,040	$17,604	$25,672 – $34,908
TOC, $/kW	98	44	60 – 78
TASC, $/kW	103	47	63 – 82

Case	1	CP + HRTR (EPA)	CP + HRTR (EPRI corrections)
Levelized Cost of Electricity, $/MWh	37.5	36.7	37.0 – 37.3
Capital, $/MWh	1.0	0.4	0.6 – 0.8
Fixed O&M, $/MWh	9.3	9.1	9.2
Variable O&M, $/MWh	7.7	7.7	7.7
Fuel, $/MWh	19.5	19.5	19.5

The variability in LRTR systems is wider than in HRTR systems; thus, at these conditions, the range for an LRTR system (with EPRI corrections) extends beyond the HRTR system's range. Additionally, the cost for the LRTR system is shown in Exhibit 3-5 as a ratio with the Case 1 numbers from "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants" [7]; that is, the ratio of LRTR and HRTR costs from EPA is used to predict the LRTR cost with the same assumptions as made over the course of that study. All the LCOE estimates are shown together in Exhibit 3-7. Overall, the different estimates of the same technology yield a $0.5–1.4/MWh increase in the overall LCOE of the sub-PC plant—a 1.4–3.9 percent increase in LCOE. In terms of a cost per m^3 influent, these treatment costs for LRTR and HRTR systems range $25.1–60.2/$m^3$ and $30.1–70.3/$m^3$, respectively.

Exhibit 3-7. LCOE estimates for the LRTR and HRTR systems, with Case 0 and Case 1 from the NETL FGD study as a reference



3.3 SENSITIVITY ANALYSES

In addition to the base case analyses, a selection of parameters were varied to produce an improved range of cost estimates for the CP/LRTR and CP/HRTR systems. The full extent of

sensitivity analyses conducted is shown in Exhibit 3-8. These sensitivities are discussed in detail in the following sections.

Exhibit 3-8. Full range of sensitivity analyses conducted

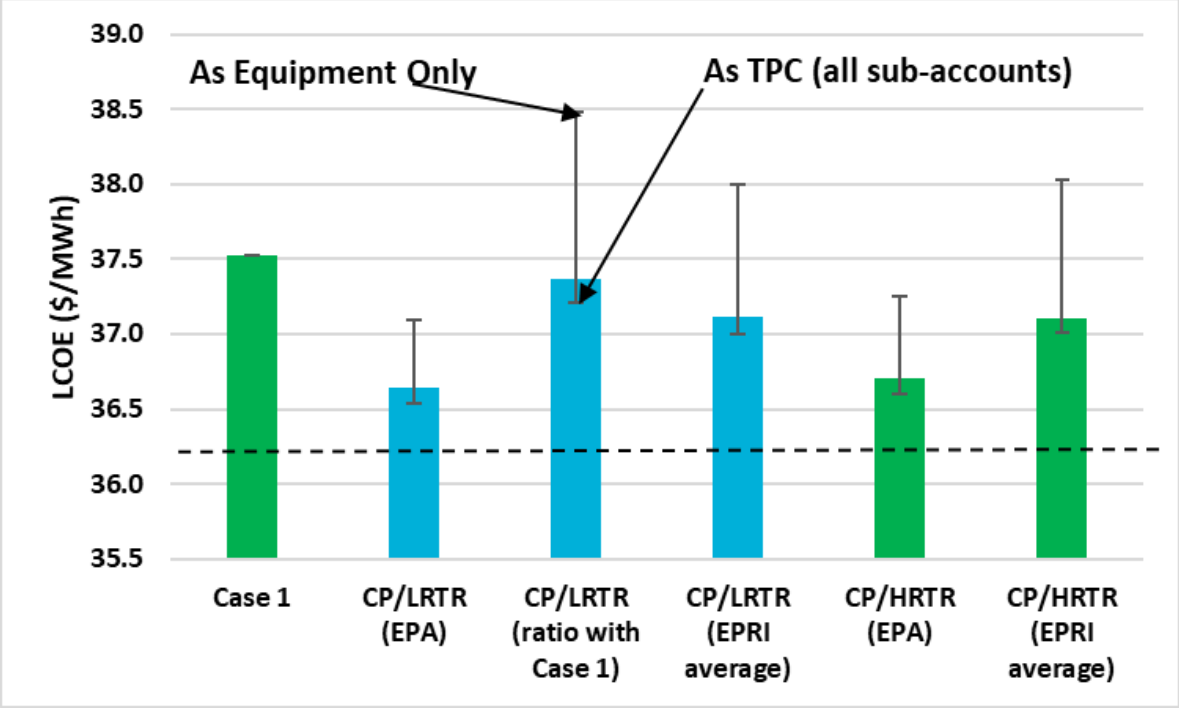
Parameter (Units)	Default Value	Range
Capital Cost Level	TPC	Equipment, TPC (3.8), TPC (all sub-accounts)
FGD Flow Rate (gpm)	57	25 – 300
Capital Cost (%)	Baseline	-50/+100%
O&M Costs (%)	Baseline	-50/+100%
Retrofit Factor (%)	Baseline	-50/+100%
Capital Expenditure Period (years)	1	1 – 3
Nitrates in FGD (mg/L)	30	30 – 105

3.3.1 Equipment Type

The base case analysis assumes that the equipment costs provided by EPA can be assumed to be total plant costs (TPCs). [4] The cost curves provided by EPA are expected to include all costs associated with the manufacture, installation and integration with the existing plant, and control of the equipment. Because of this, it is possible that not only should the costs be treated as the TPC for the wastewater treatment system alone (sub-account 3.8), but also all subsequent costs including site preparation and facilities and control schemes (as all of these are mentioned by name in the EPA supplementary documentation). On the other hand, it is entirely possible that these costs are underestimated based on the communication from industry leaders [5]; thus, these costs could also just be considered as the purchased equipment costs for sub-account 3.8, which are thereby adjusted to bare erected costs and TPCs accordingly.

Exhibit 3-9 displays the LCOE impacts of these changes versus the base case scenario (sub-account 3.8 TPC) on the five cases considered—CP/LRTR and CP/HRTR from the EPA cost curves [4], CP/LRTR and CP/HRTR using the EPRI corrections [5], and CP/LRTR referenced to Case 1 of the NETL study, "Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants." [7] Case 0 of that same study is used as a reference as the black dashed line in Exhibit 3-9. Including all sub-accounts into the wastewater treatment cost curves brings the LCOE down by around 0.3 percent; this is less significant than the impact of treating these cost curves purely as equipment costs for sub-account 3.8, which increases the LCOE by 1.2–1.4 percent.

Exhibit 3-9. The LCOE impacts of considering the capital costs as equipment (purchased or TPC) or TPC for all related sub-accounts

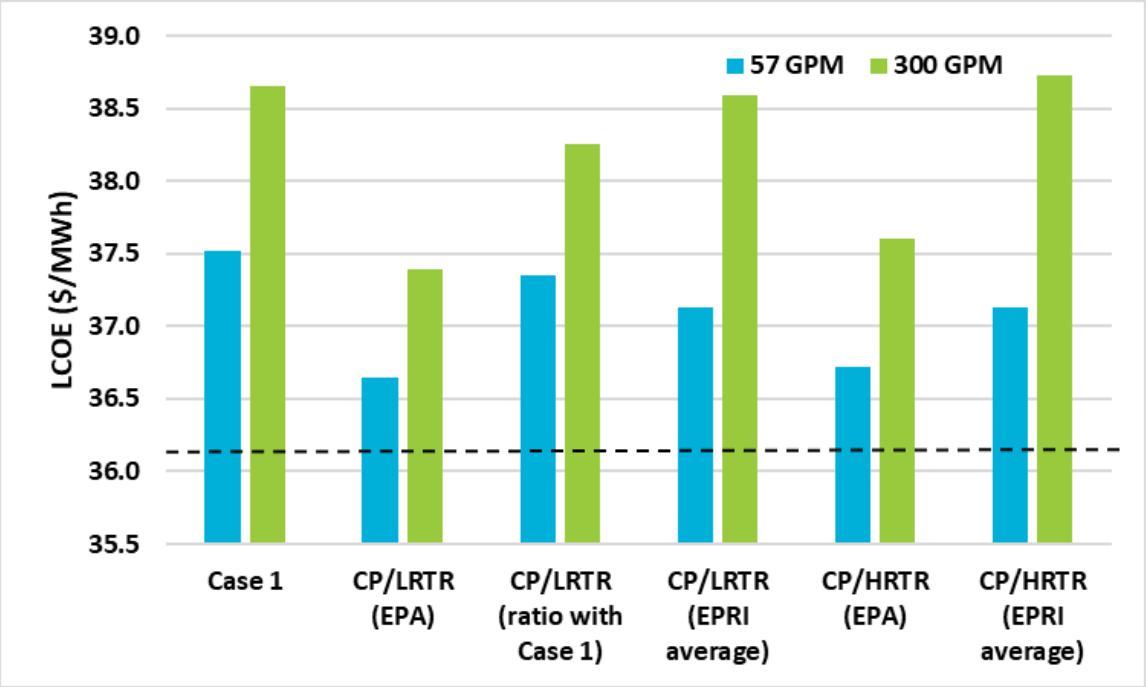


Note: "As Equipment Only" and "As TPC (all sub-accounts)" applies to all error bars.

3.3.2 FGD Wastewater Flow Rate

The flow rate of wastewater headed to the FGD treatment system was varied from 57 to 300 gpm as per Exhibit 3-8. The impact of this wastewater flowrate increase is shown in terms of LCOE in Exhibit 3-10. The cost difference between the LRTR and HRTR systems widens at higher flow rates—the difference in TPC for the two systems is roughly 25.1 percent at 300 gpm (versus 15.1 percent at 57 gpm). However, the impact of this cost savings on the LCOE is muted—the difference between the two systems for EPA cost curves [4] is $0.2/MWh (37.4 versus 37.6) as opposed to $0.1/MWh at 57 gpm (36.6 versus 36.7).

Exhibit 3-10. The impact of flow rate on the LCOE of the different cases



3.3.3 Capital and Non-Fuel O&M Costs

As per Exhibit 3-8, the capital and non-fuel O&M costs were both varied by -50 percent and +100 percent—their impacts on the overall LCOE of the different cases is shown in Exhibit 3-11 and Exhibit 3-12. Varying the capital costs is shown to be more significant than the O&M costs—when using EPA cost curves, however, these differences are both small, ranging from a 0.4–0.9 percent difference in LCOE for capital cost changes and a 0.2–0.6 percent change in LCOE for the O&M cost changes.

Exhibit 3-11. The impact of varying capital costs on the overall LCOE

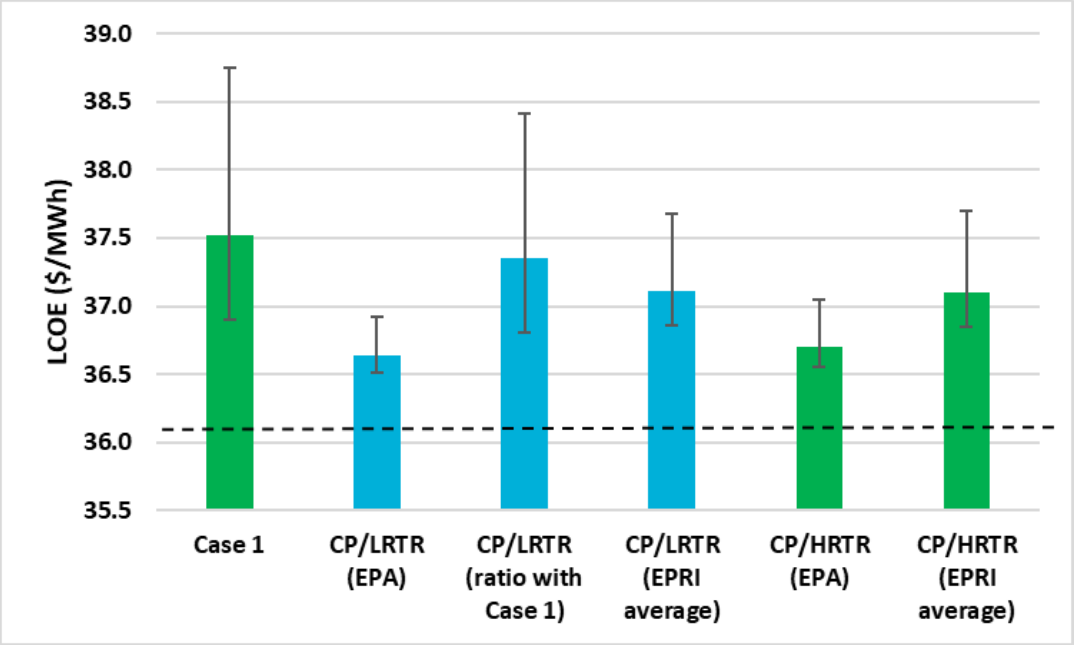
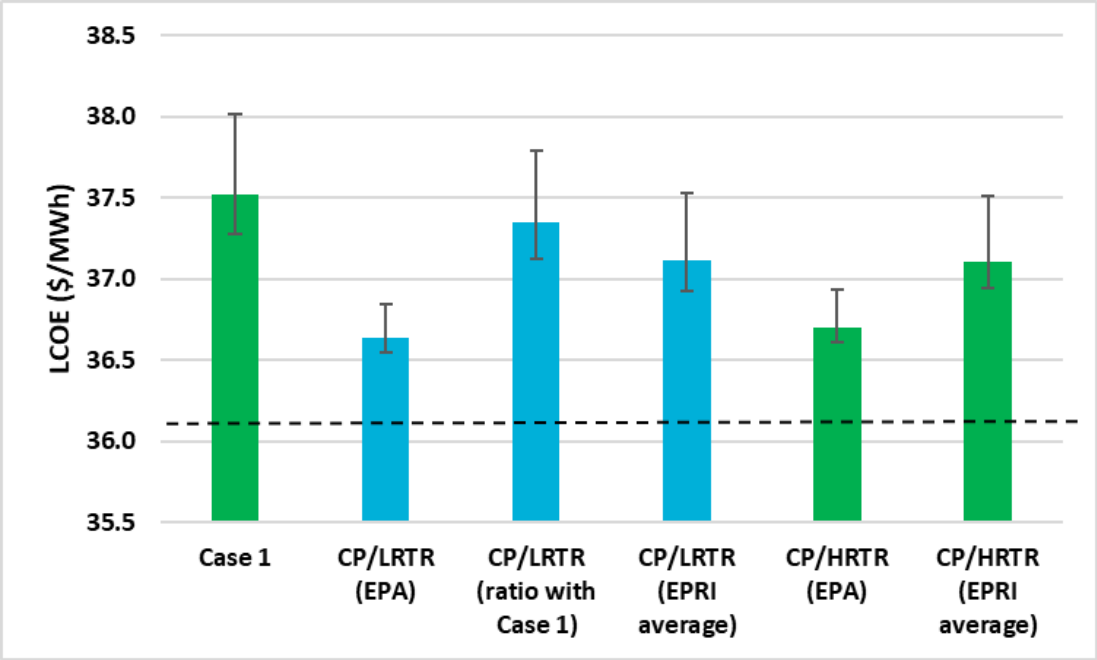


Exhibit 3-12. The impact of varying O&M costs on the overall LCOE



3.3.4 Additional Sensitivity Analyses

The full impact of the different sensitivity analyses (including FGD flow rate, capital cost and O&M costs from Section 3.3.2 and Section 3.3.3) on LCOE using EPA costs are shown in Exhibit 3-13 and Exhibit 3-14. The impact of retrofit difficulty factor, increased nitrate level, and CapEx

period (using the Bituminous Baseline, revision 4 method [9]) is smaller than the aforementioned sensitivity analyses. EPA's cost curves factor in the pretreatment required for nitrates/nitrites, but this does not have a significant impact on the capital cost of the system. Ultimately, the retrofit difficulty factor alters the capital costs of the system to a lesser degree than the capital cost sensitivity analysis, making for a more muted impact. The full extent of these sensitivities can alter the cost of the system by -0.4 percent to +2.0 percent for the CP/LRTR LCOE and -0.4 percent to +2.4 percent for the CP/HRTR LCOE. The accuracy of each of these estimates is in line with a Class 4 estimate of -15 percent/+30 percent [9]; therefore, none of these sensitivity analyses are considered to have a significant impact, and the difference between a CP/LRTR system and a CP/HRTR system for a coal-fired power plant is considered negligible.

Exhibit 3-13. The sum of impacts to LCOE for the different CP/LRTR sensitivity analyses

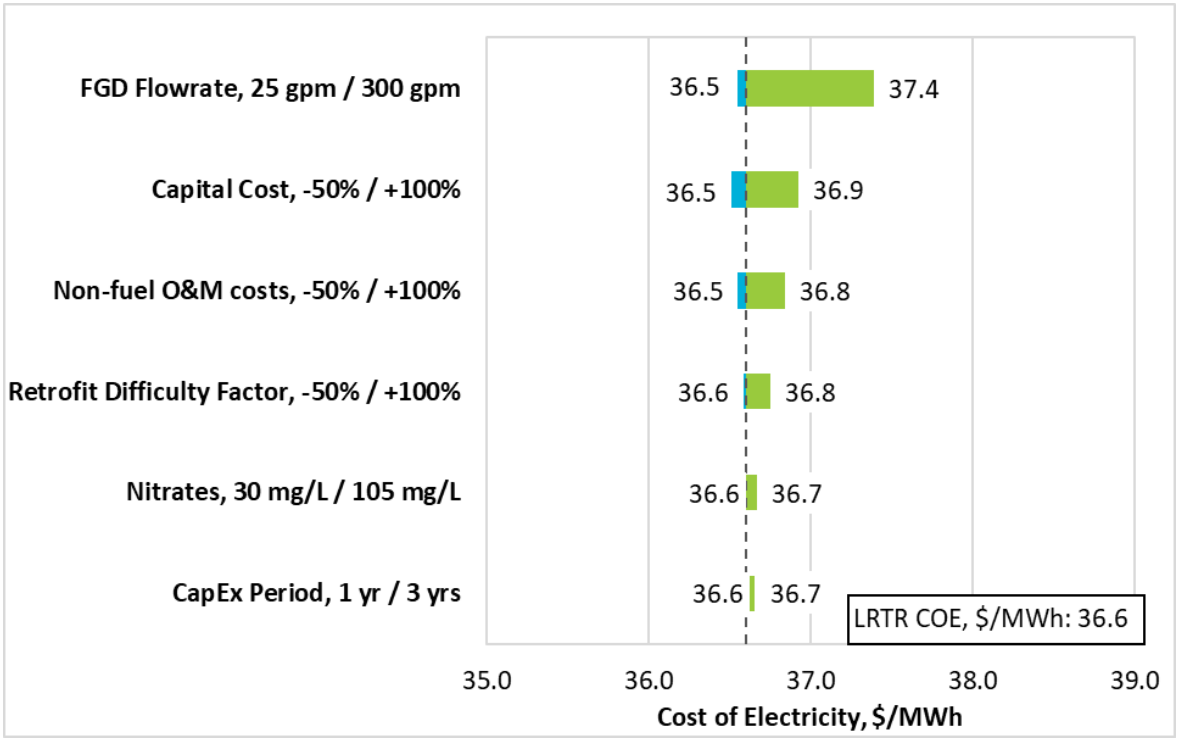
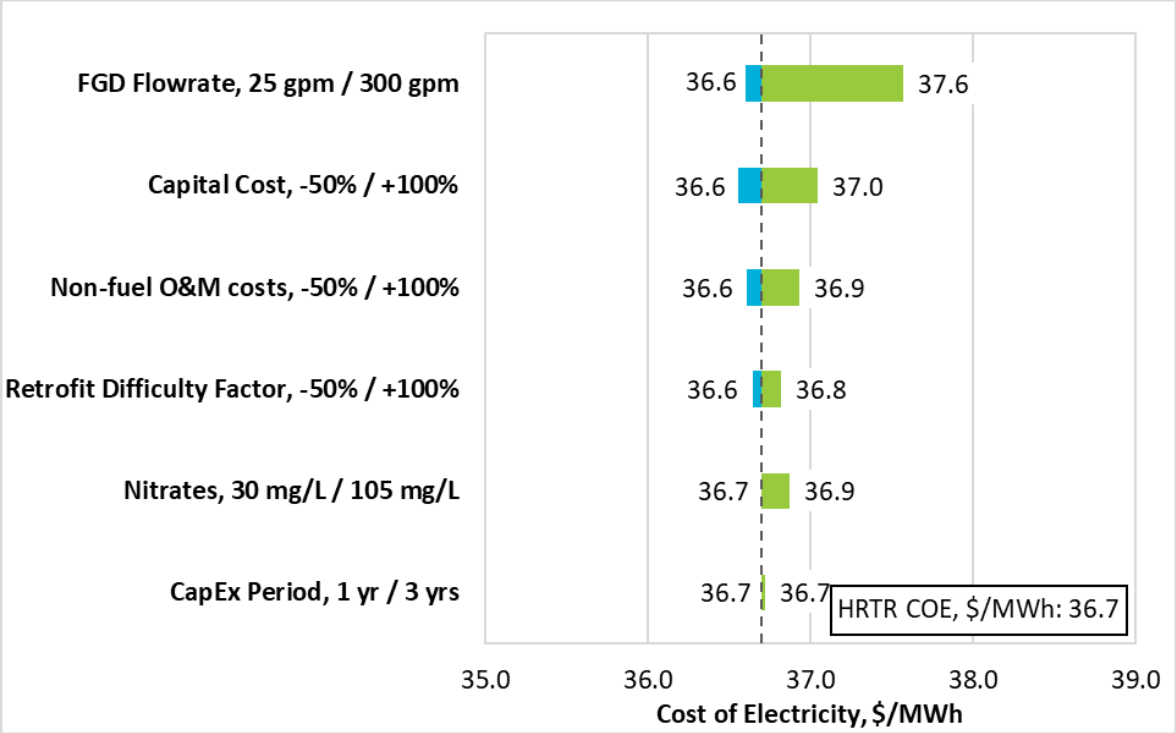


Exhibit 3-14. The sum of impacts to LCOE for the different CP/HRTR sensitivity analyses



4 LIMITATIONS AND RESEARCH AND DEVELOPMENT OPPORTUNITIES

Numerous limitations for the applicability and scope of this work are stated (or clarified) here:

- The impact of stream composition beyond TDS and nitrate/nitrite levels is not known. High nitrates/nitrites will require additional bioreactors for upstream pretreatment. High TDS levels (above 20,000 ppm Cl^-) will require dilution prior to biological treatment operation. [4]
- The impact of operating chloride level on the wet FGD scrubber liquor is unknown—where it impacts the stream composition, a higher TDS level in the FGD effluent may make treatment difficult as seen in the note above.
- The LRTR system is not known to impact plant emissions differently than the HRTR system. LRTR installations are expected to have smaller vessel sizes and should have a lower overall footprint. [4]
- The relationship between residence time and water consumption or solids production is not quantitatively known. An LRTR system would need to be evaluated at a real coal-fired power plant to determine if these impacts would change.
- The coal type is not specified in the EPA testing data (in order to anonymize the plant information)—based on the 2015 information, the assumption is still that biological treatment has not been verified for coal other than bituminous coal. Whether CP/LRTR or CP/HRTR systems would be the better choice in subbituminous or lignite coal plants remains to be seen.
- The limitations for the CP/LRTR system during off-design scenarios are similar to the CP/HRTR system—the system cannot handle sudden changes in chloride or nitrate levels and the other impacts of a differing stream composition are not known. It is unknown whether CP/LRTR systems are able to better handle changes in load or off-design conditions relative to a CP/HRTR system—vendor communications for CP/LRTR systems suggest that the same issues apply, such as sudden spikes in chloride concentrations killing off bacteria colonies within both systems.

A number of research and development opportunities for biological treatment systems in general have been discussed in the NETL study, “Techno-Economic Analysis and Evaluation of Wet FGD Wastewater Treatment Processes at Existing Plants.” [7] Most of these are still relevant for LRTR, and additional opportunities exist for LRTR specifically. If EPA plant data are made available (including residence time, or vessel volume and flow rate as a means of calculating said residence time), then these estimates can be quantified and applied to different stream concentrations and flow rates. Barring this, pilot tests using concentrations similar to the design basis in this study and LRTR systems designed for high-selenium flows is suggested. Improvements in technology will likely continue to decrease the required residence time of a biological treatment system for selenium removal; however, the inconsistent selenium removal of biological treatment systems in general [5] [7] has yet to be overcome. If a system can be

designed with consistently low selenium concentration in the effluent with a smaller residence time than an HRTR system, then that system will be the obvious BAT for the current ELGs.

5 CONCLUSIONS

This study was conducted to characterize the estimated performance, capital, and O&M costs, applicability limitations, technical challenges, and research and development opportunities associated with the new BAT provided by EPA in the 2019 proposal [2] as compared with the 2015 BAT. [1] A number of quantitative cost comparisons were conducted based on parameters such as FGD flow rate, capital and O&M costs, and influent nitrate levels. In addition, qualitative calculations were conducted in order to determine if an LRTR would have the appropriate residence time needed for selenium reduction. The findings are thus:

- Based on the calculations conducted on influent and effluent selenium levels in the five EPA plants sampled, the residence time employed at these plants may be insufficient for reduction of high-selenium flows to the levels required by ELGs. LRTR systems should still be sufficient for nitrate/nitrite reduction.
- Across all of the parameters studied, a CP/LRTR system is not expected to decrease (or increase) LCOE by a significant amount relative to a CP/HRTR system.

A number of parameters lack quantitative information—discussed in Section 4. The impact of these limitations will require full-scale implementation of CP/LRTR systems at existing coal-fired power plants. It is expected that the LRTR system will continue to improve in residence time, selenium removal, and reliability during off-design conditions; future iterations of this work should include the quantitative impact of these improvements when they become available.

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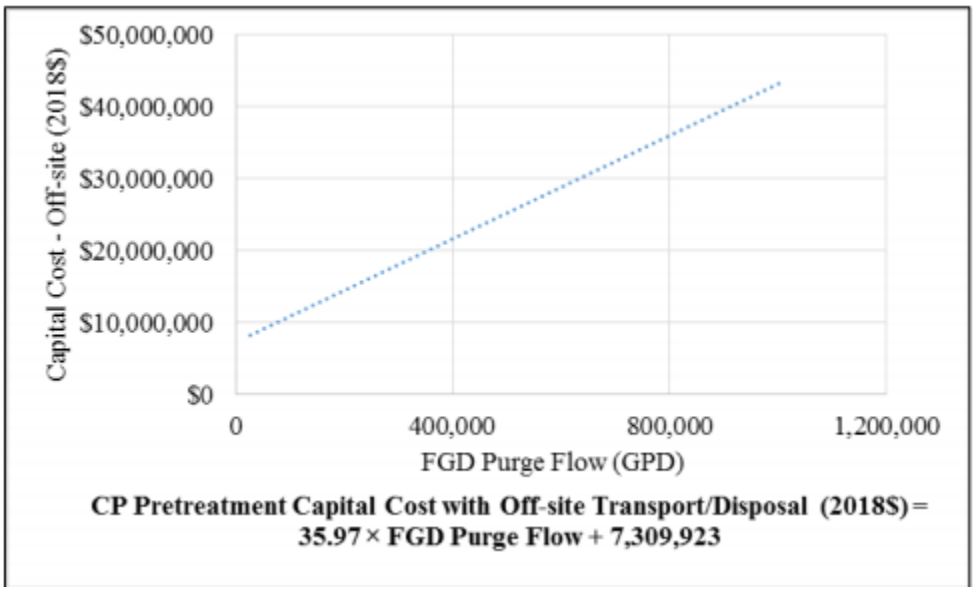
APPENDIX: EPA COST CURVES AND IMPLEMENTATION

The cost curves used in the Environmental Protection Agency (EPA) supplementary documentation [4] are directly employed in the cost estimations of this study. The curves are simple linear relationships between the flue gas desulfurization (FGD) flow rate (in gallons per day [gpd]) and the capital and operation and maintenance (O&M) costs of a given treatment system in $2018 dollars. A sample curve is presented from the supplementary document (Exhibit A-1) for use in this study for chemical precipitation (CP) with off-site transport/disposal of associated waste products. Using the FGD purge flow from this study (57 gallons per minute [gpm], or 82,080 gpd) the CP capital cost is expected to be roughly $10.3 million in 2018 dollars.

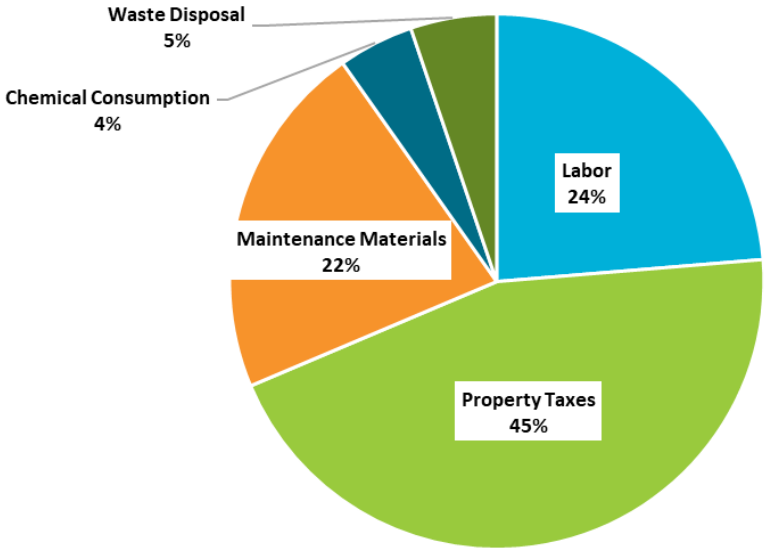
Exhibit A-1. Sample cost curve for CP



Source: EPA [4]

Using these values in the overall cost estimation is straightforward for capital costs—the value produced from the CP and biological treatment cost curves replaces the total plant cost (TPC) of the wastewater treatment system in sub-account 3.8 from the National Energy Technology Laboratory (NETL) FGD study. [7] The O&M costs are less straightforward—a simple linear relationship does not provide any estimate for labor costs, the cost of chemicals, etc. As a workaround, the difference in Case 1 and Case 0 were quantified in terms of their O&M costs, and the relative increases in each category were applied to all of the O&M increases for both the CP/low residence time reduction (LRTR) and CP/high residence time reduction (HRTR) systems. Exhibit A-2 provides the percentage breakdown for these O&M increases—thus, these percentages are applied to the O&M cost curves directly and added to the Case 0 value. This allows for a rough estimate of the fixed and variable (non-fuel) cost impacts used to calculate the levelized cost of electricity (LCOE) used in the cost estimates.

Exhibit A-2. The percentage breakdown of the O&M increases between Case 1 and Case 0



Increases in the cost curves based on Electric Power Research Institute (EPRI) input [5] were implemented using the relative increases that EPRI quantified between their own estimates and the estimates provided by EPA. These impacts were a simple multiplier applied to both the capital and O&M costs. For example, EPRI noted that the chemical pretreatment costs were 1.5–2.4 times more expensive than the EPA cost curves—thus, the range of costs were found by increasing the CP capital and O&M cost contributions by these multipliers.

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