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MODELAGEM HIDRODINÂMICA E ECOLÓGICA
DE RESERVATÓRIO TROPICAL EM CONDIÇÃO
DE ESCASSEZ HÍDRICA

Laura Melo Vieira Soares

Belo Horizonte

2018

**MODELAGEM HIDRODINÂMICA E ECOLÓGICA DE
RESERVATÓRIO TROPICAL EM CONDIÇÃO DE
ESCASSEZ HÍDRICA**

Laura Melo Vieira Soares

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UNIVERSIDADE FEDERAL DE MINAS GERAIS

Escola de Engenharia

Programa de Pós-Graduação em Saneamento, Meio Ambiente e Recursos Hídricos

Avenida Antônio Carlos, 6627 - 4º andar - 31270-901 - Belo Horizonte - BRASIL

Telefax: 55 (31) 3409-1882 - posgrad@desa.ufmg.br

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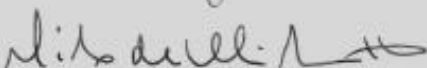
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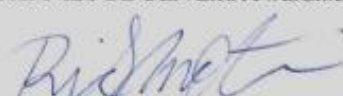
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LAURA MELO VIEIRA SOARES

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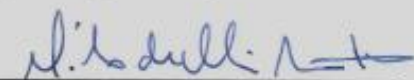

Profª. TALITA FERNANDA DAS GRAÇAS SILVA


Prof. NILO DE OLIVEIRA NASCIMENTO


Prof. RICARDO TEZINI MINOTI


Prof. MICHAEL MÄNNICH

Aprovada pelo Colegiado do PG SMARH


Prof. Nilo de Oliveira Nascimento
Coordenador

Belo Horizonte, 23 de março de 2018.

Versão Final aprovada por


Profª. Talita Fernanda das Graças Silva
Orientadora

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Ao meu companheiro para toda a vida,
Carlos, por dividir esse sonho com muito
incentivo e amor.

À minha família, pelo apoio incondicional,
suporte e carinho ao longo desta trajetória.

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RESUMO

Entre as principais preocupações da gestão de recursos hídricos estão a manutenção da qualidade da água e a redução da vulnerabilidade a eventos que oferecem riscos ambientais e sanitários. Considerando-se que os processos térmicos em sistemas lacustres influenciam as características físico-químicas da água e governam a dinâmica biológica dos organismos aquáticos, o presente estudo buscou compreender o comportamento hidrodinâmico de um reservatório tropical e as suas implicações sobre a dinâmica do fitoplâncton. Ademais, foram investigados os efeitos de um período de escassez hídrica sobre os fenômenos de estratificação térmica e mistura da coluna d'água e sobre a biomassa e a composição dos organismos fitoplanctônicos.

Para tanto, procedeu-se à análise de dados de campo e à modelagem matemática unidimensional. Empregou-se o modelo hidrodinâmico 1D-V (unidimensional vertical) *General Lake Model* acoplado ao modelo ecológico *Aquatic EcoDynamics* (GLM-AED) para simular os processos térmicos e a dinâmica do fitoplâncton no Reservatório Serra Azul (MG) de 2009 a 2016. Realizou-se análise de sensibilidade, bem como calibração manual e validação do modelo hidrodinâmico.

Os resultados da análise de sensibilidade indicaram que a temperatura da água é mais sensível às mudanças na velocidade do vento e na vazão afluente. O modelo GLM-AED representou o comportamento monomítico do reservatório e reproduziu a biomassa e a distribuição do fitoplâncton na coluna d'água. O período de escassez hídrica rompeu a estratificação térmica e o reservatório passou a apresentar um comportamento polimítico. Dados de campo indicam a redução da biomassa fitoplanctônica e mudança em sua composição durante esse período.

Esse estudo evidencia o emprego da modelagem matemática como ferramenta para auxiliar a gestão de sistemas de recursos hídricos. Os resultados apontam a necessidade de lidar com períodos de eventos extremos para o planejamento e gerenciamento dos sistemas lacustres, a fim de atender à demanda pela qualidade da água em mananciais, bem como satisfazer as funções ecológicas e ambientais de lagos e reservatórios.

Palavras-chave: sistemas lacustres, processos térmicos, dinâmica do fitoplâncton, GLM-AED, seca.

ABSTRACT

The water quality and the vulnerability reduction to events that pose environmental and health risks are among the main concerns of the management of water resources. Considering that thermal processes in lakes and reservoirs influence the physicochemical characteristics of water and govern the biological dynamics of aquatic organisms, the present study aimed to comprehend the hydrodynamic behavior of a tropical reservoir and its implications on phytoplankton dynamics. In addition, the effects of a drought period were investigated in terms of thermal stratification and mixing of the water column and of biomass and composition of phytoplanktonic organisms.

For this purpose, field data was analyzed and mathematical modelling was applied. The hydrodynamic General Lake Model 1D-V (one-dimensional vertical) was coupled with the ecological model Aquatic EcoDynamics (GLM-AED) to simulate the thermal processes and phytoplankton dynamics of Serra Azul Reservoir (MG) from 2009 to 2016. A sensitivity analysis was performed as well as manual calibration and validation of the hydrodynamic model.

The results of the sensitivity analysis indicated that the water temperature is more sensitive to changes in wind speed and inflows. The GLM-AED model represented the monomictic behavior of the reservoir and reproduced the biomass and distribution of phytoplankton in the water column. The drought period disrupted the thermal stratification and the reservoir was converted to a polymictic one. Field data indicate the reduction of phytoplankton biomass and a change in its composition during this period.

This study shows the use of mathematical modelling as a tool to assist the management of water resources systems. The results point out the need to deal with periods of extreme events for the planning and management of lakes and reservoirs, in order to meet the demand for water quality, as well as to satisfy the ecological and environmental functions of those lentic water bodies.

Keywords: lakes and reservoirs, thermal processes, phytoplankton dynamics, GLM-AED, drought.

SUMÁRIO

LISTA DE FIGURAS	IX
LISTA DE TABELAS.....	XII
LISTA DE ABREVIATURAS, SIGLAS E SÍMBOLOS.....	XIII
CAPÍTULO 1 – APRESENTAÇÃO E CONTEXTUALIZAÇÃO.....	1
1 INTRODUÇÃO.....	2
2 OBJETIVOS	4
2.1 OBJETIVO GERAL	4
2.2 OBJETIVOS ESPECÍFICOS.....	5
3 REVISÃO DA LITERATURA.....	5
3.1 PROCESSOS HIDRODINÂMICOS EM SISTEMAS LACUSTRES.....	5
3.1.1 Indicadores do comportamento hidrodinâmico de sistemas lacustres	7
3.2 DINÂMICA DO FITOPLÂNTON EM SISTEMAS LACUSTRES.....	11
3.3 EFEITOS DA ESCASSEZ HÍDRICA SOBRE SISTEMAS LACUSTRES	12
3.4 MODELAGEM MATEMÁTICA DE SISTEMAS LACUSTRES.....	13
3.5 O MODELO HIDRODINÂMICO GLM.....	16
3.6 O MÓDULO ECOLÓGICO AED.....	21
4 MATERIAIS	33
4.1 ÁREA DE ESTUDO	33
4.2 ESTAÇÕES DE MONITORAMENTO.....	36
4.3 TRATAMENTO DOS DADOS	39
REFERÊNCIAS	42
CAPÍTULO 2 – LONG-TERM CHARACTERIZATION (1987–2016) OF LIMNOLOGICAL FEATURES OF A TROPICAL RESERVOIR	47
ABSTRACT	48
1 INTRODUCTION.....	48
2 MATERIAL AND METHODS.....	50
2.1 STUDY SITE	50
2.2 FIELD DATA ANALYSIS	51
3 RESULTS AND DISCUSSION	52
3.1 THERMAL STRUCTURE	52
3.2 PHYSICO-CHEMICAL VARIABLES	53
3.3 PHYTOPLANKTON DYNAMICS.....	55
4 CONCLUSION.....	58
REFERENCES.....	59
CAPÍTULO 3 – MODELLING THERMAL DYNAMICS AND PHYTOPLANKTON IN A TROPICAL WATER SUPPLY RESRVOIR.....	63
ABSTRACT	64
1 INTRODUCTION.....	64
2 MATERIAL AND METHODS.....	66
2.1 STUDY AREA	66
2.2 MONITORING STATIONS	67

2.3	FIELD DATA	67
2.4	MODEL DESCRIPTION	68
2.5	MODEL INPUTS.....	72
2.6	SIMULATION	72
2.7	SENSITIVITY ANALYSIS	73
2.8	CALIBRATION AND VALIDATION	74
3	RESULTS AND DISCUSSION	75
3.1	SENSITIVITY ANALYSIS OF THE HYDRODYNAMIC MODEL	75
3.2	SENSITIVITY ANALYSIS OF THE ECOLOGICAL MODEL	76
3.3	PERFORMANCE OF THE HYDRODYNAMIC MODEL	77
3.4	PERFORMANCE OF THE ECOLOGICAL MODEL.....	79
3.5	THERMAL DYNAMICS.....	81
3.6	PHYTOPLANKTON DYNAMICS.....	82
4	CONCLUSIONS	83
	REFERENCES.....	84
	CAPÍTULO 4 – MODELLING DROUGHT IMPACTS ON THE HYDRODYNAMICS OF A TROPICAL WATER SUPPLY RESERVOIR.....	87
	ABSTRACT	88
1	INTRODUCTION.....	88
2	MATERIAL AND METHODS.....	90
2.1	STUDY SITE	90
2.2	HIDROLOGICAL, METEOROLOGICAL AND WATER QUALITY DATA.....	91
2.3	DATA PROCESSING	93
2.4	THE ONE-DIMENSIONAL HYDRODYNAMIC RESERVOIR MODEL	94
2.5	MODELLING SETUP	95
2.6	HYDRODYNAMICS INDICATORS.....	95
3	RESULTS.....	97
3.1	MODEL PERFORMANCE	97
3.2	DROUGHT IMPACT ON RESERVOIR HYDRODYNAMICS	98
4	DISCUSSION.....	101
4.1	MODEL PERFORMANCE	101
4.2	DROUGHT IMPACT ON RESERVOIR HYDRODYNAMICS	102
4.3	RISKS ON WATER SUPPLY DUE TO DROUGHT	103
5	CONCLUSIONS	104
	REFERENCES.....	105
	CAPÍTULO 5 – EFFECTS OF A DROUGHT PERIOD ON PHYSICO-CHEMICAL VARIABLES AND PHYTOPLANKTON DYNAMICS IN A TROPICAL RESERVOIR.....	109
	ABSTRACT	110
1	INTRODUCTION.....	110
2	MATERIAL AND METHODS.....	112
2.1	STUDY SITE	112
2.2	FIELD DATA ANALYSIS.....	114
3	RESULTS AND DISCUSSION	114
3.1	PHYSICO-CHEMICAL VARIABLES	114
3.2	PHYTOPLANKTON DYNAMICS.....	116
4	CONCLUSIONS	118

REFERENCES	118
CAPÍTULO 6 – CONSIDERAÇÕES FINAIS	121
1 CONCLUSÕES	122
2 RECOMENDAÇÕES PARA ESTUDOS FUTUROS.....	123
APÊNDICES	124
APÊNDICE A	125
APPENDIX B	127

LISTA DE FIGURAS

Figura 1.1 – Dinâmica de estratificação térmica e mistura em sistemas lacustres. Perfis de um lago a) em condições de estratificação e b) de mistura.	7
Figura 1.2 – Volume mensal do reservatório Serra Azul a partir de 2014. Fonte: COPASA, 2018.	13
Figura 1.3 – Modelo conceitual do GLM, seus dados de entrada (texto em azul) e processos simulados (texto em preto). Fonte: Hipsey <i>et al.</i> , 2014.....	16
Figura 1.4 – Modelo conceitual do AED. Fonte: Hipsey <i>et al.</i> , 2013.....	21
Figura 1.5 – Localização do reservatório Serra Azul, sua bacia hidrográfica e tributários.	33
Figura 1.6 – Uso do solo na bacia hidrográfica do reservatório Serra Azul. Tratamento de imagem do satélite Sentinel com resolução de 10 m. Fonte: Matos <i>et al.</i> , 2017.....	35
Figura 1.7 – Normais climatológicas da temperatura do ar e da precipitação acumulada da estação meteorológica Florestal de 1961 a 1990. Fonte: INMET, 2017.	36
Figura 1.8 – Localização das estações de monitoramento consultadas: estação 426, estação 371, estação Jardim, estação Serra Azul e estação Florestal.	38
Figura 1.9 – Período e frequência de monitoramento das variáveis de interesse. Fonte: COPASA, 2018; INMET, 2017.	38
Figura 1.10 – Regressão linear entre temperatura da água e variáveis ambientais: a) temperatura do ar, b) temperatura do ar máxima diária, c) temperatura do ar mínima diária, d) radiação média diária, e) radiação total diária e f) radiação máxima diária. Fonte: INMET, 2017.	40
Figure 2.1 – Mean monthly water temperature profile (blue line) and profiles monitored (grey line) from 1987 to 2016. Source of data: COPASA, 2017.....	52
Figure 2.2 – Water temperature at 0.5 m depth (red) and 15 m depth (blue) at Serra Azul reservoir from 1987 to 2016. Dots are the monitored values and lines are the linear interpolation between them. Source of data: COPASA, 2017.....	53
Figure 2.3 – Limnological variables of Serra Azul reservoir from 1987 to 2016: a) Secchi depth (m), b) turbidity (uT), c) dissolved oxygen concentration (mgL ⁻¹), d) nitrate concentration (mgL ⁻¹) and e) conductivity (µScm ⁻¹). Dotted lines correspond to July, 1 st of each year. Source of data: COPASA, 2017.....	54
Figure 2.4 – Phytoplankton biomass of each class from 1987 to 2016. Source of data: COPASA, 2017.	56
Figure 2.5 – Seasonal variability of mean phytoplankton biomass in Serra Azul reservoir for all years (from 1987 to 2016): a) chlorophyceae and cyanobacteria, and b) chrysophyceae and pyrophyceae. Source of data: COPASA, 2017.	57
Figure 2.6 – Trophic State Index of Serra Azul reservoir from 1987 to 2016. Source of data: COPASA, 2017.	58
Figure 3.1 – Serra Azul reservoir, its tributaries and monitoring stations: 426, 371, Jardim, Serra Azul and Florestal.	66
Figure 3.2 – Meteorological and hydrological variables measured at Florestal station and Jardim station: a) short-wave radiation, b) air temperature, c) air relative	

humidity, d) windspeed, e) rainfall and f) streamflow. Source fo data: INMET and CPRM.	69
Figure 3.3 – Total phytoplankton biomass per year, distribution of groups and percentage of cyanobacteria in total biomass from 2009 to 2013. Source of data: COPASA.	70
Figure 3.4 – Water temperature simulated (blue line) and observed (red marker) at a) 0.5 m depth, b) 5 m depth and c) 15 m depth.	77
Figure 3.5 – Simulated (blue line) and observed (red markers) water temperature profiles of Serra Azul reservoir.	78
Figure 3.6 – Simulated and observed water temperature at 0.5 m, 5 m and 15 m depth for a) the calibration period (2009-2012) and for b) the validation period (2013). Dashed line represents the equality between simulated and observed values.	78
Figure 3.7 – Simulated daily mean chlorophyll-a concentration at Secchi depths (from 1.4 m depth until 5 m depth) (blue line) and observed field data (red marker) for a) cyanobacteria and b) other groups.	80
Figure 3.8 – Simulated and observed chlorophyll-a concentration at Secchi depths (from 1.4 m depth until 5m depth) from 2009 to 2013 for a) cyanobacteria and b) other groups.	80
Figure 3.9 – Simulated water temperature in Serra Azul reservoir.	81
Figure 3.10 – Simulated water temperature at a) 0.5 m depth and b) 15 m depth.	82
Figure 3.11 – Simulated chlorophyll-a concentration in Serra Azul reservoir.	83
 Figure 4.1 – Serra Azul volume from 2014 to 2018 in percentage of its total storage capacity ($8.16 \times 10^7 \text{m}^3$) (COPASA 2018).	91
Figure 4.2 – Serra Azul reservoir, its tributaries and monitoring stations: 426, 371, Jardim, Serra Azul and Florestal.	92
Figure 4.3 – Meteorological (Florestal station – INMET) and hydrological (Jardim station – CPRM) variables: short-wave radiation (Wm^{-2}), air temperature ($^{\circ}\text{C}$), air relative humidity (%), wind speed (ms^{-1}), rainfall (mm) and streamflow (m^3s^{-1}).	92
Figure 4.4 – Simulated (blue line) and observed (red marker) water levels in Serra Azul reservoir.	97
Figure 4.5 – Simulated (blue line) and observed (red marker) water temperature at (a) 0.5 m depth, and at (b) 15 m depth.	98
Figure 4.6 – Contour plot of simulated water temperature in Serra Azul reservoir.	98
Figure 4.7 – Water temperature difference between surface (depth 0 m) and bottom (20 m depth) in Serra Azul reservoir: a) daily values and b) annual values. The horizontal line inside the box plot indicates the median, the cross marker indicates the mean, the boundaries of the box plot indicate the 25 th and 75 th percentiles and the circle indicates outliers. Whiskers indicate the maximum and minimum value.	100
Figure 4.8 – Simulated Schmidt stability of Serra Azul reservoir.	100
Figure 4.9 – Simulated hydrodynamics indicators of Serra Azul reservoir: a) Schmidt stability, b) Lake number, and c) Wedderburn number. The horizontal line inside the box plot indicates the median, the cross marker indicates the mean and the boundaries of the box plot indicate the 25 th and 75 th percentiles. Whiskers indicate the maximum and minimum value.	101
 Figure 5.1 – Serra Azul reservoir, its tributaries and catchment.	113
Figure 5.2 – Serra Azul volume from 2014 to 2018 in percentage of its total storage capacity ($8.16 \times 10^7 \text{m}^3$) (COPASA, 2018).	114

Figure 5.3 – Boxplots of monitored physico-chemical variables of Serra Azul reservoir for the period from 2009 to 2013 and the period from 2014 to 2016 at different depths: a) Secchi depth; b) turbidity; c) total solids concentration; d) total phosphorus concentration; e) ammonia concentration; f) nitrate concentration; g) dissolved oxygen concentration; h) pH and i) conductivity. The horizontal line inside the boxplot indicates the median, the cross marker indicates de mean, the boundaries of the box plot indicate the 25 th and 75 th percentiles and the circle indicates the outliers. Whiskers indicate the maximum and minimum value. Source of data: COPASA, 2017.	115
Figure 5.4 – Relative biomass of each phytoplankton class. Source of data: COPASA, 2017.	116
Figure 5.5 – Boxplots of monitored phytoplankton classes of Serra Azul reservoir from 2009 to 2013 and from 2014 to 2016: a) cyanobacteria, b) chlorophyceae, c) chrysophyceae and d) pyrrophyceae. The horizontal line inside the box plot indicates the median, and the boundaries of the box plot indicate the 25 th and 75 th percentiles. Whiskers indicate the maximum and minimum value. Souce of data: COPASA, 2017.	117
Figura 1A.1 – Corpo d'água, torre de tomada d'água, barragem, vertedouro de soleira livre e rebaixamento do nível d'água no reservatório Serra Azul.	125
Figura 1A.2 – Tributários do reservatório Serra Azul: (a) Curralinho; (b) Estiva; (c) Diogo; (d) Ribeirão Serra Azul; (e) Jacu; e (f) Pedreira.	126
Figura 1A.3 – Aproveitamento agrícola na bacia hidrográfica do reservatório Serra Azul.	126

LISTA DE TABELAS

Tabela 1.1 – Equações empregadas no modelo GLM (Hipsey <i>et al.</i> , 2014).	22
Tabela 1.2 – Parâmetros físicos empregados no modelo GLM (Hipsey <i>et al.</i> , 2014).	23
Tabela 1.3 – Equações do modelo AED (Hipsey <i>et al.</i> , 2013).	28
Tabela 1.4 – Parâmetros biogeoquímicos empregados no modelo AED (Hipsey <i>et al.</i>, 2013).	30
Tabela 1.5 – Parâmetros do fitoplâncton empregados no modelo AED para cianobactérias e clorofíceas (Hipsey <i>et al.</i> , 2013).	32
Tabela 1.6 – Estações de monitoramento consultadas.	36
Tabela 1.7 – Valores médios adotados para as concentrações de fósforo, oxigênio dissolvido e nitrato na estação 371.	41
Table 2.1 – Water body classification according to the trophic state index (Lamparelli, 2004).	51
Table 2.2 – Statistical metrics of the limnological variables. Source of data: COPASA, 2017.	56
Table 3.1 – Monitoring stations.	67
Table 3.2 – Sensitivity coefficients, calibrated parameters and range values of the hydrodynamic model.	75
Table 3.3 – Sensitivity coefficient and range values of the ecological model.	76
Table 3.4 – RMSE reported for different lakes using 1D models.	79
Table 3.5 – Thermal characteristics of the hydrodynamics in Serra Azul reservoir.	82
Table 4.1 – Monitoring stations characteristics.	92
Table 4.2 – Annual mean values (standard deviation) of meteorological (Florestal - INMET) and hydrological (Serra Azul - CPRM) variables from 2009 to 2016.	93
Table 4.3 – Calibrated parameters and their range values.	95
Table 4.4 – Simulated hydrodynamics indicators of Serra Azul reservoir.	99

LISTA DE ABREVIATURAS, SIGLAS E SÍMBOLOS

1D-V	Unidimensional vertical
AED	<i>Aquatic EcoDynamics</i>
ANA	Agência Nacional de Águas
CAPES	Coordenação de Aperfeiçoamento de Pessoal de Nível Superior
CH ₄	Metano
CID	Carbono inorgânico dissolvido
Chl	Chlorofila- <i>a</i>
COD	Carbono orgânico dissolvido
CONAMA	Conselho Nacional de Meio Ambiente
COP	Carbono orgânico particulado
COPASA	Companhia de Saneamento de Minas Gerais
CPRM	Serviço Geológico do Brasil
GLM	<i>General Lake Model</i>
INMET	Instituto Nacional de Meteorologia
MG	Minas Gerais
n	Número de observações
N	Nitrogênio
N ²	Frequência de Brunt-Väisälä
NH ₃	Amônia
NH ₄	Amônio
N _L , L _N	Número do lago
NO ₃	Nitrato
NOD	Nitrogênio orgânico dissolvido
NOP	Nitrogênio orgânico particulado
OD, DO	Oxigênio dissolvido
P	Fósforo
PHY	Biomassa do fitoplâncton
PO ₄	Fosfato
POD	Fósforo orgânico dissolvido
POP	Fósforo orgânico particulado
Q	Vazão
r	Coefficiente de correlação de Pearson
R ²	Coefficiente de determinação
Ri	Número de Richardson
RMBH, BHMR	Região Metropolitana de Belo Horizonte
RMSE	Raiz do Erro Quadrático Médio
S _c	Coefficiente de Sensibilidade
SD	Desvio padrão
S _T	Índice de estabilidade de Schmidt
TP	Fósforo total
TS	Sólidos totais
TSI	Índice de estado trófico
UFMG	Universidade Federal de Minas Gerais
UnB	Universidade de Brasília
UNESCO	Organização das Nações Unidas para a Educação, a Ciência e a Cultura
USP	Universidade de São Paulo
W _d	Número de Wedderburn

CAPÍTULO 1

Apresentação e Contextualização

1 INTRODUÇÃO

Lagos e reservatórios proveem serviços ecossistêmicos, como a retenção e disponibilização de nutrientes e sedimentos, a regulação de processos hidrológicos e são hábitat para comunidades aquáticas (Schallenberg *et al.*, 2013). Esses corpos d'água lênticos também dão suporte a atividades antrópicas, a partir da reserva de água para abastecimento público, irrigação de culturas agrícolas, geração de energia hidroelétrica, abastecimento de água para consumo industrial, navegação, recreação e prática de esportes (Silva *et al.*, 2016).

A temperatura da água constitui-se em uma característica física determinante para os processos que ocorrem em ambientes lênticos, uma vez que: está relacionada à estratificação térmica e mistura da coluna d'água; influencia a concentração de oxigênio dissolvido; influencia as taxas metabólicas e reprodutivas em toda a cadeia alimentar; e controla os processos de reaeração, volatilização e absorção de substâncias (Ji, 2007). Em sistemas lacustres, os processos térmicos que ocorrem na coluna d'água afetam consideravelmente a distribuição vertical de nutrientes, contaminantes, oxigênio dissolvido, sólidos suspensos e do fitoplâncton (Tundisi e Tundisi, 2008). A dinâmica térmica é, em grande parte, responsável pelo gradiente vertical de qualidade da água.

O fitoplâncton constitui a base da cadeia trófica em vários corpos d'água lênticos e sob determinadas condições ambientais, pode crescer intensivamente e em curto intervalo de tempo, originando florações. As florações podem causar impactos nos ecossistemas aquáticos, como a depleção de oxigênio dissolvido devido à respiração pelos próprios organismos ou durante a decomposição das suas células pelas bactérias, que consomem o oxigênio da água. O problema se agrava quando as florações são tóxicas, causando sintomas negativos aos organismos, inclusive à saúde humana.

Lagos e reservatórios são especialmente sensíveis às condições ambientais (Adrian *et al.*, 2009). As principais variáveis atuantes sobre os processos hidrodinâmicos e ecológicos são a profundidade e morfometria do lago, as vazões afluentes e defluentes e as condições meteorológicas atuantes na superfície d'água (Peeters *et al.*, 2002). Alterações nas condições meteorológicas locais, como o aumento da temperatura do ar e a intensificação ou escassez de eventos chuvosos, são capazes de impactar direta e quase imediatamente a estrutura térmica e

a dinâmica ecológica em lagos e reservatórios devido à interconexão entre a temperatura do ar e a temperatura das águas superficiais (Livingstone e Lotter, 1998).

Períodos de escassez hídrica podem afetar os processos hidrodinâmicos e o balanço energético em lagos e reservatórios (Zohary e Ostrovsky, 2011). Os efeitos são refletidos em mudanças nas características físicas, químicas e biológicas, levando a alterações na qualidade da água (Naselli-Flores e Barone, 2005). A biomassa e a composição das espécies do fitoplâncton são afetadas devido aos impactos sobre a estrutura térmica e os processos biogeoquímicos (Fadel *et al.*, 2017). Mudanças nas variáveis hidrológicas e meteorológicas podem causar maior estresse em sistemas de recursos hídricos já afetados pelo crescimento populacional, aumento dos padrões de consumo e má gestão (Raje e Mujumdar, 2010).

Os processos atuantes em corpos d'água lênticos podem ser estudados de maneira integrada acoplando-se modelos hidrodinâmicos e ecológicos, que permitem simular as interrelações entre a estrutura térmica da coluna d'água e a dinâmica dos ecossistemas aquáticos, bem como seus reflexos sobre a qualidade da água. Considerando-se que a degradação da qualidade da água pode comprometer seus usos e inviabilizar o fornecimento de água potável, a compreensão das dinâmicas físicas e ecológicas é um requisito para a avaliação da vulnerabilidade do corpo hídrico e para a implementação de medidas de gestão que antecipem a mitigação de possíveis impactos causados por eventos extremos. O emprego da modelagem torna-se especialmente relevante em ambientes de clima tropical, nos quais lagos e reservatórios são mais sensíveis às variabilidades climáticas e os organismos aquáticos apresentam taxas metabólicas mais aceleradas (Lewis, 2000). Quando se trata dos impactos da escassez hídrica sobre a dinâmica de lagos e reservatórios, sobretudo daqueles localizados em ambientes tropicais, a literatura existente é particularmente escassa (Zohary e Ostrovsky, 2011).

Esta dissertação de mestrado se interessou pela aplicação de modelagem hidrodinâmica e ecológica como ferramenta para auxiliar a gestão de recursos hídricos, com foco em um reservatório metropolitano estratégico para o abastecimento de água. Pretende-se fornecer subsídios que auxiliem a manutenção da qualidade da água, conforme disposto na Lei Federal n. 9433/1997, que institui a Política Nacional de Recursos Hídricos, buscando assegurar disponibilidade hídrica com padrões de qualidade compatíveis com seus usos atuais e futuros.

O presente estudo é desenvolvido no âmbito do projeto de pesquisa “Modelagem de mananciais metropolitanos estratégicos como insumo para a gestão de água e território em face à mudança climática – MoMa-SE”, financiado pela Agência Nacional de Águas (ANA) e pela Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES). O projeto MoMa-SE iniciou-se em 2016, tem término previsto para 2020 e é desenvolvido conjuntamente pela Universidade Federal de Minas Gerais (UFMG), Universidade de São Paulo (USP) e Universidade de Brasília (UnB). A hipótese central do projeto baseia-se na possibilidade de aumentar a resiliência de mananciais estratégicos a pressões antrópicas e a mudanças climáticas. Busca-se tornar claras as ameaças à segurança hídrica em regiões metropolitanas.

Visando expor os métodos empregados e os resultados alcançados de maneira mais compartimentada e organizada, a dissertação foi estruturada em cinco capítulos: o Capítulo 1 apresenta, além dessa introdução, os objetivos dessa pesquisa; a revisão da literatura, por meio do referencial teórico que sustenta o tema em estudo; e os materiais empregados, abordando a área de estudo, as estações de monitoramento consultadas e o tratamento dos dados. O Capítulo 2 apresenta, na forma de artigo, a caracterização da área de estudo baseada em série histórica de trinta anos de dados monitorados. O Capítulo 3 apresenta, na forma de artigo, a modelagem matemática hidrodinâmica e ecológica do reservatório Serra Azul. O Capítulo 4 apresenta, na forma de artigo, a simulação dos impactos da escassez hídrica sobre os processos hidrodinâmicos e a estrutura térmica da coluna d’água. O Capítulo 5 apresenta, também na forma de artigo, os efeitos da escassez hídrica sobre a dinâmica fitoplanctônica do reservatório, abordando sua composição e biomassa. O Capítulo 6 apresenta as conclusões alcançadas nesse estudo. Os elementos pós-textuais são o Apêndice A, que apresenta o registro fotográfico de visitas em campo na área de estudo e o Apêndice B, que apresenta o *script* utilizado no R para rodar o modelo GLM e analisar os dados de saída.

2 OBJETIVOS

2.1 *Objetivo geral*

O objetivo geral desta pesquisa é avaliar os impactos do período de escassez hídrica sobre a estrutura térmica e a dinâmica fitoplanctônica do reservatório Serra Azul (MG), estratégico para o abastecimento de água potável, a partir de modelagem matemática e dados de campo.

2.2 Objetivos específicos

- Caracterizar os processos térmicos e a dinâmica do fitoplâncton do reservatório Serra Azul;
- Identificar os parâmetros e as variáveis meteorológicas e hidrológicas mais influentes na estrutura térmica do reservatório;
- Avaliar os impactos da escassez hídrica sobre os processos de estratificação térmica e mistura da coluna d'água;
- Avaliar os efeitos da escassez hídrica sobre a composição e a biomassa do fitoplâncton.

3 REVISÃO DA LITERATURA

3.1 Processos hidrodinâmicos em sistemas lacustres

A água possui elevado calor específico ($1 \text{ cal g}^{-1} \text{ } ^\circ\text{C}^{-1}$) proporcionando ambientes térmicos excepcionalmente estáveis em sistemas lacustres, uma vez que a coluna d'água se aquece ou esfria lentamente. A incidência de ondas curtas sobre lagos e reservatórios é parcialmente absorvida na camada superficial e convertida em calor. Essa energia se propaga por condução lentamente de molécula a molécula ou por convecção devido, em grande parte, à ação do vento. A temperatura da água se eleva mais rapidamente na superfície em relação ao fundo, promovendo a estratificação térmica da coluna d'água. A estabilidade dessa estratificação depende de muitos fatores, entre eles profundidade, morfometria, condições meteorológicas, magnitude e direção do vento e vazões afluentes e defluentes ao corpo d'água. A estratificação é um importante fenômeno nos sistemas aquáticos continentais e grande parte dos processos e mecanismos de funcionamento resulta do gradiente vertical formado (Tundisi e Tundisi, 2008).

O desenvolvimento da estratificação térmica em lagos e reservatórios ocorre devido a três fatores principais (Imboden e Wüest, 1995): primeiramente, devido à baixa velocidade da água, ocorrendo um mínimo de agitação entre as camadas do fluido, o que favorece um gradiente estável de temperatura; em segundo lugar, sistemas lacustres apresentam longos tempos de residência da água. Em consequência, os processos físicos de aquecimento e resfriamento e os processos químicos tornam-se lentos; finalmente, lagos e reservatórios apresentam, na maioria dos casos, topografia encaixada, formando depressões internas que

reduzem a interação entre a superfície e o fundo. Dessa forma, a estratificação térmica se torna mais estável à medida que o fluxo d'água torna-se mais laminar, o tempo de residência da água torna-se mais longo e o corpo d'água torna-se mais profundo.

A estratificação térmica leva à formação de três camadas (Figura 1.1a) (Tundisi e Tundisi, 2008): (i) epilímnio: camada superficial caracterizada por maior temperatura e circulação interna; (ii) metalímnio: região de transição, a qual apresenta uma inflexão em seu perfil vertical, a termoclina, que possui o maior gradiente vertical de temperatura da água; e (iii) hipolímnio: camada mais profunda caracterizada por menor temperatura, relativa estabilidade e geralmente com ausência de luz. Essa camada não interage ativamente com a atmosfera e, em alguns casos, pode apresentar ausência de oxigênio dissolvido.

Em ambientes tropicais, o resfriamento da camada superficial durante o inverno e a ação de ventos podem romper a estrutura de camadas (Figura 1.1b). À medida que a temperatura da camada superficial reduz, sua densidade aumenta e a camada se desloca para baixo. Como resultado, as águas de superfície e de fundo misturam-se livremente, homogeneizando a temperatura ao longo da coluna d'água. O vento é responsável por ondas de superfície e, acima de determinada velocidade, por ondas internas, sendo uma importante fonte de energia cinética para promover a mistura da coluna d'água. As vazões afluentes ao corpo d'água também contribuem para a mistura através de sua própria energia cinética e da formação de correntes internas. Outros agentes são a emissão de ondas longas, a precipitação, a evaporação e o resfriamento da temperatura do ar, que promovem o resfriamento da superfície, podendo levar à instabilidade do sistema (Ji, 2007).

A estratificação e a mistura são acompanhadas por outras alterações físicas e químicas na água, pois exercem implicações importantes sobre a disponibilidade de nutrientes (Komatsu *et al.*, 2006), de material dissolvido e particulado (Robertson e Imberger, 1994), sobre a liberação de ferro e manganês (Tundisi e Tundisi, 2008), bem como sobre a distribuição vertical, migração e nutrição do fitoplâncton (Vidal *et al.*, 2010) e de níveis tróficos superiores, como zooplâncton e peixes (Read *et al.*, 2011). Logo após a estratificação ter sido estabelecida, o hipolimínio encontra-se rico em oxigênio dissolvido. Com o passar do tempo, ocorre uma diminuição gradual pelo consumo biológico. Além disso, a estratificação frequentemente resulta em um hipolimínio rico em nutrientes, mas limitado por luz, em contraste com um epilimínio rico em luz, mas pobre em nutrientes. Quando a mistura ocorre,

os nutrientes que estavam depositados no hipolímnio tornam-se disponíveis para os organismos presentes no restante da coluna d'água.

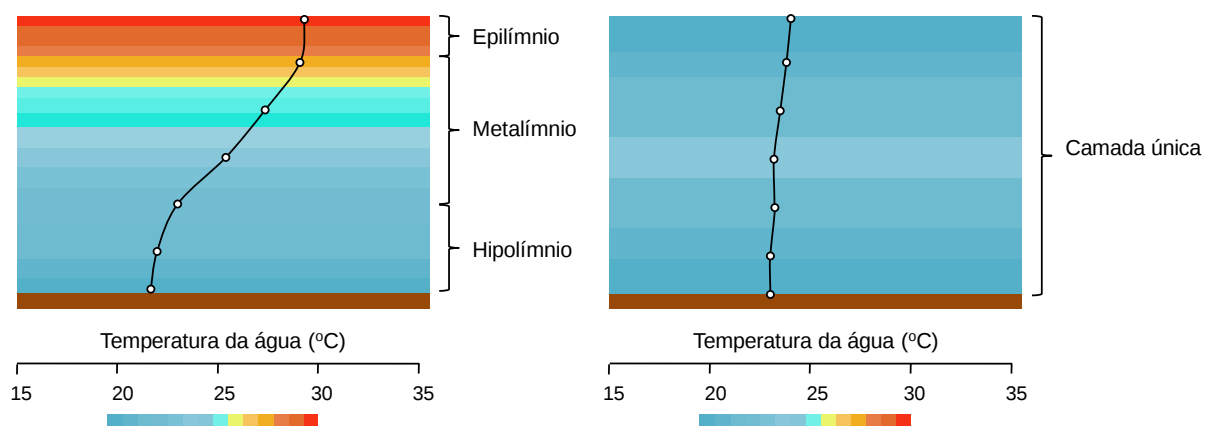


Figura 1.1 – Dinâmica de estratificação térmica e mistura em sistemas lacustres. Perfis de um lago a) em condições de estratificação e b) de mistura.

Os ambientes lacustres podem ser classificados de acordo com seu padrão térmico vertical (Tundisi e Tundisi, 2008): (1) lagos monomíticos apresentam um período regular de circulação total que ocorre em alguma época do ano. Existem dois tipos básicos de lagos monomíticos: os quentes, que apresentam uma circulação anual de outono/inverno, e os frios, que apresentam uma circulação anual de verão; (2) lagos dimíticos apresentam dois períodos anuais de circulação em toda a coluna d'água; (3) lagos polimíticos apresentam muitos períodos anuais de circulação; (4) lagos meromíticos nunca apresentam circulação que alcança toda a coluna d'água; e (5) lagos holomíticos apresentam circulação completa e não têm estratificação ocasional. Lagos tropicais profundos, de maneira geral, são monomíticos quente, apresentando apenas um período de mistura total da coluna d'água no outono/inverno (Lewis, 2000).

3.1.1 Indicadores do comportamento hidrodinâmico de sistemas lacustres

Um dos mais importantes indicadores do comportamento hidrodinâmico de sistemas lacustres é o tempo de residência da água (τ), também denominado tempo de retenção ou tempo de detenção hidráulica. A influência do tempo de residência da água na estabilidade vertical de reservatórios foi primeiramente demonstrada por Straskraba (1999). Este indicador é calculado pela Equação 1.1 e representa o tempo médio necessário para o escoamento completo da água do lago ou reservatório:

$$\tau = \frac{V}{Q} \quad (1.1)$$

na qual V é o volume e Q é a vazão do corpo d'água. O tempo de residência da água pode variar de dias em pequenos represamentos a anos em grandes lagos (Ji, 2007). O tempo de residência influencia os padrões longitudinais e verticais e permite prever *a priori* a ocorrência de estratificação térmica (Straskraba, 1999): curtos tempos de residência estão relacionados a instabilidade da coluna d'água e maior susceptibilidade à mistura; por outro lado, tempos de residência mais longos favorecem a estabilidade térmica e a ocorrência da estratificação.

Indicadores físicos relacionam as forças estabilizadoras e desestabilizadoras atuantes sobre um corpo d'água e podem auxiliar na identificação e caracterização de períodos de estratificação e mistura da coluna d'água. As forças desestabilizadoras do sistema são a ação do vento, o resfriamento da temperatura do ar e vazões afluentes e defluentes. Dentre os principais indicadores físicos, pode-se citar a estabilidade de Schmidt, o número de Wedderburn, o número do Lago, a frequência de Brunt-Väisälä e o número de Richardson (Read *et al.*, 2011).

O índice de estabilidade de Schmidt (S_T) foi elaborado por Schmidt (1928) e posteriormente formalizado por Idso (1973). É um indicador do processo de estratificação, indicando a resistência à mistura devido à energia potencial. Representa o trabalho por unidade superficial de área necessário para romper a estratificação térmica e tornar a coluna d'água homogênea em termos de densidade. A estabilidade de Schmidt é calculada segundo Equação 1.2 (Idso, 1973):

$$S_T = \frac{g}{A_s} \times \int_0^{z_D} (z - z_v) \times \rho_z \times A_z \times \partial z \quad (1.2)$$

na qual g é a aceleração da gravidade (ms^{-2}), A_s é a área superficial do reservatório (m^2), z_D é a profundidade máxima do reservatório (m), z é a profundidade da camada (m), z_v é a profundidade do centro volumétrico do reservatório (m), ρ_z é a massa específica da água na profundidade z (kgm^{-3}), A_z é a área do reservatório na profundidade z (m^2) e ∂z é a variação da profundidade (m).

O número de Wedderburn (W_d) foi desenvolvido por Thompson e Imberger (1980) e descreve a possibilidade de ressurgência de águas do metalímnio. Se $W_d > 1$, a estratificação da coluna d'água se intensifica; por outro lado, se $W_d \leq 1$, o epilímnio sofre a ação turbulenta do vento e o processo de mistura é favorecido. O número de Wedderburn é calculado por

$$W_d = \frac{g' \times z_e^2}{u_*^2 \times L_s} \quad (1.3)$$

sendo

$$g' = g \times \frac{\Delta\rho}{\rho_h} \quad (1.4)$$

na qual g' é a gravidade reduzida (ms^{-2}), z_e é a profundidade do limite superior do metalímnio (m), u^* é a tensão de cisalhamento na superfície da água devido ao vento (ms^{-1}), L_s é o comprimento do lago na direção do vento (m), g é a aceleração da gravidade (ms^{-2}), $\Delta\rho$ é a variação da massa específica da água entre o hipolímnio e o epilímnio (kgm^{-3}) e ρ_h é a massa específica da água no hipolímnio (kgm^{-3}).

O número do lago (N_L) foi definido por Imberger e Patterson (1990) e descreve processos relevantes para a mistura da coluna d'água induzida pela ação do vento, indicando o potencial de ressurgência de águas do hipolímnio (Macintyre e Melack, 2009). Valores inferiores a 1 indicam estratificação fraca e elevado potencial de ocorrência de mistura devido à ação de ondas internas; valores superiores a 1 indicam forte estratificação, a qual supera as forças introduzidas pelo vento na superfície. Sob condições de estratificação térmica, quando $W_d \leq 1$ e $N_L > 1$ apenas o epilímnio encontra-se misturado; quando $W_d \leq 1$ e $N_L < 1$, a mistura ocorre em toda a profundidade do corpo d'água, inclusive em águas profundas (Robertson e Imberger, 1994). O número do lago é dado por

$$N_L = \frac{S_T \times (z_e + z_h)}{2 \times \rho_h \times u_*^2 \times \sqrt{A_s \times z_v}} \quad (1.5)$$

na qual S é o índice de estabilidade de Schmidt (Jm^{-2}), z_e é a profundidade do limite superior do metalímnio (m), z_h é a profundidade do limite inferior do metalímnio (m), ρ_h é a massa específica da água no hipolímnio (kgm^{-3}), u^* é a tensão de cisalhamento na superfície da água devido ao vento (ms^{-1}), A_s é a área superficial do reservatório (m^2) e z_v é a profundidade do centro volumétrico do reservatório (m).

A frequência de Brunt-Väisälä (N^2) representa a intensidade da estratificação, baseada no gradiente de densidade da água apenas, e permite determinar as oscilações periódicas das camadas líquidas sem considerar a ação do vento. É um parâmetro útil no estudo da diversidade do fitoplâncton em função da oscilação vertical do sistema. A frequência de Brunt-Väisälä é dada pela seguinte equação

$$N^2 = \frac{g}{\rho} \frac{\partial \rho}{\partial z} \quad (1.6)$$

na qual g é a aceleração da gravidade (ms^{-2}), ρ é a massa específica da água de referência (1000 kgm^{-3}) e $\frac{\partial \rho}{\partial z}$ é o gradiente de densidade em função da profundidade ($\text{kgm}^{-3}\text{m}^{-1}$).

O número de Richardson (Ri) quantifica a estabilidade térmica a partir da relação entre a geração de turbulência devido ao empuxo e a geração de turbulência devido ao vento. Quando $Ri > 0,25$, a estratificação é considerada estável e as camadas de água apresentam um fluxo sem turbulência; se $Ri < 0,25$, a estratificação é rompida e o perfil se torna instável, ocorrendo mistura turbulenta. O número de Richardson é calculado por

$$Ri = \frac{g}{\rho} \frac{\partial \rho}{\partial z} \frac{\partial u^2}{\partial z} \quad (1.7)$$

na qual g é a aceleração da gravidade (ms^{-2}), ρ é a massa específica da água (kgm^{-3}), $\frac{\partial \rho}{\partial z}$ é o gradiente de densidade em função da profundidade ($\text{kgm}^{-3}\text{m}^{-1}$) e u é a velocidade média da água (ms^{-1}).

A aplicação dos indicadores do comportamento hidrodinâmico de sistemas lacustres permite uma rápida caracterização do estado do reservatório quanto à possibilidade de estratificação e permite responder, entre outras questões, o quão influente é a ação do vento sobre o corpo d'água. Os indicadores são simples de serem determinados e por isso são amplamente utilizados no estudo hidrodinâmico de lagos e reservatórios (*e.g.* Brighenti, 2014; Shintani *et al.*, 2010; Read *et al.*, 2011). No entanto, ainda persistem algumas nuances sobre a definição correta de alguns conceitos, como a extensão da camada de mistura ou a profundidade da termoclina, dificultando a comparação de resultados entre áreas de estudo (Read *et al.*, 2011).

3.2 Dinâmica do fitoplâncton em sistemas lacustres

O fitoplâncton compreende o conjunto de organismos que realizam fotossíntese e não dispõem de movimentos próprios capazes de opor aos movimentos da água. Esses organismos estão sujeitos aos seguintes processos biológicos: (a) crescimento, o qual é uma função da taxa máxima de crescimento, temperatura da água, horas de luz no dia, intensidade da luz na superfície da água, condições da luz na água e concentração de fósforo e nitrogênio no meio; (b) assimilação de nutrientes, sendo que nitrogênio e fósforo são necessários às reações químicas celulares e ao armazenamento e liberação de energia; (c) respiração, para obtenção da energia requerida para manutenção vital; (d) excreção de nutrientes; (e) sedimentação e ressuspensão, os quais podem ocorrer devido a fatores externos, como a circulação da coluna d'água, ou de algum mecanismo adaptativo; (f) mortalidade natural, devido à ausência de condições ambientais adequadas ou ao completar o ciclo de vida do organismo; e (g) consumo por herbívoros, como peixes, zooplâncton e bentos.

Os fenômenos hidrodinâmicos influenciam a densidade e a distribuição do fitoplâncton (Fragoso Jr. *et al.*, 2009). A relação entre a ocorrência do fitoplâncton com o período de estratificação térmica da coluna d'água é evidenciada em estudos (*e.g.* Figueredo e Giani, 2009; Yang *et al.*, 2016; Curtarelli *et al.*, 2014; Gagliardi, 2015), o que pode ser devido à maior estabilidade e transparência da coluna d'água (Gomes, 2008).

A composição da comunidade fitoplanctônica se relaciona com o funcionamento dos ecossistemas aquáticos e os filos de maior ocorrência em água doce são baccillariophyta, chlorophyta, chryptophyta, chrysophyta, cianobactéria, dinophyta e euglenopyta (Tundisi e Tundisi, 2008). Dentre esses, deve-se destacar como de grande interesse ambiental, sanitário e econômico, as cianobactérias. Sua floração é uma preocupação para a gestão de mananciais de água potável, uma vez que esses organismos podem levar à interrupção do abastecimento pela deterioração da qualidade da água através de: redução da concentração de oxigênio dissolvido no hipólímnio; maus odores e produção de espuma; e obstrução de canalização e danos em equipamentos hidráulicos (Smith, 2003). Ademais, as florações de cianobactérias estão relacionadas à produção de toxinas (hepatotoxinas, neurotoxinas, citotoxinas, dermatotoxinas, anatoxinas e endotoxinas) (*e.g.* Fadel *et al.*, 2014), as quais podem prejudicar o trato digestivo e o fígado, causar sintomas de pneumonia e afetar o sistema nervoso central (Smith, 2003).

3.3 Efeitos da escassez hídrica sobre sistemas lacustres

Períodos de escassez hídrica são fenômenos naturais caracterizados por uma diminuição considerável da disponibilidade de água e sua ocorrência pode variar em termos de duração, intensidade e frequência. O déficit hídrico se propaga através do ciclo hidrológico e reduz os escoamentos subterrâneos e superficiais, resultando na redução do nível d'água em lagos e reservatórios. Períodos de escassez hídrica estão se tornando mais frequentes e severos em muitas regiões do mundo devido ao aumento da variabilidade do regime pluviométrico (Trenberth *et al.*, 2003).

No Brasil, estiagens e secas observadas desde 2012 impactaram importantes reservatórios no país e têm prejudicado a oferta de água para abastecimento público, irrigação, geração de energia hidrelétrica e navegação. Para citar alguns exemplos, a vazão média defluente dos reservatórios de Sobradinho (BA) e Xingó (AL/SE) reduziram de $1300 \text{ m}^3\text{s}^{-1}$ em 2013 para $550 \text{ m}^3\text{s}^{-1}$ em 2017; em fevereiro de 2015, o reservatório equivalente do sistema Paraíba do Sul (RJ) atingiu o volume útil de 0,33%, o menor valor observado em todo o histórico; a vazão média no sistema Cantareira (SP) em 2014 foi $8,70 \text{ m}^3\text{s}^{-1}$, menor valor desde 1930, correspondente a cerca de 22% da média histórica; e o reservatório Descoberto (DF) atingiu menos de 20% de seu volume em 2016 (ANA, 2017).

O reservatório Serra Azul (MG) vem enfrentando um período de escassez hídrica desde 2014. Em quase todas as bacias hidrográficas da sua região, os ciclos hidrológicos 2013/2014, 2014/2015 e 2015/2016 registraram as menores precipitações e vazões da série histórica em 70 anos de monitoramento (CPRM, 2017). O ano de 2014 foi o mais seco, com um período de retorno superior a 100 anos (ANA, 2014). A condição de escassez hídrica permaneceu no ano hidrológico de 2016/2017, em que a precipitação registrada foi menor do que a média histórica. O reservatório Serra Azul foi o mais impactado da Região Metropolitana de Belo Horizonte e seu volume atingiu 5,2% de sua capacidade total em novembro de 2014 (Figura 1.2). Seu volume vem se recuperando, mas sua condição prévia a 2014 ainda não foi restabelecida.

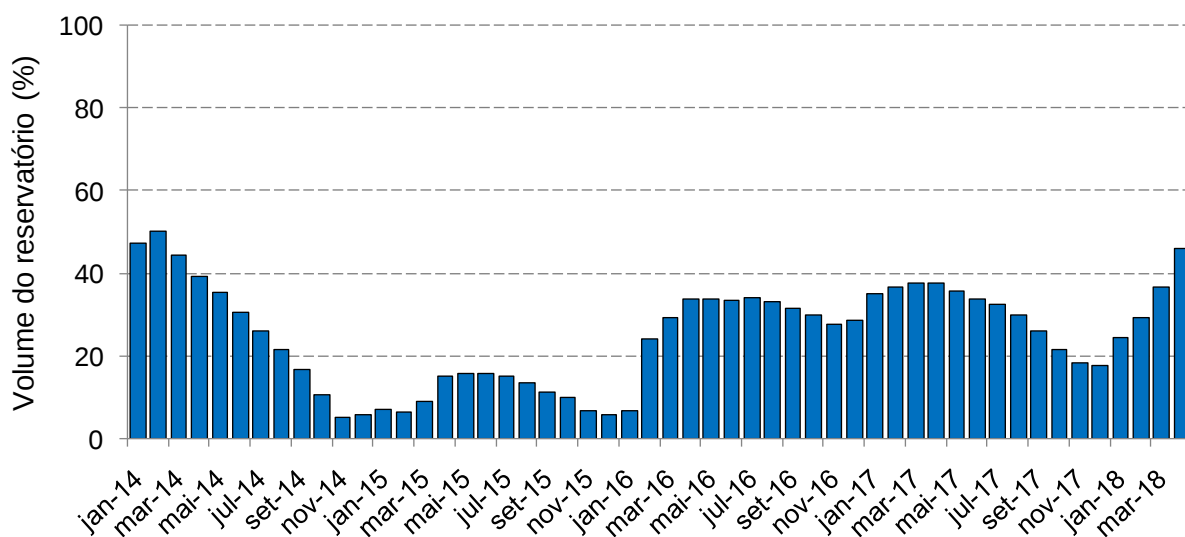


Figura 1.2 – Volume mensal do reservatório Serra Azul a partir de 2014. Fonte: COPASA, 2018.

Estudos indicam que a escassez hídrica afeta a qualidade da água em sistemas lacustres alterando mecanismos internos, como a fotossíntese e o balanço energético (Mosley, 2015; Li *et al.*, 2017; Wedderburn *et al.*, 2012). Foram verificados impactos sobre os processos hidrodinâmicos e a estrutura térmica da coluna d'água (Zohary e Ostrovsky, 2011), aumento do tempo de residência da água (Mosley, 2015), aumento substancial da salinidade, condutividade e concentração de fósforo (Kingsford *et al.*, 2011) e redução da transparência da água (Jeppesen *et al.*, 2015). Ademais, Fadel *et al.* (2017) verificaram alterações na biomassa e composição do fitoplâncton.

Os impactos da escassez hídrica podem ser bastante variáveis e específicos, dependendo das características biofísicas do corpo d'água e de sua bacia hidrográfica. De acordo com Jeppesen *et al.* (2015), os efeitos da escassez hídrica variam conforme o clima local e a extensão dos efeitos sobre o ciclo hidrológico.

3.4 Modelagem matemática de sistemas lacustres

A modelagem matemática tem se firmado como ferramenta para prognóstico dos fenômenos que atuam sobre determinados sistemas e como esses reagem e se comportam. A modelagem ecológica associada aos ambientes aquáticos surgiu nos anos 1920 com os modelos de Streeter-Phelps para oxigênio dissolvido e demanda bioquímica de oxigênio em rios e sofreu avanços com o modelo presa-predador de Lotka-Volterra (Jorgensen *et al.*, 2009). Na década de 1970, foram desenvolvidos modelos de lagos que descrevem o processo de eutrofização

visando auxiliar os sistemas de suporte à decisão e, durante os anos 1980 e 1990, os modelos foram aplicados para gestão ambiental (Jorgensen, 2010).

O progresso recente na área de hidrologia, no sentido de avaliar e prognosticar com mais precisão os processos quantitativos, impulsionou a construção de modelos matemáticos eco-hidrológicos, os quais integram a hidrodinâmica com as variáveis biológicas e de qualidade da água, melhorando, assim, a capacidade preditiva de constituintes químicos e processos biológicos. Nos últimos anos, os modelos ecológicos acoplados a modelos hidrodinâmicos cresceram em complexidade e em processos retratados (Dokulil, 2013). Atualmente, os modelos vêm sendo empregados para fins de previsão de cenários futuros (Hetherington *et al.*, 2015), entendimento dos processos (Bueche e Vetter, 2014), preenchimento de variáveis de interesse em um período sem levantamento de dados (Read *et al.*, 2014) e geração de hipóteses, as quais podem ser testadas experimentalmente ou *in situ* (Gaeta *et al.*, 2014).

Mooij *et al.* (2010) apontaram três perspectivas para a modelagem ecológica: a melhoria na interface do modelo com o usuário para favorecer sua aplicabilidade pelos gestores de recursos hídricos; o desenvolvimento de modelos que abranjam uma ampla gama de características de lagos, incluindo tamanho e morfologia, condições climáticas e características hidrológicas, para sua aplicação em projeções de escala global; e o aprofundamento na representação da biodiversidade aquática pois o nível de detalhamento ainda está longe do observado em sistemas lacustres.

Os modelos de simulação de sistemas lacustres podem ser classificados com relação ao número de dimensões espaciais (Fragoso Jr, 2008): a) modelos de balanço ou de dimensão zero: não têm compromisso de representar os gradientes espaciais das variáveis hidrodinâmicas, químicas e biológicas. Este tipo de modelo é aplicado em estudos de balanço de massa simplificados que servem para uma avaliação preliminar das condições de armazenamento e concentração de poluentes em reservatórios; b) modelos unidimensionais: consideram os gradientes espaciais em uma direção, geralmente na direção vertical ou longitudinal; c) modelos bidimensionais na vertical integrados lateralmente: simulam os fluxos e concentrações nas dimensões vertical e longitudinal, desprezando as variações na transversal. Esta simplificação é aceitável em reservatórios bem encaixados no sentido longitudinal do rio e com grandes profundidades; d) modelos bidimensionais na horizontal integrados verticalmente: simulam os fluxos e concentrações nas dimensões longitudinal e

transversal. Eles permitem uma visualização das velocidades e concentrações no plano e desprezam as variações verticais no perfil. Este modelo pode ser útil quando existe uma significativa variação longitudinal das concentrações e das velocidades e; e) modelos tridimensionais: são os modelos mais adequados conceitualmente para a simulação do escoamento e da qualidade da água, porém apresentam dificuldades práticas relacionadas à demanda de tempo computacional e ao número de parâmetros para controlar.

Considerando-se que a estrutura térmica de sistemas lacustres é dominada pelos gradientes de temperatura na vertical, em muitos casos se justifica a desconsideração dos gradientes horizontais e o problema pode ser abordado usando um modelo unidimensional vertical. Muitos estudos adotaram modelos unidimensionais para simular o comportamento dos processos hidrodinâmicos e ecológicos em lagos e reservatórios (Read *et al.*, 2014; Vinçon-Leite *et al.*, 2014; Rigosi e Rueda, 2012; Weinberger e Vetter, 2012; Silva *et al.*, 2016; Taner *et al.*, 2011). A suposição fundamental desses modelos é que os gradientes horizontais são muito menores que os gradientes verticais, isto é, suas propriedades são lateralmente homogêneas. Esta suposição é válida se houver estratificação térmica, forças perturbadoras provenientes do vento e processos que geram gradientes horizontais desprezíveis (Hornung, 2002). O modelo unidimensional de temperatura pode ser uma importante ferramenta em estudos de reservatórios, tendo em vista a pequena quantidade de dados de entrada no modelo (uma vez que tradicionalmente os monitoramentos ocorrem dessa maneira), poucos parâmetros de calibração e tempo de execução bastante reduzido em relação a modelos mais complexos.

Um dos maiores desafios na modelagem hidrodinâmica e ecológica está na atribuição dos valores dos parâmetros. Muitos estudos experimentais foram realizados com o intuito de determinar uma faixa de valores consistente para diversos parâmetros. Entretanto, grande parte desses estudos foi desenvolvida em ecossistemas aquáticos temperados, que apresentam padrões bem diferentes dos encontrados em ecossistemas tropicais e subtropicais (Fragoso Jr, 2008). Dessa forma, o modelador sempre deverá usar o bom senso na escolha de um valor adequado para seu sistema.

3.5 O modelo hidrodinâmico GLM

No presente estudo, empregou-se o modelo hidrodinâmico *General Lake Model* (GLM) por ele possuir código aberto e livre, o que facilita a replicação de seu uso, e ter sido aplicado em estudos que alcançaram desempenhos satisfatórios (Gebremariam, 2015; Read *et al.*, 2014). Além disso, esse modelo vem sendo empregado em estudos conduzidos no Brasil, em projetos de parceria das instituições UFMG, UnB e USP em cooperação com a França, o que permitirá a comparação de resultados. O GLM foi desenvolvido pelo Grupo de Pesquisa de Dinâmicas Aquáticas da *University of Western Australia* para simular a dinâmica térmica de corpos d'água lânticos.

O GLM é um modelo hidrodinâmico unidimensional destinado à simulação do balanço hídrico e dos processos de estratificação vertical e mistura. São calculados, em um intervalo de tempo horário, os perfis verticais de temperatura, salinidade e densidade, considerando o efeito das afluências e defluências no balanço hídrico, além do aquecimento e resfriamento da superfície e a mistura vertical (Hipsey *et al.*, 2014) (Figura 1.3).

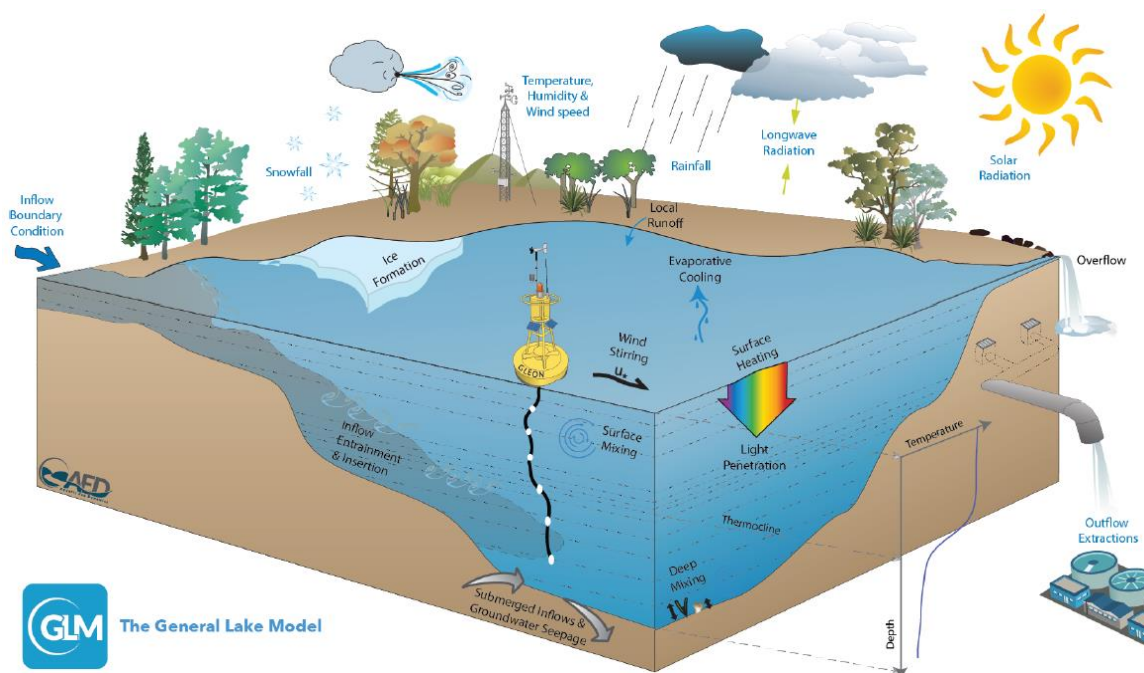


Figura 1.3 – Modelo conceitual do GLM, seus dados de entrada (texto em azul) e processos simulados (texto em preto). Fonte: Hipsey *et al.*, 2014.

O modelo requer os seguintes dados de entrada: (i) vazões afluentes, suas temperaturas e salinidade; (ii) vazões defluentes; (iii) séries meteorológicas, incluindo-se precipitação, temperatura do ar, umidade relativa do ar, velocidade do vento, radiação solar e nebulosidade

ou radiação de ondas longas; (iv) curva hipsográfica do segmento selecionado para descrever as relações entre elevação, área e volume do lago; e (v) perfil inicial de temperatura e salinidade da água em diferentes profundidades para o primeiro dia de simulação. O GLM adota a abordagem Lagrangeana, em que uma série de camadas horizontais, numeradas do fundo do lago à superfície, descreve a variação nas propriedades da coluna d'água. Assume-se que cada camada é um volume de controle que possui propriedades homogêneas. As camadas se contraem e expandem em resposta às vazões de entrada e saída e à mistura entre camadas adjacentes. Quando a energia suficiente torna-se disponível para superar as diferenças de densidade entre as camadas adjacentes, elas se mesclam, promovendo a mistura da coluna d'água.

O volume da camada é determinado por interpolação a partir da curva hipsográfica do lago. A primeira camada é calculada assumindo uma forma cônica e o volume das demais camadas (V_b) é calculado segundo a Equação 1.8. O número de camadas e a sua espessura são ajustados iterativamente em cada intervalo de tempo da simulação, a fim de alcançar uma resolução fina do gradiente vertical de temperatura.

$$V_b = V_{b-1} + [A_{b-1} + 0,5 \times (A_b - A_{b-1})] \times (h_b - h_{b-1}) \quad (1.8)$$

na qual V, A, h são o volume (m^3), a área (m^2) e a espessura da camada (m), respectivamente, b representa o intervalo de tempo na modelagem e b – 1 representa o intervalo de tempo anterior na modelagem.

Cada camada possui massa específica baseada em sua salinidade e temperatura, a qual é calculada de acordo com equação proposta por UNESCO (1981). À medida que a simulação progride, a densidade das camadas varia devido à transferência de energia no lago, à mistura vertical e a afluições e defluências. Quando se dispõe de energia suficiente para superar as diferenças de densidade entre as camadas adjacentes, elas se fundem, provocando a mistura da coluna d'água. O balanço de massa é calculado em um intervalo de tempo diário segundo a Equação 1.9, considerando-se: (i) afluições (Q_{afl}), incluindo vazões afluentes superficiais e profundas e o escoamento superficial da bacia de contribuição direta ao reservatório; (ii) defluências (Q_{def}), incluindo vazões defluentes e transbordamento no vertedouro; (iii) infiltração (Q_{inf}); (iv) evaporação (E) e; (v) precipitação (P_{rec}). As contribuições por evaporação e precipitação são consideradas na camada superficial; e a contribuição por infiltração é considerada na camada mais profunda.

$$\frac{dh_s}{dt} = Q_{afl} - Q_{def} - Q_{inf} - E + P_{rec} \quad (1.9)$$

na qual h_s é o nível d'água e t é o intervalo de tempo. A infiltração é considerada constante e é limitada para garantir que não mais do que 50% do volume da última camada possa ser reduzido em qualquer intervalo de tempo. As vazões afluentes superficiais adentram pela coluna d'água e, ao alcançar uma camada de mesma densidade, é criada uma nova camada, com espessura dependente do volume de entrada. Vazões afluentes submersas misturam-se com camadas adjacentes até se alcançar a mesma densidade, atingindo-se flutuabilidade neutra. A taxa de inserção da vazão afluente ($I_{Q_{afl}}$) é calculada segundo Fischer *et al.* (1979):

$$I_{Q_{afl}} = 1,6 \times \frac{C_{Di}^{3/2}}{Ri} \quad (1.10)$$

na qual C_{Di} é o coeficiente de arraste do leito e Ri é o número de Richardson, dado pela Equação 1.6. A espessura da camada de vazão defluente pode ser calculada a partir do número interno de Froude (Fr) ou do número de Grashof (Gr):

$$Fr = \frac{Q_{def}}{N^2 \times W \times L^2} \quad (1.11)$$

$$Gr = \frac{N^2 \times A^2}{\nu^2} \quad (1.12)$$

$$\delta = 2 \times L \times Gr^{-1/6} = 2 \times L \times Fr^{1/2} \quad (1.13)$$

na qual Q_{def} é a vazão defluente (m^3s^{-1}), N^2 é a frequência de Brunt-Väisälä (s^{-2}), W é a espessura do lago na profundidade de saída da vazão defluente (m), L é o comprimento do lago na profundidade de saída da vazão defluente (m), A é a área do lago na profundidade de saída da vazão defluente (m^2), ν é a difusividade vertical (m^2s^{-1}) e δ é a espessura da camada de vazão defluente (m).

O balanço de energia na camada superficial é calculado considerando-se a incidência de radiações de ondas curtas (radiação solar) (ϕ_{oc}), incidência de ondas longas provenientes da atmosfera (ϕ_{atm}), emissão de ondas longas pela superfície da água (ϕ_{sup}), trocas de calor sensível por convecção (ϕ_{sens}) e perdas de calor latente por evaporação (ϕ_{lat}):

$$\frac{c_p}{A_s \times z} \frac{dT_s}{dt} = \phi_{oc} + \phi_{atm} - \phi_{sup} + \phi_{sens} - \phi_{lat} \quad (1.14)$$

sendo:

$$\phi_{oc} = \frac{\phi_{oc}^{dir} + \phi_{oc}^{esp}}{1 - (\alpha_s \times \alpha_a)} \quad (1.15)$$

$$\phi_{atm} = (1 - \alpha_s) \times \varepsilon_a \times \sigma \times (T_a + 273,15)^4 \quad (1.16)$$

$$\phi_{sup} = \varepsilon_s \times \sigma \times (T_s + 273,15)^4 \quad (1.17)$$

$$\phi_{sens} = -\rho_a \times c_p \times C_H \times U_x \times (T_s - T_a) \quad (1.18)$$

$$\phi_{lat} = -\rho_a \times C_E \times \lambda \times U_x \times \frac{k}{p} \times [e_s(T_s) - e_a(T_a)] \quad (1.19)$$

na qual c_p é o calor específico do ar ($1.005 \text{ Jkg}^{-1}\text{°C}^{-1}$), T_s é a temperatura da superfície da água ($^{\circ}\text{C}$), A é a área superficial do reservatório (m^2), z é a profundidade da camada superficial (m), ϕ_{oc}^{dir} é a incidência direta de ondas curtas (Wm^{-2}), ϕ_{oc}^{esp} é a incidência de ondas curtas espalhadas pela atmosfera (Wm^{-2}), α_s é o albedo da superfície da água, α_a é o albedo da atmosfera, ε_a é a emissividade da atmosfera com ausência de nebulosidade (Wm^{-2}), σ é a constante de Stefan-Boltzman ($5,67 \times 10^{-8} \text{ Wm}^{-2}\text{K}^{-4}$), T_a é a temperatura do ar ($^{\circ}\text{C}$), ε_s é a emissividade da superfície da água ($0,985 \text{ Wm}^{-2}$), T_s é a temperatura da superfície da água ($^{\circ}\text{C}$), ρ_a é a massa específica do ar (kgm^{-3}), C_H é o coeficiente da transferência de calor sensível, U_x é a velocidade da camada superficial (ms^{-1}), C_E é o coeficiente da transferência de calor latente, λ é o calor latente de evaporação ($2,453 \times 10^6 \text{ Jkg}^{-1}$), k é a razão entre peso molecular da água e do ar ($0,622$), p é a pressão do ar (hPa), e_s é a pressão de vapor da superfície da água (hPa) e e_a é a pressão de vapor do ar (hPa).

A penetração da radiação de ondas curtas no lago e ao longo das camadas é modelada de acordo com a Lei de Beer-Lambert:

$$\phi_{oc}(z) = (1 - \alpha_s) f_s f_{PAR} \hat{\phi}_{oc} \exp[-K_W z] \quad (1.20)$$

na qual α_s é o albedo da superfície da água, f_s é o fator de escala, f_{PAR} é a fração de radiação fotossinteticamente ativa, que corresponde a 45% da incidência de ondas curtas, $\hat{\phi}_{oc}$ é a irradiância total (Wm^{-2}), K_W é o coeficiente de extinção da luz (m^{-1}) e z é a profundidade da camada (m).

A ocorrência dos processos de estratificação e mistura é calculada a partir de balanços energéticos. O modelo assume como premissa que a mistura ocorre se a energia cinética interna disponível (E_C , Equação 1.21) supera a energia potencial (E_P , Equação 1.22) requerida para manter as camadas estratificadas. O processo de mistura se encerra quando não há energia cinética interna suficiente para superar a energia potencial armazenada. A partir de então, inicia-se o processo de estratificação térmica. A energia cinética disponível é calculada a partir de quatro mecanismos, sendo vento, movimento de convecção, cisalhamento entre camadas e ondas de Kelvin-Helmholtz, que se constituem em vórtices formados pela diferença de velocidade entre camadas adjacentes.

$$E_C = 0,5 C_K C_W u_*^3 \Delta t + 0,5 C_K w_*^3 \Delta t + 0,5 C_S \left(u_b^2 + \frac{u_b^2 d\varepsilon}{6dz} + \frac{u_b \varepsilon du_b}{3dz} \right) \Delta z_{k-1} \quad (1.21)$$

$$E_P = \left[0,5 C_T (w_*^3 + C_W u_*^3)^{2/3} + \frac{\Delta \rho}{\rho_0} g z + \frac{g \varepsilon^2 d \Delta \rho}{24 \rho_0 dz} + \frac{g \varepsilon \Delta \rho d \varepsilon}{12 \rho_0 dz} \right] \Delta z_{k-1} \quad (1.22)$$

na qual C_K é a eficiência da mistura convectiva, C_W é a eficiência da mistura pelo vento, u_* é a tensão de cisalhamento na superfície da água devido ao vento (ms^{-1}), w_* é a velocidade da mistura turbulenta (ms^{-1}), C_S é a eficiência da mistura por cisalhamento, u_b é a velocidade de cisalhamento na interface da camada de mistura (ms^{-1}), ε é o comprimento de onda de Kelvin-Helmholtz (λ), z é a espessura da camada de mistura (m), $k-1$ é a profundidade inferior da camada de mistura (m), C_T é a eficiência da mistura por turbulência instável, ρ é a massa específica da água (kgm^{-3}), g é a aceleração da gravidade ($9,81 \text{ ms}^{-2}$) e t é o tempo (s).

Abaixo da camada superficial, o processo de mistura é calculado a partir da difusividade vertical (K_Z) segundo Weinstock (1981):

$$K_z = \frac{\omega \times \delta_*}{N^2 + 0,6 \times \tau_*^2 \times u_*^2} \quad (1.23)$$

na qual ω é o coeficiente de eficiência da mistura ($\sim 0,8$), δ_* é o coeficiente de difusividade, N^2 é a frequência de Brunt-Väisälä (s^{-2}), τ_* é o número de onda turbulenta e u_* é a tensão de cisalhamento na superfície da água devido ao vento (ms^{-1}). Outras equações empregadas pelo modelos GLM e os parâmetros físicos necessários ao modelo estão apresentados na Tabela 1.1 e Tabela 1.2.

3.6 O módulo ecológico AED

O módulo ecológico AED (*Aquatic EcoDynamics*) pode ser acoplado ao modelo hidrodinâmico GLM para simular a dinâmica ecológica de ambientes aquáticos a partir dos processos que governam a qualidade da água. Consiste em nove módulos que são projetados como componentes individuais que podem ser habilitados e configurados de forma a facilitar a representação do ecossistema aquático, seja ele mais simples ou complexo: (1) luz; (2) oxigênio dissolvido; (3) carbono, nitrogênio, fósforo e sílica; (4) sedimentos em suspensão e turbidez; (5) dinâmica geoquímica; (6) matéria orgânica; (7) patógenos; (8) fitoplâncton; e (9) zooplâncton (Hipsey *et al.*, 2013) (Figura 1.4). São também contabilizados os fluxos nas interfaces ar-água e sedimento-água.

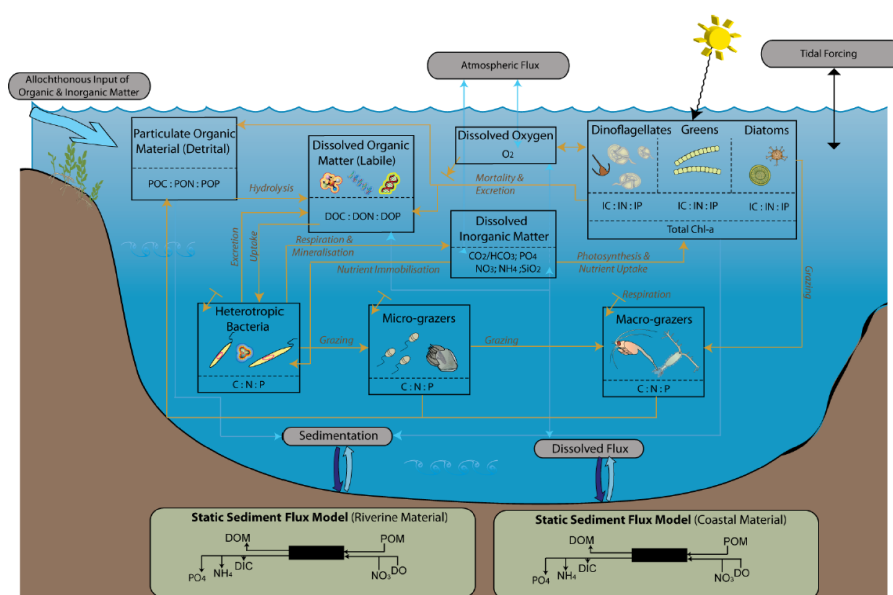


Figura 1.4 – Modelo conceitual do AED. Fonte: Hipsey *et al.*, 2013.

Tabela 1.1 – Equações empregadas no modelo GLM (Hipsey *et al.*, 2014).

Descrição	Símbolo	
Espessura da camada	$h = 1,2 \times i_E \times z + h_0$	(1.24)
Fator da densidade a nível do mar	$\rho(T, S, 0) = A + BS + CS^{3/2} + DS^2$	(1.25)
Fator da densidade	$K(T, S, P) = E + FP + GP^2$	(1.26)
Coefficiente A	$A = 999,8425 + 6,794 \times 10^{-2}T - 9,095 \times 10^{-3}T^2 + 1,002 \times 10^{-4}T^3 - 1,120 \times 10^{-6}T^4 + 6,536 \times 10^{-9}T^5$	(1.27)
Coefficiente B	$B = 8,245 \times 10^{-1} - 4,089 \times 10^{-3}T + 7,643 \times 10^{-5}T^2 - 8,246 \times 10^{-7}T^3 + 5,387 \times 10^{-9}T^4$	(1.28)
Coefficiente C	$C = -5,725 \times 10^{-3} + 1,022 \times 10^{-4}T - 1,654 \times 10^{-6}T^2$	(1.29)
Coefficiente D	$D = 4,831 \times 10^{-4}$	(1.30)
Coefficiente E	$E = (19652,21 + 54,675S + 7,944 \times 10^{-2}S^{3/2}) + (148,421 + 0,604S + 1,648 \times 10^{-2}S^{3/2})T + (-2,327 + 0,011S - 5,301 \times 10^{-4}S^{3/2})T^2 + (1,36 \times 10^{-2} - 6,167 \times 10^{-5}S)T^3 + (-5,155 \times 10^{-5})T^4$	(1.31)
Coefficiente F	$F = (3,240 + 2,284 \times 10^{-3}S + 1,91075 \times 10^{-4}S^{3/2}) + (1,437 \times 10^{-3} + 1,098 \times 10^{-5}S)T + (1,161 \times 10^{-4} - 1,608 \times 10^{-6}S)T^2 + (-5,779 \times 10^{-7})T^3$	(1.32)
Coefficiente G	$G = (8,510 \times 10^{-5} - 9,935 \times 10^{-7}S) + (-6,123 \times 6 + 2,081 \times 10^{-8}S)T + (5,279 \times 10^{-8} + 9,170 \times 10^{-10}S)T^2$	(1.33)
Evaporação	$E = -\frac{\phi_{ia\tau}A}{Le} + \rho AR$	(1.34)
Número de Richardson	$Ri = \frac{C_{Di}(1+0,21\sqrt{C_{Di}sen\alpha})}{sen\alpha \tan\beta}$	(1.35)
Espessura da camada superficial	$h_0 = \left(\frac{2Q_{in}^2 Ri}{g'} \tan^2 \alpha\right)^{1/5}$, sendo $g' = \frac{\rho_{in} - \rho_s}{\rho_s}$	(1.36)
Albedo da superfície da água	$\alpha_s = 0,08 - 0,02sen\left(\frac{2\pi}{365}d - \frac{\pi}{2}\right)$	(1.37)
Emissividade atmosférica	$\varepsilon_a = (1 + 0,275C)(1 - 0,261e^{-0,000777T_a^2})$	(1.38)
Velocidade da camada superficial	$U_x = \frac{u_*^2}{\Delta z} \Delta t$	(1.39)
Pressão de vapor de saturação da água	$e_s = exp\left[2,3026\left(\frac{7,5T_s}{T_s + 237,3} + 0,7858\right)\right]$	(1.40)
Pressão de vapor do ar	$e_a = \frac{UR}{100} e_s$	(1.41)
Tensão de cisalhamento na superfície	$u_* = U_x \sqrt{C_D}$	(1.42)
Velocidade da mistura turbulenta	$w_*^3 = \frac{g}{\rho_m \Delta t} (\rho_k \Delta z_k h_k - h_m \rho_k \Delta z_k)$	(1.43)
Velocidade na interface da camada de mistura	$u_b = u_{b-1} + \frac{u_*^2 \times A_E}{V}$, sendo $A_E = A \times \tanh \frac{A}{A_c}$	(1.44)
Comprimento de onda de Kelvin-Helmholtz	$\xi = \frac{C_{KH} u_b^2}{g'}$	(1.45)
Tensão induzida pela onda	$f_w = 0,00251 exp\left[5,213 \left(\frac{U_{orb} T_w}{4\pi D}\right)^{-0,19}\right]$, sendo $T_w = 7,54 \left(\frac{U_x}{g}\right) \tanh \gamma \times \tanh\left(\frac{0,0379 \left(\frac{gF}{U_x^2}\right)^{0,333}}{\tanh \gamma}\right)$, $\gamma = 0,833 \left(\frac{gh_{avg}}{U_x^2}\right)^{0,375}$, $F = 2 \sqrt{\frac{A}{\pi}}$	(1.46)
Velocidade da onda orbital no leito	$U_{orb} = \frac{\pi H_s}{T_w senh\left(\frac{2\pi d}{L}\right)}$, sendo $H_s = 0,283 \left(\frac{U_x^2}{g}\right) \tanh \mu \times \tanh\left(\frac{0,00565 \left(\frac{gF}{U_x^2}\right)^{0,5}}{\tanh \mu}\right)$, sendo $\mu = 0,53 \left(\frac{gh_{avg}}{U_x^2}\right)^{0,75}$, $L = \frac{gT^2}{2\pi} \tanh\left(\frac{2\pi h_{avg}}{\frac{gT^2}{2\pi}}\right)$	(1.47)
Tensão induzida pela corrente	$f_c = \frac{0,24}{\log\left(\frac{12h_{avg}}{2,5D}\right)}$	(1.48)

Tabela 1.2 – Parâmetros físicos empregados no modelo GLM (Hipsey *et al.*, 2014).

Parâmetro	Símbolo	Valor	Unidade
<i>Estrutura do modelo</i>			
Espessura mínima da camada	$h_{\min}$	0,5	m
Espessura máxima da camada	$h_{\max}$	1,5	m
<i>Balanço de massa</i>			
Coefficiente de arraste do leito	C_{Di}	0,016	-
Ângulo de entrada da vazão afluyente	α	2	°
Declividade do curso d'água afluyente	β	65	°
Calor latente de evaporação da água	L_e	540	calg ⁻¹
<i>Termodinâmica na superfície</i>			
Calor específico do ar	c_p	1005	Jkg ⁻¹ °C ⁻¹
Albedo da atmosfera	α_a	0,03	-
Fração fotossinteticamente ativa	k_{PAR}	0,45	-
Constante de Stefan-Boltzman	σ	$5,67 \times 10^{-8}$	Wm ⁻² K ⁻⁴
Emissividade da superfície da água	ε_s	0,985	-
Coefficiente de transferência de calor sensível	C_H	0,0013	-
Coefficiente de transferência de calor latente	C_E	0,0013	-
Calor latente de evaporação	λ	$2,453 \times 10^6$	Jkg ⁻¹
Razão entre peso molecular da água e do ar	k	0,622	-
<i>Balanço energético</i>			
Eficiência da mistura convectiva	C_K	0,2	-
Eficiência da mistura pelo vento	C_W	0,23	-
Eficiência da mistura por cisalhamento	C_S	0,3	-
Eficiência da mistura por turbulência instável	C_T	0,51	-
Coefficiente de transferência de momento	C_D	0,0013	-
Eficiência da mistura por ondas de Kelvin-Helmholtz	C_{KH}	0,3	-
Área crítica	A_c	10^7	m ²
Aceleração da gravidade	g	9,81	ms ⁻²

O modelo AED requer os seguintes dados de entrada: (i) concentração diária de oxigênio dissolvido, nutrientes, matéria orgânica e fitoplâncton na vazão afluente ao reservatório; (ii) concentração inicial e concentração mínima durante o período de simulação de cada grupo do fitoplâncton e do zooplâncton; e (iii) concentração inicial de oxigênio dissolvido, nutrientes, matéria orgânica e pH.

A incidência de ondas curtas é fornecida pelo modelo hidrodinâmico GLM. O coeficiente de extinção da luz é dinamicamente ajustado (K_D) de acordo com as concentrações de sólidos suspensos, de carbono orgânico dissolvido e particulado e de fitoplâncton na coluna d'água:

$$K_D = K_w + K_e[SS] + K_e[COD] + K_e[COP] + \sum K_e PHY \quad (1.49)$$

na qual K_w é o coeficiente de extinção da luz, K_e é o coeficiente multiplicativo, $[SS]$ é a concentração de sólidos suspensos (mmolm^{-3}), $[COD]$ é a concentração de carbono orgânico dissolvido (mmolm^{-3}), $[COP]$ é a concentração de carbono orgânico particulado (mmolm^{-3}) e PHY é a biomassa de cada grupo fitoplanctônico (mmolm^{-3}).

A dinâmica do oxigênio dissolvido (OD) contempla a troca atmosférica, a demanda de oxigênio pelo sedimento, o consumo microbiano durante os processos de mineralização e nitrificação da matéria orgânica, a produção de oxigênio pela fotossíntese e o consumo de oxigênio pela respiração dos organismos:

$$\frac{\partial OD}{\partial t} = \pm f_{atm}^{OD} - f_{sed}^{OD} - \frac{f_{miner}^{COD}}{X_{C:OD}^{miner}} - \frac{f_{nitrif}^{NH_4}}{X_{N:OD}^{nitrif}} + \sum \left(\frac{f_{cons}^{PHY}}{X_{C:OD}^{PHY}} \right) - \sum \left(\frac{f_{resp}^{PHY}}{X_{C:OD}^{PHY}} \right) \quad (1.50)$$

na qual f_{atm}^{OD} é a função da troca atmosférica de OD, f_{sed}^{OD} é a função de demanda de OD pelo sedimento, f_{miner}^{COD} é a função do consumo de OD no processo de mineralização da matéria orgânica, $X_{C:OD}^{miner}$ é a taxa de conversão do carbono em OD no processo de mineralização, $f_{nitrif}^{NH_4}$ é a função do consumo de OD no processo de nitrificação, $X_{N:OD}^{nitrif}$ é a taxa de conversão do nitrogênio em OD no processo de nitrificação, f_{cons}^{PHY} é função de liberação de OD pela fotossíntese, $X_{C:OD}^{PHY}$ é a taxa de conversão de carbono em OD no processo de fotossíntese e f_{resp}^{PHY} é a função da respiração do fitoplâncton.

As formas inorgânica, orgânica, dissolvida e particulada de carbono (C), nitrogênio (N), fósforo (P) e sílica (Si) são modeladas de forma explícita ao longo da via de degradação de matéria orgânica particulada para matéria orgânica dissolvida e então para matéria inorgânica dissolvida. Os processos de decomposição e mineralização variam em resposta à temperatura e podem ser reduzidos sob condições anaeróbicas. O balanço de massa desses componentes está apresentado nas equações seguintes:

$$[C_{total}] = [COD] + [COP] + \sum PHY_C \quad (1.51)$$

$$\frac{dCH_4}{dt} = +f_{sed}^{CH_4} - f_{ox}^{CH_4} \quad (1.52)$$

$$\frac{\partial CID}{\partial t} = f_{miner}^{COD} + \sum [f_{resp}^{PHY} - f_{cons}^{PHY}] \quad (1.53)$$

$$[N_{total}] = [NO_3] + [NH_4] + [NOD] + [NOP] + \sum PHY_N \quad (1.54)$$

$$\frac{dNH_4}{dt} = +f_{sed}^{NH_4} + f_{miner}^{NOD} - f_{nitrif}^{NH_4} - \sum_a^{PHY} [p_{NH_4}^a \times f_{abs}^{PHY_N}] \quad (1.55)$$

$$\frac{dNO_3}{dt} = -f_{sed}^{NO_3} + f_{nitrif}^{NH_4} - f_{desnit}^{NO_3} - \sum_a^{PHY} [p_{NO_3}^a \times f_{abs}^{PHY_N}] \quad (1.56)$$

$$[P_{total}] = [PO_4] + [PO_4]^{ads} + [POD] + [POP] + \sum PHY_P \quad (1.57)$$

$$\frac{dPO_4}{dt} = +f_{sed}^{PO_4} + f_{miner}^{POD} \pm f_{ads}^{PO_4} - \sum_a^{PHY} f_{abs}^{PHY_P} \quad (1.58)$$

$$\frac{dPO_4^{ads}}{dt} = \pm f_{ads}^{PO_4} - f_{sed}^{PO_4^{ads}} \quad (1.59)$$

$$\frac{\partial (C,N,P)OD}{\partial t} = f_{decom}^{(C,N,P)OD} - f_{miner}^{(C,N,P)OD} + f_{sed}^{(C,N,P)OD} + \sum f_{excr}^{PHY_{C,N,P}} \quad (1.60)$$

$$\frac{\partial (C,N,P)OP}{\partial t} = -f_{decom}^{(C,N,P)OP} - f_{sed}^{(C,N,P)OP} + \sum f_{mort}^{PHY_{C,N,P}} \quad (1.61)$$

$$\frac{dSi}{dt} = +f_{sed}^{Si} - \sum_a^{PHY} f_{abs}^{PHY_{Si}} + \sum_a^{PHY} f_{excr}^{PHY_{Si}} \quad (1.62)$$

na qual $[C_{total}]$ é a concentração de carbono total, $[COD]$ é a concentração de carbono orgânico dissolvido, $[COP]$ é a concentração de carbono orgânico particulado, $PHY_{C,N,P,Si}$ é

a parcela de carbono, nitrogênio, fósforo e sílica, respectivamente, no fitoplâncton, CH₄ é metano, f_{sed} é a função de sedimentação, f_{ox} é a função de oxidação, CID é carbono inorgânico dissolvido, f_{miner} é a função de mineralização, f_{resp} é a função de respiração, f_{cons} é função de liberação de OD pela fotossíntese, $[N_{total}]$ é a concentração de nitrogênio total, $[NO_3]$ é a concentração de nitrato, $[NH_4]$ é a concentração de amônio, $[NOD]$ é a concentração de nitrogênio orgânico dissolvido, $[NOP]$ é a concentração de nitrogênio orgânico particulado, f_{nitrif} é a função de nitrificação, P_{NH_4,NO_3}^a é a preferência do grupo a por NH₄ e NO₃, respectivamente, f_{abs} é a função de absorção, f_{desnit} é a função de desnitrificação, $[P_{total}]$ é a concentração de fósforo total, $[PO_4]$ é a concentração de fosfato, $[PO_4]^{ads}$ é a concentração de fosfato adsorvido, $[POD]$ é a concentração de fósforo orgânico dissolvido, $[POP]$ é a concentração de fósforo orgânico particulado, f_{ads} é a função de adsorção, f_{decom} é a função de decomposição, f_{excr} é a função de excreção, f_{mort} é a função de mortalidade natural e Si é sílica.

O fitoplâncton pode ser simulado em grupos e seu balanço de massa ($dPHY/dt$) é calculado considerando-se a absorção de nutrientes e carbono (f_{abs}) e as perdas por excreção (f_{excr}), mortalidade natural (f_{mort}), respiração (f_{resp}), sedimentação (f_{sed}) e assimilação pelo zooplâncton (f_{assim}):

$$\frac{dPHY}{dt} = +f_{abs} - f_{excr} - f_{mort} - f_{resp} - f_{sed} - f_{assim} \quad (1.63)$$

Para cada grupo do fitoplâncton, a taxa de crescimento máximo a 20 °C é multiplicada pelos fatores limitantes de luminosidade e nutrientes (Equação 1.64). Embora possa haver alguma interação entre os fatores limitantes, considerar o valor mínimo entre eles provavelmente proporciona uma representação realista da limitação do crescimento (Rhee e Gotham, 1981).

$$f_{abs}^{PHY_{Ca}} = R_{cresc}^{PHY_a} (1 - k_{pr}^{PHY_a}) \Phi_{tem}^{PHY_a}(T) \Phi_{str}^{PHY_a}(T) \dots \min\{ \Phi_{luz}^{PHY_a}(I), \Phi_N^{PHY_a}(NO_3, NH_4, PHY_N), \Phi_P^{PHY_a}(PO_4, PHY_{Pa}), \Phi_{Si}^{PHY_a}(Si) \} [PHY_{Ca}] \quad (1.64)$$

na qual $f_{abs}^{PHY_{Ca}}$ é a assimilação de carbono pela fotossíntese, $R_{cresc}^{PHY_a}$ é a taxa de crescimento máximo a 20 °C, $k_{pr}^{PHY_a}$ é a taxa de crescimento algal pela fotorrespiração, $\Phi_{tem}^{PHY_a}(T)$ é o fator de inibição pela temperatura, $\Phi_{str}^{PHY_a}(T)$ é o fator de inibição pelo estresse metabólico,

$\Phi_{luz}^{PHY_a}(I)$ é o fator de inibição pela luminosidade, $\Phi_N^{PHY_a}(NO_3, NH_4, PHY_N)$ é o fator de inibição pelo nitrogênio, $\Phi_P^{PHY_a}(PO_4, PHY_{P_a})$ é o fator de inibição pelo fósforo, $\Phi_{Si}^{PHY_a}(Si)$ é o fator de inibição pela sílica e PHY_{C_a} é o grupo do fitoplâncton.

O nível de limitação da luminosidade no crescimento do fitoplâncton pode ser modelado considerando-se ou não a condição de fotoinibição. Sob condição de temperatura ótima, ocorre a produtividade máxima do fitoplâncton; acima dessa temperatura, a produtividade reduz até tornar-se nula na temperatura máxima suportada; e abaixo da temperatura padrão, a produtividade segue a equação de Arrhenius.

A absorção de nutrientes pode ser estimada considerando-se uma razão constante de conversão do nutriente em carbono ou considerando-se dinamicamente as reservas intracelulares. A absorção de nitrogênio é subdividida em absorção de NO_3 , NH_4 e NOD e a distinção entre a absorção de cada componente é estimada empregando-se um fator de preferência. A absorção de sílica é considerada apenas para o grupo fitoplanctônico das diatomáceas.

As perdas de biomassa por excreção, mortalidade natural e respiração são modeladas empregando-se um único coeficiente como "taxa de respiração". O efeito da salinidade sobre a mortalidade natural é dado por várias formulações, dependendo da sensibilidade dos grupos à salinidade. Outras equações e parâmetros biogeoquímicos e do fitoplâncton necessários ao modelo e os valores adotados pelo AED estão apresentados na Tabela 1.3, Tabela 1.4 e Tabela 1.5. Os parâmetros do fitoplâncton são apresentados para os grupos de cianobactérias e clorofíceas. Embora existam muitos parâmetros, muitos deles não são obrigatórios e dependem dos módulos selecionados pelo modelador.

Tabela 1.3 – Equações do modelo AED (Hipsey et al., 2013).

Processo	Equação	
Troca atmosférica de oxigênio dissolvido	$f_{atm}^{OD} = c_{atm}^{OD} ([OD]_{atm} - [OD]_z) / dz_s$	(1.65)
Demanda de oxigênio dissolvido pelo sedimento	$f_{sed}^{OD} = F_{max}^{OD} ([OD] / (K_{sed}^{OD} + [OD])) (\theta_{sed}^{OD})^{T-20} (A_z / dz_z)$	(1.66)
Mineralização da matéria orgânica	$f_{miner}^{COD} = R_{miner}^{COD} \theta_{miner}^{T-20} [COD] ([OD] / K_{decom}^{COD} + [OD])$	(1.67)
Nitrificação do amônio	$f_{nitrif}^{NH_4} = R_{nitrif} \theta_{nitrif}^{T-20} [NH_4] ([OD] / K_{nitrif} + [OD])$	(1.68)
Decomposição do carbono orgânico particulado	$f_{decom}^{COP} = R_{decom}^{COP} \theta_{decom}^{T-20} [COP] ([OD] / K_{decom}^{COP} + [OD])$	(1.69)
Sedimentação do carbono orgânico dissolvido	$f_{sed}^{COD} = F_{max}^{COD} [K_{sed}^{COD} / (K_{sed}^{COD} + [COD])] (\theta_{sed}^{COD})^{T-20} (A_z / dz_z)$	(1.70)
Sedimentação do carbono orgânico particulado	$f_{sed}^{COP} = [COP] \omega_{COP} / dz_z$	(1.71)
Sedimentação do amônio	$f_{sed}^{NH_4} = F_{max}^{NH_4} [K_{sed}^{NH_4} / (K_{sed}^{NH_4} + [OD])] (\theta_{sed}^{NH_4})^{T-20} (A_z / dz_z)$	(1.72)
Sedimentação do nitrato	$f_{sed}^{NO_3} = F_{max}^{NO_3} ([OD] / (K_{sed}^{NO_3} + [OD])) (\theta_{sed}^{NO_3})^{T-20} (A_z / dz_z)$	(1.73)
Sedimentação do nitrogênio orgânico dissolvido	$f_{sed}^{NOD} = F_{max}^{NOD} [K_{sed}^{NOD} / (K_{sed}^{NOD} + [NOD])] (\theta_{sed}^{NOD})^{T-20} (A_z / dz_z)$	(1.74)
Sedimentação do nitrogênio orgânico particulado	$f_{sed}^{NOP} = [NOP] \omega_{NOP} / dz_z$	(1.75)
Mineralização do nitrogênio orgânico dissolvido	$f_{miner}^{NOD} = R_{miner}^{NOD} \theta_{miner}^{T-20} [NOD] ([OD] / K_{decom}^{COP} + [OD])$	(1.76)
Fator de preferência do fitoplâncton por NH_4	$p_{NH_4}^a = [NO_3] [NH_4]^2 K_N / ([NH_4] + K_N) ([NO_3] + K_N) ([NH_4] + [NO_3]) ([NO_3] + K_N)$	(1.77)
Fator de preferência do fitoplâncton por NO_3	$p_{NO_3}^a = 1 - p_{NH_4}^a$	(1.78)
Desnitrificação do nitrato	$f_{desnit}^{NO_3} = R_{desnit} \theta_{desnit}^{T-20} [NO_3] (K_{desnit} / K_{desnit} + [OD])$	(1.79)
Decomposição do nitrogênio orgânico particulado	$f_{decom}^{NOP} = R_{decom}^{NOP} \theta_{decom}^{T-20} [NOP] ([OD] / R_{decom}^{NOP} + [OD])$	(1.80)
Sedimentação do fosfato	$f_{sed}^{PO_4} = F_{max}^{PO_4} [K_{sed}^{PO_4} / (K_{sed}^{PO_4} + [OD])] (\theta_{sed}^{PO_4})^{T-20} (A_z / dz_z)$	(1.81)
Sedimentação do fosfato adsorvido	$f_{sed}^{PO_4^{ads}} = [PO_4^{ads}] \omega_{ss} / dz_z$	(1.82)
Sedimentação do fósforo orgânico dissolvido	$f_{sed}^{POD} = F_{max}^{POD} [K_{sed}^{POD} / (K_{sed}^{POD} + [POD])] (\theta_{sed}^{POD})^{T-20} (A_z / dz_z)$	(1.83)
Sedimentação do fósforo orgânico particulado	$f_{sed}^{POP} = [POP] \omega_{POP} / dz_z$	(1.84)
Mineralização do fósforo orgânico dissolvido	$f_{miner}^{POD} = R_{miner}^{POD} \theta_{miner}^{T-20} [POD] ([OD] / R_{miner}^{POD} + [OD])$	(1.85)
Adsorção / Dessorção do fosfato	$f_{ads}^{PO_4} = [\phi_{ads}^{PO_4} ([PO_4]^{t+1}, SS, pH) \times [PO_4]^{t+1} - PO_4^{ads}] (1/\Delta t)$	(1.86)
Decomposição do fósforo orgânico particulado	$f_{decom}^{POP} = R_{decom}^{POP} \theta_{decom}^{T-20} [POP] ([OD] / R_{decom}^{POP} + [OD])$	(1.87)
Sedimentação da sílica	$f_{sed}^{Si} = F_{max}^{Si} [K_{sed}^{Si} / (K_{sed}^{Si} + [Si])] (\theta_{sed}^{Si})^{T-20} (A_z / dz_z)$	(1.88)
Fator de inibição pela temperatura	$\Phi_{tem} = \theta^{T-20} - \theta^{k(T-a)+b}$, sendo: $\begin{cases} T = T_{std}: f(T) = 1 \\ T = T_{opt}: \partial f(T) / \partial T = 0 \\ T = T_{máx}: f(T) = 0 \end{cases}$	(1.89)
Crescimento do fitoplâncton abaixo da temperatura padrão	$f(T) = \theta^{T-20}$	(1.90)
Fator de inibição pela luminosidade	$\Phi_{luz}^{PHY} = 1 - e^{(-I/I_K)}$, para não-fotoinibição	(1.91a)
	$\Phi_{luz}^{PHY} = \frac{I}{I_S} e^{(1-(I/I_S))}$, para fotoinibição	(1.91b)

Fator de inibição pelo nitrogênio	$\Phi_N^{PHY} = \frac{NH_4+NO_3}{NH_4+NO_3+K_N}$, para razão de conversão constante	(1.92a)
	$\Phi_N^{PHY} = \frac{[N]_{max}}{[N]_{max}-[N]_{min}} \left[1 - \frac{[N]_{min}}{[N]} \right]$, para razão de conversão dinâmica	(1.92b)
Fator de inibição pelo fósforo	$\Phi_P^{PHY} = \frac{FRP}{FRP+K_P}$, para razão de conversão constante	(1.93a)
	$\Phi_P^{PHY} = \frac{[P]_{max}}{[P]_{max}-[P]_{min}} \left[1 - \frac{[P]_{min}}{[P]} \right]$, para razão de conversão dinâmica	(1.93b)
Fator de inibição pela sílica	$\Phi_{Si}^{PHY} = \frac{SiO_2-SiO_{2min}}{(SiO_2-SiO_{2min})+K_{Si}}$, para razão de conversão constante	(1.94)
Excreção de carbono pelo fitoplâncton	$f_{excr}^{PHYC} = (1 - k_{fres}^{PHY}) k_{fdom}^{PHY} \hat{R}[PHY_C]$	(1.95)
Excreção de nitrogênio e fósforo pelo fitoplâncton	$f_{excr}^{PHY_{N,P}} = k_{fdom}^{PHY} \hat{R}[PHY_{N,P}]$	(1.96)
Excreção de sílica pelo fitoplâncton	$f_{excr}^{PHY_{Si}} = \hat{R}[PHY_{Si}]$	(1.97)
Perda de carbono pela morte do fitoplâncton	$f_{mort}^{PHYC} = (1 - k_{fres}^{PHY})(1 - k_{fdom}^{PHY}) \hat{R}[PHY_C]$	(1.98)
Perda de nitrogênio e fósforo pela morte do fitoplâncton	$f_{mort}^{PHY_{N,P}} = (1 - k_{fres}^{PHY}) \hat{R}[PHY_{N,P}]$	(1.99)
Respiração do fitoplâncton	$f_{resp}^{PHY} = k_{fres}^{PHY} \hat{R}[PHY_C]$	(1.100)
Coeficiente de perda do fitoplâncton	$\hat{R} = R_{resp}^{PHY} \Phi_{sal}^{PHY} (\theta_{resp}^{PHY})^{T-20}$	(1.101)
Fator de inibição pela salinidade	$\Phi_{sal}^{PHY} = \begin{cases} 1, & S < S_{ot}^{PHY} \\ 1 + \frac{(S_{bep}^{PHY}-1)S^2}{(S_{max}^{PHY}-S_{ot}^{PHY})^2} - \frac{2(S_{bep}^{PHY}-1)S_{ot}^{PHY}S}{(S_{max}^{PHY}-S_{ot}^{PHY})^2} + \frac{(S_{bep}^{PHY}-1)(S_{ot}^{PHY})^2}{(S_{max}^{PHY}-S_{ot}^{PHY})^2}, & S \geq S_{ot}^{PHY} \end{cases}$	(1.102)
Sedimentação do fitoplâncton	$f_{sed}^{PHY} = [PHY] \omega_{PHY} / dz_z$	(1.103)
Assimilação do fitoplâncton pelo fitoplâncton	$f_{assim}^{ZOO} = R_{cresc}^{ZOO} \Phi_{tem}$	(1.104)

Tabela 1.4 – Parâmetros biogeoquímicos empregados no modelo AED (Hipsey *et al.*, 2013).

Parâmetro	Símbolo	Valor <i>default</i>	Unidade
<i>Intensidade luminosa</i>			
Coefficiente multiplicativo	K_e	0,017	m^{-1}
<i>Concentração de oxigênio dissolvido</i>			
Taxa de conversão do carbono em oxigênio dissolvido no processo de mineralização	$X_{C:OD}^{miner}$	12:32	$mmolCmmolOD^{-1}$
Taxa de conversão do nitrogênio em oxigênio dissolvido no processo de nitrificação	$X_{N:OD}^{nitrif}$	14:32	$mmolNmmolOD^{-1}$
Taxa de conversão do carbono em oxigênio dissolvido no processo de fotossíntese	$X_{C:OD}^{PHY}$	12:32	$mmolCmmolOD^{-1}$
Fluxo máximo de oxigênio dissolvido para o sedimento	F_{max}^{OD}	6 – 38	$mmolm^{-2}d^{-1}$
Constante de meia-saturação do fluxo de oxigênio dissolvido para o sedimento	K_{sed}^{OD}	150,0	$mmolm^{-3}$
Fator de correção da temperatura para o fluxo de oxigênio dissolvido para o sedimento	θ_{sed}^{OD}	1,08	-
Taxa máxima de mineralização do carbono orgânico dissolvido	R_{miner}^{COD}	0,003 – 0,05	d^{-1}
Fator de correção da temperatura para a mineralização	θ_{miner}^{COD}	1,08	-
Constante de meia-saturação da mineralização	K_{miner}^{COD}	47 – 78	$mmolm^{-3}$
Taxa máxima de nitrificação	R_{nitrif}	0,03 – 0,05	d^{-1}
Fator de correção da temperatura para a nitrificação	θ_{nitrif}	1,08	-
Constante de meia-saturação da nitrificação	K_{nitrif}	62,5 – 93,7	$mmolm^{-3}$
<i>Ciclos do carbono, nitrogênio, fósforo e sílica</i>			
Taxa máxima de decomposição do carbono orgânico particulado	R_{decom}^{COP}	0,01 – 0,07	d^{-1}
Fator de correção da temperatura para a decomposição do carbono orgânico particulado	θ_{decom}^{COP}	1,08	-
Constante de meia-saturação da decomposição do carbono orgânico particulado	K_{decom}^{COP}	31,25	$mmolm^{-3}$
Fluxo máximo do carbono orgânico dissolvido para o sedimento	F_{max}^{COD}	5,2	$mmolm^{-2}d^{-1}$
Constante de meia-saturação do fluxo de carbono orgânico dissolvido para o sedimento	K_{sed}^{COD}	100,0	$mmolm^{-3}$
Fator de correção da temperatura para o fluxo de carbono orgânico dissolvido para o sedimento	θ_{sed}^{COD}	1,08	-
Taxa de sedimentação do carbono orgânico particulado	ω_{COP}	-1,0	md^{-1}
Fluxo máximo do amônio para o sedimento	$F_{max}^{NH_4}$	1,35 – 6,42	$mmolm^{-2}d^{-1}$
Constante de meia-saturação do fluxo de amônio para o sedimento	$K_{sed}^{NH_4}$	1,56 – 15,6	$mmolm^{-3}$
Fator de correção da temperatura para o fluxo de amônio para o sedimento	$\theta_{sed}^{NH_4}$	1,08	-

Fluxo máximo de nitrato para o sedimento	$F_{max}^{NO_3}$	-21,4 – -7,14	$\text{mmolm}^{-2}\text{d}^{-1}$
Constante de meia-saturação do fluxo de nitrato para o sedimento	$K_{sed}^{NO_3}$	2,14 – 15,6	mmolm^{-3}
Fator de correção da temperatura para o fluxo de nitrato para o sedimento	$\theta_{sed}^{NO_3}$	1,08	-
Fluxo máximo de nitrogênio orgânico dissolvido para o sedimento	F_{max}^{NOD}	0,07 – 0,57	$\text{mmolm}^{-2}\text{d}^{-1}$
Constante de meia-saturação do fluxo de nitrogênio orgânico dissolvido para o sedimento	K_{sed}^{NOD}	100	mmolm^{-3}
Fator de correção da temperatura para o fluxo de nitrogênio orgânico dissolvido para o sedimento	θ_{sed}^{NOD}	1,08	-
Taxa de sedimentação do nitrogênio orgânico particulado	ω_{NOP}	-1,0	md^{-1}
Taxa máxima de mineralização do nitrogênio orgânico dissolvido	R_{miner}^{NOD}	0,003 – 0,05	d^{-1}
Taxa máxima de desnitrificação	R_{desnit}	0,01 – 0,04	d^{-1}
Fator de correção da temperatura para desnitrificação	θ_{desnit}	1,05	-
Constante de meia-saturação da desnitrificação	K_{desnit}	12,5 – 15,6	mmolm^{-3}
Taxa máxima de decomposição do nitrogênio orgânico particulado	R_{decom}^{NOP}	0,005 – 0,010	d^{-1}
Fluxo máximo de fosfato para o sedimento	$F_{max}^{PO_4}$	0,080 – 0,125	$\text{mmolm}^{-2}\text{d}^{-1}$
Constante de meia-saturação do fluxo de fosfato para o sedimento	$K_{sed}^{PO_4}$	15,6	mmolm^{-3}
Fator de correção da temperatura para o fluxo de fosfato para o sedimento	$\theta_{sed}^{PO_4}$	1,08	-
Taxa de sedimentação do fitoplâncton	ω_{ss}	0,01 – 3,1	md^{-1}
Fluxo máximo do fósforo orgânico dissolvido para o sedimento	F_{max}^{POD}	0,03	$\text{mmolm}^{-2}\text{d}^{-1}$
Constante de meia-saturação do fluxo de fósforo orgânico dissolvido para o sedimento	K_{sed}^{POD}	150	mmolm^{-3}
Fator de correção da temperatura para o fluxo de fósforo orgânico dissolvido para o sedimento	θ_{sed}^{POD}	1,08	-
Taxa de sedimentação do fósforo orgânico particulado	ω_{POP}	-1,0	md^{-1}
Taxa máxima de mineralização do fósforo orgânico dissolvido	R_{miner}^{POD}	0,01 – 0,05	d^{-1}
Taxa máxima de decomposição do fósforo orgânico particulado	R_{decom}^{POP}	0,01 – 0,03	d^{-1}
Fluxo máximo de sílica para o sedimento	F_{max}^{Si}	0,6	$\text{mmolm}^{-2}\text{d}^{-1}$
Constante de meia-saturação do fluxo de sílica para o sedimento	K_{sed}^{Si}	150	mmolm^{-3}
Fator de correção da temperatura para o fluxo de sílica para o sedimento	θ_{sed}^{Si}	1,08	-

Tabela 1.5 – Parâmetros do fitoplâncton empregados no modelo AED para cianobactérias e clorofíceas (Hipsey *et al.*, 2013).

Parâmetro	Símbolo	Unidade	Cianobactérias	Clorofíceas
Razão de conversão de carbono em clorofila	Y_{cc}	mgCmgChla ⁻¹	50	50
Taxa de crescimento máximo a 20 °C	R_{cresc}^{PHY}	d ⁻¹	1,6	0,8
Taxa de crescimento algal pela fotorrespiração	k_{pr}^{PHY}	d ⁻¹	0,06	0,06
Temperatura ótima	T_{opt}	°C	28	26
Temperatura máxima	$T_{máx}$	°C	35	33
Temperatura padrão	T_{std}	°C	20	20
Taxa de respiração a 20 °C	R_{resp}^{PHY}	d ⁻¹	0,05	0,03
Fator de correção da temperatura para a respiração	θ_{resp}^{PHY}	-	1,08	1,05
Fração de perdas devido unicamente à respiração	k_{fres}^{PHY}	-	0,25	0,25
Fração de perdas de nutrientes que será convertida em matéria orgânica	k_{fdom}^{PHY}	-	0,2	0,2
Taxa de sedimentação do fitoplâncton	ω_{PHY}	md ⁻¹	-0,02	-0,01
Taxa máxima de assimilação a 20 °C	R_{cresc}^{ZOO}	d ⁻¹	1,5	1,3
Fator de escala da temperatura de Arrhenius	θ	-	1,05	1,10
Constante de meia-saturação para a limitação pela luminosidade	I_K	μEm ⁻² s ⁻¹	130	80
Intensidade de saturação pela luminosidade	I_S	μEm ⁻² s ⁻¹	150	150
Concentração de meia-saturação do nitrogênio	K_N	mmolm ⁻³	1,0	2,7
Concentração interna de nitrogênio máxima	$[N]_{máx}$	mmolNmmolC ⁻¹	0,07	0,07
Concentração interna de nitrogênio mínima	$[N]_{min}$	mmolNmmolC ⁻¹	0,02	0,02
Concentração de meia-saturação do fósforo	K_P	mmolm ⁻³	0,05	0,07
Concentração interna de fósforo máxima	$[P]_{máx}$	mmolPmmolC ⁻¹	0,005	0,005
Concentração interna de fósforo mínima	$[P]_{min}$	mmolPmmolC ⁻¹	0,0005	0,0005
Limitação pela salinidade	S_{dep}^{PHY}	-	35	35
Salinidade máxima	S_{max}^{PHY}	gkg ⁻¹	0,0	0,0
Salinidade ótima	S_{ot}^{PHY}	gkg ⁻¹	0,0	0,0

4 MATERIAIS

4.1 Área de estudo

O reservatório Serra Azul localiza-se no município de Juatuba, na Região Metropolitana de Belo Horizonte (RMBH) (19,97°S, 44,34°O), a 760 m de altitude (Figura 1.5). Possui forma dendrítica, área superficial de 9,1 km², volume de 8,2 x 10⁷ m³, profundidade média de 9,0 m, profundidade máxima de 47,0 m (Fernandes, 2012) e o tempo de residência da água é de 375 dias. É considerado um reservatório de porte intermediário, segundo classificação de Straskraba e Tundisi (1999).

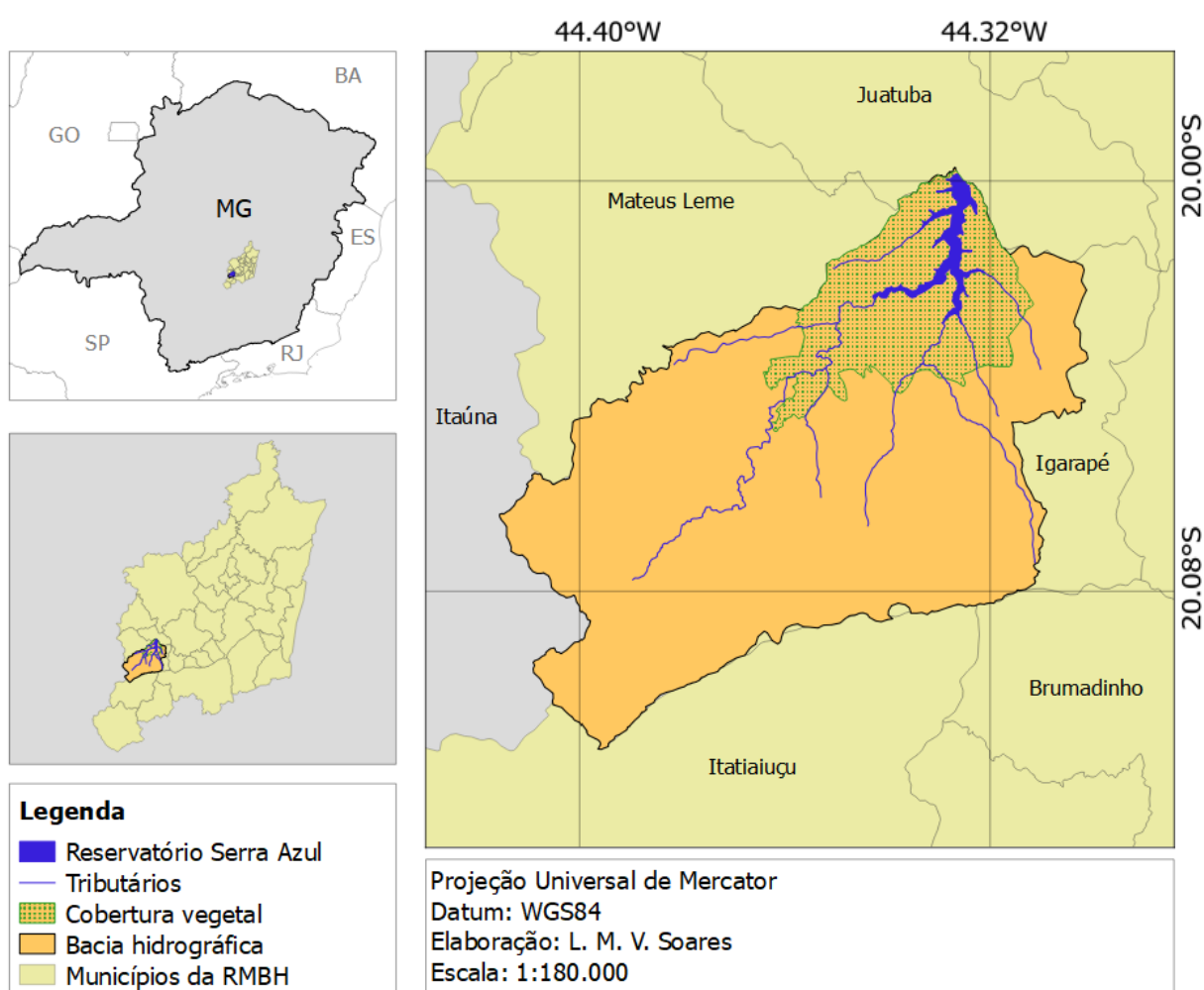


Figura 1.5 – Localização do reservatório Serra Azul, sua bacia hidrográfica e tributários.

O reservatório encontra-se em operação desde 1982 e destina-se exclusivamente ao abastecimento de água, constituindo-se atualmente em um dos principais mananciais da RMBH, atendendo cerca de 800.000 habitantes. A Companhia de Saneamento que opera o

reservatório (COPASA) possui outorga para captação de $2,9 \text{ m}^3\text{s}^{-1}$, mas no ano de 2017 operou com vazões aduzidas entre $0,25$ e $0,80 \text{ m}^3\text{s}^{-1}$ (informação verbal¹). O reservatório conta com três comportas em sua torre de captação e caracteriza-se pela reduzida presença de matéria orgânica e sedimentos, conduzindo a um baixo custo de tratamento para o abastecimento humano. Não há registros da ocorrência de floração de algas.

O reservatório possui oito tributários: Curralinho, Potreiro, Estiva, Diogo, Pedreira, Serra Azul, Jacu e Brejo. Em 1980, o Decreto nº. 20.792, do Governo do Estado de Minas Gerais, instituiu a Área de Proteção Especial Serra Azul, que compreende toda a bacia de contribuição do reservatório. Sua finalidade é garantir a qualidade e a manutenção desta área ambiental, uma vez que esta se encontra inserida em área periurbana. A Companhia de Saneamento é proprietária do entorno do reservatório, compreendendo uma área de 30 km^2 . Essa área é ocupada por vegetação (Figura 1.5) e contribui para a manutenção da qualidade da água no reservatório.

A bacia hidrográfica do reservatório Serra Azul possui área de drenagem de 262 km^2 . Aproximadamente 87% da área da bacia foi classificada como uma zona de interesse metropolitano dentro do Plano Metropolitano da RMBH, tendo sido esse enquadramento justificado devido às pressões em termos de uso e ocupação do solo na bacia e sua importância prioritária para fins de abastecimento de água para a região. O uso do solo está subdividido em: vegetação densa e esparsa (46%), usos agropastoris (26%), mancha urbana (10%), solo exposto (7%), afloramento rochoso (5%), mineração (2%) e corpos hídricos (4%) (Matos *et al.*, 2017) (Figura 1.6).

Apesar de sofrer alguma pressão antrópica, a qualidade da água do reservatório é pouco impactada. Dois programas de caráter ambiental se destacam na bacia do reservatório: (1) Programa Guardião dos Igarapés: desenvolvido pela Prefeitura Municipal de Igarapé em parceria com a Agência Nacional de Águas (ANA), visa à recuperação e preservação das nascentes do município para aumentar a produção de água. São realizadas ações de reflorestamento com espécies nativas, cercamento de áreas de proteção permanente, adequação de estradas, construção de bacias de infiltração (barraginhas) e trabalho de educação ambiental; e (2) Programa Pró-Mananciais: desenvolvido pela COPASA, contempla

¹ Informação fornecida por funcionário da COPASA em visita técnica ao reservatório Serra Azul.

ações de proteção e preservação ambiental, com o objetivo de recuperar os recursos naturais. Implementa ações como regularização do cadastro ambiental rural, cercamento de nascentes e construção de barraginhas.

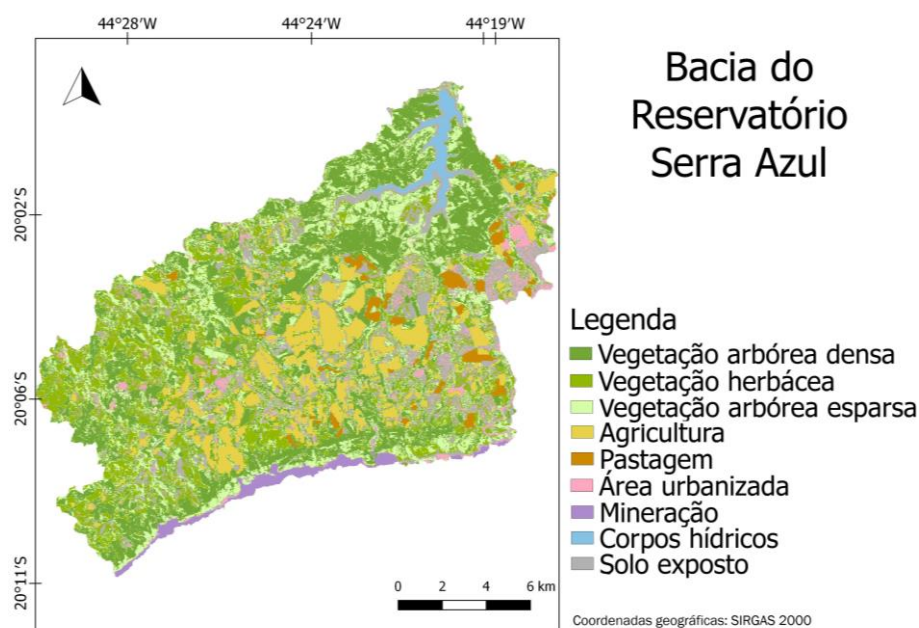


Figura 1.6 – Uso do solo na bacia hidrográfica do reservatório Serra Azul. Tratamento de imagem do satélite Sentinel com resolução de 10 m. Fonte: Matos *et al.*, 2017.

Na bacia hidrográfica do reservatório Serra Azul são encontrados solo podzólico vermelho amarelo álico (50% de ocorrência na bacia hidrográfica), latossolo vermelho (20%), solo podzólico vermelho amarelo distrófico (10%), solo aluvial, geralmente em áreas de várzeas e baixadas, próximos às margens dos principais cursos d'água (8%), solo litólico álico (7%) e solo litólico distrófico (5%) (Neves, 2005). A vegetação é composta por Cerrado, com variações de matas ciliares, campos e mata estacional semidecidual.

O clima na região é considerado tropical sub-úmido (Cwb) (Köppen, 1948), com período chuvoso de outubro a março e período seco de abril a setembro. A série histórica de 1961 a 1990 monitorada por estação meteorológica localizada próximo ao reservatório apresenta temperatura do ar média de 20,0 °C, atingindo valor mínimo médio de 7,9 °C em julho e valor máximo médio de 29,5 °C nos meses de fevereiro e março (Figura 1.7). No mesmo período, a precipitação média anual foi de 1.427 mm, variando em média de 12,0 mm no período seco a 309,2 mm no período chuvoso (INMET, 2017).

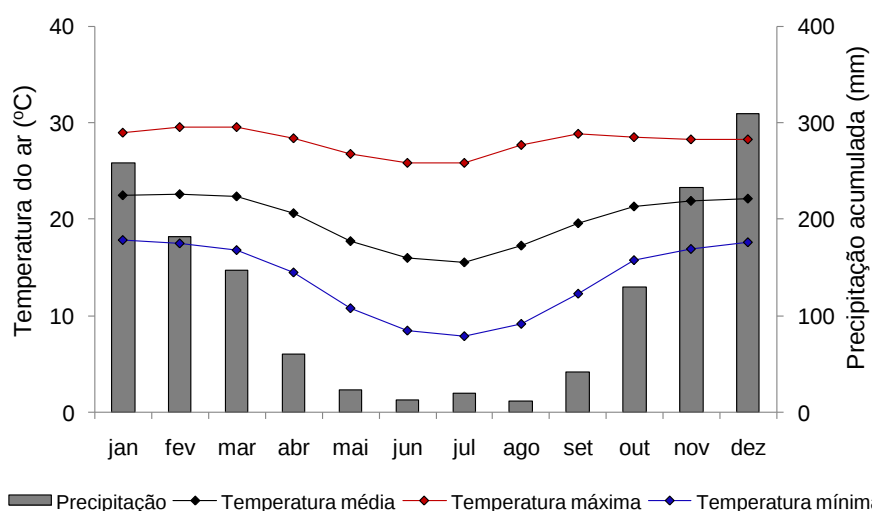


Figura 1.7 – Normais climatológicas da temperatura do ar e da precipitação acumulada da estação meteorológica Florestal de 1961 a 1990. Fonte: INMET, 2017.

O registro fotográfico das visitas em campo ao reservatório Serra Azul e sua bacia hidrográfica está apresentado no Apêndice A.

4.2 Estações de monitoramento

As séries históricas de qualidade da água, vazão afluente ao reservatório e dados meteorológicos foram obtidas a partir de cinco estações de monitoramento: estações 426, 371, Jardim, Serra Azul e Florestal (Tabela 1.6).

Tabela 1.6 – Estações de monitoramento consultadas.

Nome	Código	Monitoramento	Operação	Latitude	Longitude
426	-	Qualidade da água	COPASA ¹	19,97° S	44,34° O
371	-	Qualidade da água	COPASA	20,02° S	44,35° O
Jardim	40811100	Fluviométrico	CPRM ²	20,05° S	44,41° O
Serra Azul	02044054	Qualidade da água	CPRM	20,09° S	44,43° O
Florestal	86798	Meteorológico	INMET ³	19,88° S	44,42° O

¹ Companhia de Saneamento de Minas Gerais

² Serviço Geológico do Brasil

³ Instituto Nacional de Meteorologia

A estação 426 localiza-se no corpo do reservatório, a 200 m da torre de tomada d'água, e a estação 371 localiza-se no braço do reservatório Serra Azul, próximo à afluência dos córregos Estiva e Diogo (Figura 1.8). Essas estações são operadas pela COPASA e monitoram parâmetros físico-químicos na coluna d'água desde 1981 e 1990, respectivamente. As seguintes variáveis são monitoradas por ambas as estações: acidez, alcalinidade, cianetos, cloretos, condutividade elétrica, cor, demanda bioquímica de oxigênio, demanda química de oxigênio, dureza, fenóis, fluoreto, fósforo, série de nitrogênio, óleos e graxas, ortofosfatos,

oxigênio dissolvido, pH, sólidos dissolvidos, sólidos suspensos, sólidos totais, sulfatos, sulfetos, temperatura da água, transparência, turbidez, fitoplâncton, zooplâncton, coliformes e metais. As variáveis físico-químicas foram monitoradas em diferentes profundidades ao longo da coluna d'água e o fitoplâncton foi coletado na profundidade de Secchi. Os dados monitorados foram disponibilizados pela COPASA.

O regime de vazões do ribeirão Serra Azul, principal tributário do reservatório, é monitorado pela estação fluviométrica Jardim, localizada a cerca de 3 km à montante do reservatório. O monitoramento das vazões iniciou-se em 1979 e os dados são coletados diariamente. Dados diários da qualidade da água do ribeirão são monitorados desde 1987 pela estação Serra Azul, localizada a aproximadamente 7 km à montante do reservatório. Ambas as estações localizam-se no município de Mateus Leme (Figura 1.8), sendo operadas pela CPRM e tendo como responsável a ANA. As séries históricas dessas estações encontram-se disponíveis para consulta através do Sistema de Informações Hidrológicas Hidroweb. Dados sobre as vazões defluentes foram fornecidos pela COPASA.

A estação climatológica Florestal, operada pelo INMET, localiza-se a 11 km do reservatório, no município de Florestal. Trata-se de uma estação automática em operação desde 2008. São monitorados os dados horários de temperatura do ar, umidade relativa do ar, temperatura do ponto de orvalho, pressão atmosférica, velocidade e direção do vento, radiação solar e precipitação. Temperatura do ar, umidade relativa do ar e velocidade do vento são monitorados a 10 m da superfície do solo e radiação solar e precipitação são monitorados ao nível do solo. A temperatura é medida por um termômetro de mercúrio em vidro; a umidade relativa do ar é estimada por um psicrômetro; a velocidade do vento é medida por um anemômetro de conchas; a radiação solar é medida por um piranômetro; e a precipitação é medida por um pluviômetro.

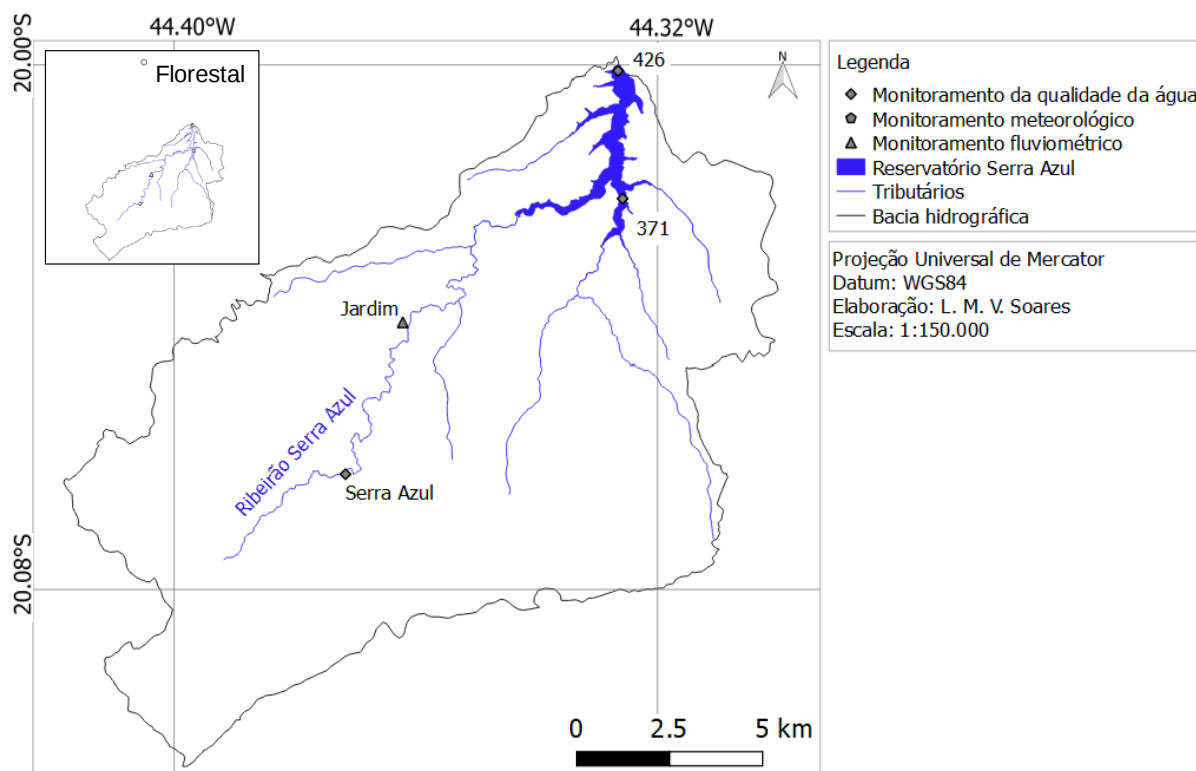


Figura 1.8 – Localização das estações de monitoramento consultadas: estação 426, estação 371, estação Jardim, estação Serra Azul e estação Florestal.

O período de dados disponíveis e a frequência de monitoramento de cada variável de interesse está apresentado na Figura 1.9.

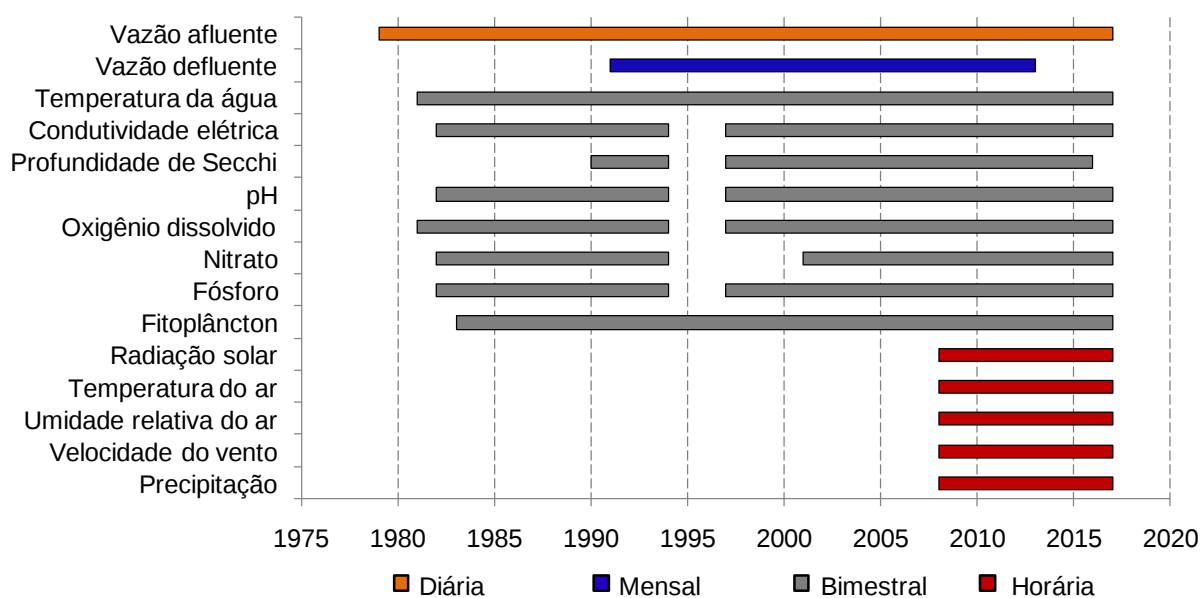


Figura 1.9 – Período e frequência de monitoramento das variáveis de interesse. Fonte: COPASA, 2018; INMET, 2017.

4.3 Tratamento dos dados

O reservatório Serra Azul é alimentado por oito tributários, no entanto, apenas o tributário principal (Ribeirão Serra Azul) possui monitoramento do seu regime de vazões a partir da estação Jardim. Dessa forma, adotou-se uma única contribuição afluenta ao reservatório aplicando-se um fator de correção baseado na área da bacia de contribuição do ribeirão Serra Azul (112,4 km²) e do reservatório (262 km²):

$$Q_{afl} = \frac{262}{112,4} \times Q_{RSA} \quad (1.105)$$

na qual Q_{afl} é a vazão afluenta ao reservatório (m³s⁻¹) e Q_{RSA} é a vazão do ribeirão Serra Azul (m³s⁻¹).

Considerou-se que a bacia de contribuição do reservatório possui características físicas homogêneas e que as vazões são proporcionais à área de drenagem de cada tributário. Falhas no registro de vazões foram preenchidas por interpolação linear e, para o período de 30 dias consecutivos de falhas no registro, adotou-se a média histórica de 1979 a 2015 para período com falha. Como perspectivas para estudos futuros, poderá ser empregada a série de vazões afluentes ao reservatório obtida a partir de modelagem hidrológica da bacia hidrográfica utilizando-se o *software* SWAT (*Soil and Water Assessment Tool*).

A série de dados da temperatura diária da água do ribeirão Serra Azul possui falhas de monitoramento que foram preenchidas a partir de regressão linear com os dados de temperatura diária do ar ($R^2 = 0,66$), uma vez que a temperatura do ar apresentou maior coeficiente de determinação em comparação com outras variáveis ambientais (Figura 1.10). Para os dias com dados de temperatura do ar ausentes, estimou-se a temperatura diária da água a partir da média histórica de 2009 a 2015 para cada dia.

A salinidade da vazão afluenta não é monitorada. Portanto, calculou-se a salinidade (Sal) a partir da condutividade elétrica corrigida para 20 °C (k_{20}), segundo equação empírica de Bäuerle *et al.* (1998) (Equação 1.106) A correção da condutividade em função da temperatura foi realizada segundo a Equação 1.107. Devido ao valor da salinidade ser inferior a 0,03, adotou-se um valor único de salinidade para cada ano.

$$Sal = 8,50 \times 10^{-4} \times k_{20} - 4,72 \times 10^{-4} \quad (1.106)$$

$$k_{20} = \frac{100}{100 + \theta \times (T - T_{20})} \times k_T \quad (1.107)$$

sendo:

$$\theta = \frac{(k_{T2} - k_{T1}) \times 100}{(T_2 - T_1) \times k_{T1}} \quad (1.108)$$

na qual θ é o coeficiente de temperatura, T é a temperatura da amostra ($^{\circ}\text{C}$), T_{20} é a temperatura de referência (20°C), k_T é a condutividade elétrica na temperatura monitorada (μScm^{-1}), k_{T1} e k_{T2} são a condutividade elétrica nas temperaturas T_1 e T_2 (μScm^{-1}) e T_1 e T_2 são a temperatura da água ($^{\circ}\text{C}$) durante monitoramento.

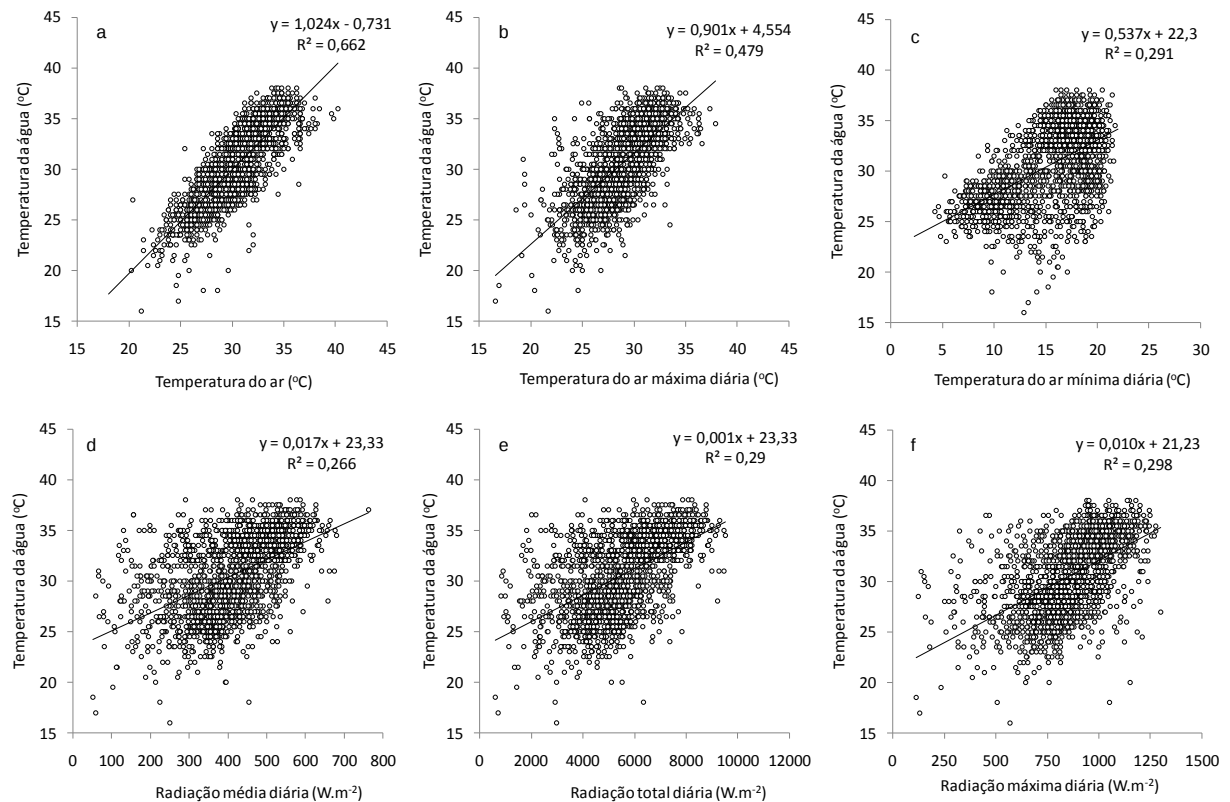


Figura 1.10 – Regressão linear entre temperatura da água e variáveis ambientais: a) temperatura do ar, b) temperatura do ar máxima diária, c) temperatura do ar mínima diária, d) radiação média diária, e) radiação total diária e f) radiação máxima diária. Fonte: INMET, 2017.

As séries de fósforo, oxigênio dissolvido e nitrato monitorados na estação 371 apresentaram treze dados durante o período de estudo. Devido à reduzida quantidade de dados, adotou-se o valor médio de cada variável (Tabela 1.7).

Tabela 1.7 – Valores médios adotados para as concentrações de fósforo, oxigênio dissolvido e nitrato na estação 371.

Variável	Média (mgL ⁻¹)	Desvio padrão (mgL ⁻¹)
Fósforo	0,01	0,00
Oxigênio dissolvido	6,59	0,55
Nitrato	0,28	0,08

Fonte: COPASA, 2018.

A ausência de dados meteorológicos devido a falhas em instrumentos ou manutenção dos mesmos totalizou 62 horas para cada variável ao longo do período de 2009 a 2016. Essas falhas foram preenchidas por interpolação linear.

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CAPÍTULO 2

Long-term characterization (1987-2016) of limnological features of a tropical reservoir

ABSTRACT

This study investigates key limnological features (water temperature, Secchi depth, turbidity, dissolved oxygen concentration, nitrate concentration, conductivity and phytoplankton biomass) over thirty years (1987–2016) of monitoring to provide a long-term limnological characterization of a tropical reservoir with good water quality. Phytoplankton biomass of each group was analyzed and the trophic state index was calculated based on chlorophyll-*a* concentration for each year of the studied period. Results showed that thermal structure was characterized with a warm-monomictic water circulation pattern. A seasonal pattern of phytoplankton biomass was identified, with higher biomass occurring under thermal stratification condition and lower biomass occurring under mixing events. In most of the long-term period, the reservoir presented an ultraoligotrophic or oligotrophic state.

Keywords: 30-years field data; water quality; phytoplankton biomass; trophic state.

1 INTRODUCTION

Reliable long-term data sets are very useful in order to understand temporal evolution of lake ecological behavior and also to detect trends since the rate of most ecological changes is very slow, usually occurring over decades to centuries (Smeltzer *et al.*, 2012). However, long-term series of water temperature profiles and other limnological variables are very scarce and only available for a few large lakes, *e.g.* Lake Geneva, Lake Constance, Lake Zürich, Mondsee and Lake Garda (Vinçon-Leite *et al.*, 2014). Important knowledge has been gained from long-term monitoring of lakes, including insights about patterns of phytoplankton fluctuation (Andrade and Girollo, 2014; Salmaso and Cerasino, 2012) and lake response to hydrological changes (Han *et al.*, 2016), nutrient loadings (Dietzel *et al.*, 2013), invasions by nonnative species (Minns *et al.*, 2011), and climate change (Hostetler, 2009; Wilhelm *et al.*, 2006).

Long-term studies of the temporal evolution of water temperature are recorded in literature. Twenty years (1983–2002) of temperature data from the epilimnion of Müggelsee showed a long-term increase with the daily air temperature minima increase (Wilhelm *et al.*, 2006). Eighty years (1911–1990) of monthly mean lake surface temperature data from eight lakes in Austria reflected much of the temporal structure of the regional air temperature (Livingstone and Dokulil, 2001). A unique data set of 50 years (1948–1997) of monthly temperature

profiles from Lake Zürich found a significant degree of hypolimnetic warming with a potentially negative impact on the lake ecosystem (Peeters *et al.*, 2002). August mean surface water temperatures increased during 1964–2009 in most Lake Champlain regions at rates 0.035–0.085 °C per year (Smeltzer *et al.*, 2012). The warming of surface waters has increased the maximum Schmidt's stability values at the deeper sites but bottom temperatures have not changed in Lake Ontario from 1972 to 2008 (Minns *et al.*, 2011).

Regarding long-term studies of the phytoplankton community, Lake Geneva revealed the good ability of the phytoplankton to resist both inter-annual fluctuations and brutal shifts in the annual functioning of the system from 1974 to 1998 (Anneville *et al.*, 2002). A reduction trend of chlorophyll-*a* and an increase trend of water transparency were observed in Lake Cote from 2002 to 2010, resulting in a reduction in primary productivity (Umaña, 2014). In Lake Ladoga, a long-term investigation (1956–2006) revealed that the ecological state deteriorated as a result of human impact and its trophic state has changed from oligotrophic to mesotrophic, with elevated nutrient concentrations, reduced water transparency and poorer fish catches (Viljanen *et al.*, 2008).

Long-term monitoring of lakes typically begins in response to perceived or documented deterioration of water quality (Larson *et al.*, 2007). Many lakes worldwide have been facing anthropogenic impacts and environmental stressors, such as land use changes, agricultural and industrial contaminants in water runoff, input of fertilizers, salinization, sewage, transport of exotic species and eutrophication, with diverse ecological impacts (Kalff, 2002). A study conducted across Brazil showed that the conditions in most of its inland water bodies ranged from mesotrophic to eutrophic (Abe *et al.*, 2006) mainly due to the occurrence of cyanobacterial blooms (Sant'anna *et al.*, 2008). However, the studied reservoir is a different example in terms of temporal evolution of water quality, especially when compared to other Brazilian reservoirs.

The water supply reservoir Serra Azul seems to be little impacted by land use stressors and blooms of cyanobacteria have never been detected in the last thirty years. To prevent water quality degradation, the Sanitation Company (COPASA) bought an area of 30 km² (about 11% of the total catchment area) around the reservoir which is completely covered by native vegetation. It is important to consider that some stressors may interact in synergistic ways

with preservation strategies, resulting in effects which are not always simple to disentangle (Salmaso and Cerasino, 2012).

Understanding the lake functionality is an essential step for the selection and application of effective long-term conservation and management strategies (Silva *et al.*, 2016). For a water supply reservoir such as Serra Azul, continuous monitoring is crucial for societal welfare and a secured water supply (Rhodes *et al.*, 2017). The aim of this study was to provide a long-term limnological characterization of Serra Azul reservoir over thirty years (1987–2016), and to classify the temporal evolution of the water quality using a trophic state index. It has expected to develop a better understanding of physical, chemical, and biological characteristics of Serra Azul reservoir, using this study site as a representative of high water quality reservoirs located in tropical regions.

2 MATERIAL AND METHODS

2.1 Study site

Serra Azul reservoir (19.97° S, 44.34° W) is located in Belo Horizonte Metropolitan Region (BHMR), southeastern Brazil, at elevation 760 m. At full capacity, the reservoir has a surface area of 9.11 km², its volume is 8.16 x 10⁷ m³, the average depth is 9.0 m and the maximum depth is 47.3 m (Fernandes, 2012). Its hydraulic residence time is 375 days. This reservoir was built in 1981 to meet the demand for drinking water of BHMR. The reservoir is exclusively used to supply drinking water to 800,000 inhabitants in the BHMR.

Its catchment covers an area of 262 km², mostly covered by vegetation (46%), farming (26%) and urbanization (10%). Serra Azul creek is the main tributary of the reservoir and its mean annual inflow is approximately 1.4 m³s⁻¹. The climate in the area is sub-humid tropical (Cwb) according to Köppen (1948). The region presents a striking seasonality with a dry season from April to September and a rainy season from October to March. Based on monthly hydrological and meteorological measurements for the period 1961–1990, mean annual precipitation is 1,427 mm, mean air temperature of the coldest month (July) is 15.5 °C, mean air temperature of the hottest month (February) is 22.6 °C, mean wind velocity is 1.26 ms⁻¹ and mean air relative humidity is 73.4% (INMET, 2017).

2.2 Field data analysis

More than 30 limnological variables are measured in the body of the reservoir, 200 m from the water withdrawal by the Sanitation Company (COPASA). The maximum depth of this location is 47 meters. For this study, phytoplankton biomass (μgL^{-1}) measured at Secchi depth and six limnological variables measured at different depths were selected based on the availability of data from 1987 to 2016: water temperature ($^{\circ}\text{C}$) at 0.5 m depth and 15.0 m depth, Secchi depth (m), turbidity (uT), dissolved oxygen concentration (mgL^{-1}) at 0.5 m depth and 15.0 m depth, nitrate concentration (mgL^{-1}) and conductivity (μScm^{-1}). According to Ji (2007), those are key parameters to represent the hydrodynamic and water quality conditions of surface waters. Nitrate concentration data were considered only for the values above the limit of detection of the measuring method (0.2 mgL^{-1}). Although phosphorus concentration is monitored and it is an important indicator of the water quality, this variable was not included in this study because many values were below the limit of detection (0.03 mgL^{-1}).

The biomass seasonal pattern of each phytoplankton class was identified. The trophic state index (TSI) of the reservoir was calculated according to Carlson (1977) modified by Lamparelli (2004) for each year in the long-term period (Equation 2.1). For this index, it was considered only the mean annual chlorophyll-*a* (Cla) concentration (Lamparelli, 2004) because transparency and phosphorus concentration data are not available for the entire period. TSI for chlorophyll-*a* is given by the following equation and the classification is given according to Table 2.1.

$$TSI(Cla) = 10 \left(6 - \left(\frac{0.92 - 0.34 \times \ln Cla}{\ln 2} \right) \right) \quad (2.1)$$

Table 2.1 – Water body classification according to the trophic state index (Lamparelli, 2004).

Classification	Range	Chlorophyll- <i>a</i> (μgL^{-1})
Ultraoligotrophic	$TSI \leq 47$	$Cla \leq 1.17$
Oligotrophic	$47 < TSI \leq 52$	$1.17 < Cla \leq 3.24$
Mesotrophic	$52 < TSI \leq 59$	$3.24 < Cla \leq 11.03$
Eutrophic	$59 < TSI \leq 63$	$11.03 < Cla \leq 30.55$
Supereutrophic	$63 < TSI \leq 67$	$30.55 < Cla \leq 69.05$
Hypereutrophic	$TSI > 67$	$Cla > 69.05$

3 RESULTS AND DISCUSSION

3.1 Thermal structure

Water temperature profile computed from field measurements in the period 1987–2016 indicates that the reservoir is thermally stratified from September to April (Figure 2.1). During these months, the seasonal epilimnion extends to a maximum depth of about 15 m. Mean temperature difference between surface (0.5 m depth) and 15 m depth is 4.9 °C. From May to August, temperature differences between surface and bottom ranged from 0.7 °C to 3.0 °C, indicating mixing of water column.

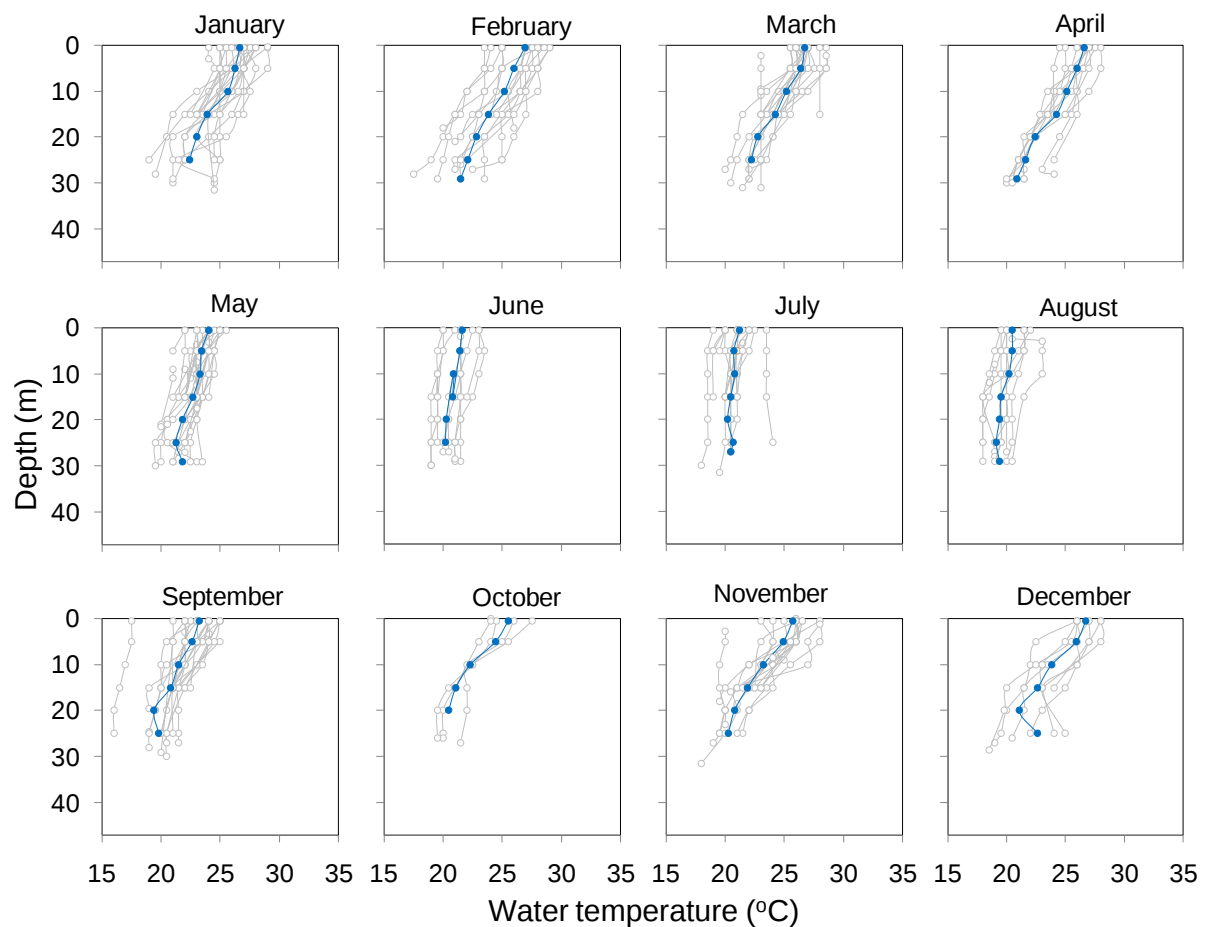


Figure 2.1 – Mean monthly water temperature profile (blue line) and profiles monitored (grey line) from 1987 to 2016. Source of data: COPASA, 2017.

Thermal structure of Serra Azul reservoir exhibits a warm-monomictic water circulation pattern. This is the result of the climatic conditions associated with the tropical latitudes in which it is located. Tropical lakes are fundamentally warm-monomictic, such as Aguamilpa reservoir in Mexico (Rangel-Peraza *et al.*, 2012), Dom Helvécio Lake in Brazil (Maia-

Barbosa *et al.*, 2010) and Lake Lanao in Philippines (Metillo and Garcia-Hansel, 2016), to cite some. Lakes in temperate latitudes are most often dimictic, although it is not universal. For example, Lake Tahoe in United States and Lake Bourget in France are warm-monomictic (Perroud *et al.*, 2009; Vinçon-Leite *et al.*, 2014).

Water temperature at 0.5 m depth ranges from 17.5 °C to 29 °C and at 15 m depth ranges from 16.5 °C to 28 °C. During the mixing event, the water column is around 21 °C. These values are similar to other reservoirs in tropical regions. Water temperature at 0.5 m depth in Lake Pampulha ranged from 19 °C to 28 °C (Silva *et al.*, 2016); in Garças reservoir, water temperature ranged from 14.6–17.3 °C in June-July to 24.4–26.6 °C in January-February (Bicudo *et al.*, 2007); average surface temperature in Tapacurá reservoir was 28.2 °C and the homogeneous water column in June was characterized by lower temperatures around 26 °C (Bouvy *et al.*, 2003). Field measurements of water temperature at 0.5 m depth and at 15 m depth follow a seasonal pattern over the long-term study period (1987 – 2016) (Figure 2.2).

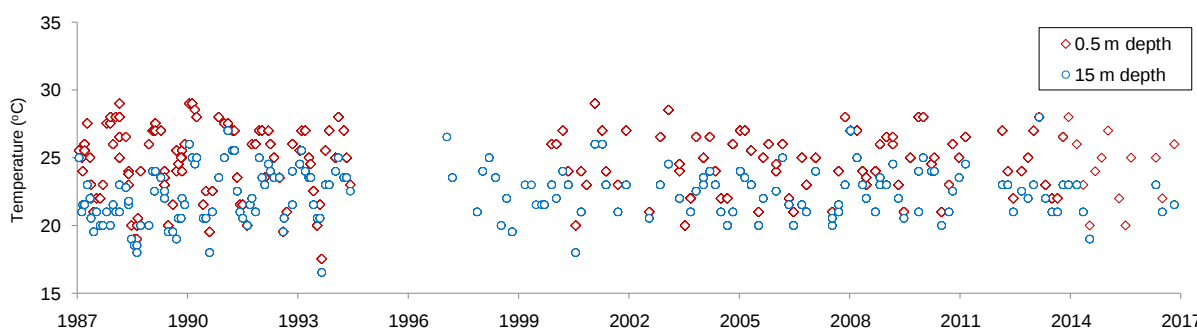


Figure 2.2 – Water temperature at 0.5 m depth (red) and 15 m depth (blue) at Serra Azul reservoir from 1987 to 2016. Dots are the monitored values and lines are the linear interpolation between them. Source of data: COPASA, 2017.

3.2 *Physico-chemical variables*

Reduction of water transparency was recorded during a shorter monitoring period from 2001 – 2016 (Figure 2.3a). Secchi depth varied from 0.5 m (November, 2014) to 5.8 m (September, 2004), with an average depth of 2.3 m. Mean turbidity over the period was 6.8 uT and it ranged from 0.2 uT to 84.9 uT (Figure 2.3b).

Dissolved oxygen concentration varied from 3.5 mgL⁻¹ to 10 mgL⁻¹ at 0.5 m depth (Figure 2.3c). At 15 m depth, it ranged from 0.3 mgL⁻¹ to 8.2 mgL⁻¹, which means that hypoxic conditions (concentration less than 2.0 mgL⁻¹) (Ji, 2007) are occasionally verified. Hydrodynamic effects appeared to strongly influence intra-annual variability of dissolved

oxygen concentration. Higher dissolved oxygen concentration at 15 m depth is observed in dotted vertical lines (Figure 2.3c), which correspond to July, 1st of each year, when mixing events usually occur. Low dissolved oxygen concentration persists in the deepest water during commonly annual stratification period from September to April since there is no atmospheric exchange and it is consumed by respiration and microbial degradation.



Figure 2.3 – Limnological variables of Serra Azul reservoir from 1987 to 2016: a) Secchi depth (m), b) turbidity (uT), c) dissolved oxygen concentration (mgL^{-1}), d) nitrate concentration (mgL^{-1}) and e) conductivity (μScm^{-1}). Dotted lines correspond to July, 1st of each year. Source of data: COPASA, 2017.

Given the central importance of dissolved oxygen for aquatic ecosystems, it has been at the center of numerous investigations of lake ecosystems. Lake Constance stores high concentrations of dissolved oxygen and its minimum value approached 5 mgL^{-1} at the lake's deepest point (250 m depth) (Rhodes *et al.*, 2017). Mean oxygen concentration was 10.1 mgL^{-1} at surface and 8.4 mgL^{-1} at bottom with minimum value reaching 0.2 mgL^{-1} in Lake Ontario (Minns *et al.*, 2011). In Bovilla reservoir, oxygen concentration ranged from approximately 5 mgL^{-1} at 20 m depth to 10 mgL^{-1} at surface (Shuka *et al.*, 2011).

Nitrate concentration ranged from 0.2 mgL⁻¹ (method detection limit) to 2.0 mgL⁻¹ (Figure 2.3d). Nitrate is not a concern in Serra Azul reservoir. Historically nitrate concentration in the reservoir has never exceeded the drinking water-quality standard of 10 mgL⁻¹ according to the national environmental council (CONAMA, 2005).

The water electrical conductivity of Serra Azul reservoir has been low for the whole period of observation, with a maximum of 58.9 µScm⁻¹ in September, 2013 and a minimum of 15.0 µScm⁻¹ in December, 1999 and March, 2005. The mean conductivity was 24.4 µScm⁻¹ (Figure 2.3e). Usually, larger values are reported in literature. In Monte Alegre reservoir, conductivity range was 66–142 µScm⁻¹ and mean value was 81.2 µScm⁻¹ (Huszar *et al.*, 1998). The conductivity of the water in Lake Cote varied from 17 µScm⁻¹ at the surface to 168 µScm⁻¹ at the bottom (Umaña, 2014). Mean values of conductivity in Bovilla reservoir ranged from 296.4 µScm⁻¹ in November to 324.3 µScm⁻¹ in March (Shuka *et al.*, 2011). Electrical conductivity in Lake Paranoá is between 100 and 180 µScm⁻¹ (Lorz *et al.*, 2012).

In Serra Azul reservoir, an increasing trend of conductivity seems to happen along the period. This trend needs to be statistically verified in further study. Similar trends in conductivity values were reported in literature. Conductivity values have increased gradually from 75 µScm⁻¹ to 87 µScm⁻¹ since the late 1960s due to human impact in Lake Ladoga (Viljanen *et al.*, 2008). On the other hand, the annual mean conductivity showed very small changes at any depth from 1972 to 1997 in Lake Zürich and it remained essentially constant at around 232 µScm⁻¹ at the surface during the whole period (Livingstone, 2003). Although the increase tendency in conductivity, the effect of salinity gradients on density profiles of Serra Azul reservoir can be neglected considering that stratification of the water column is dominated by temperature gradients, as in Lake Zürich (Peeters *et al.*, 2002).

Conductivity depends on the dissolved constituents in water and derives from inflow loading or local salt sources, often associated with mining (Boehrer and Schultze, 2008). The statistical metrics of studied variables are summarized in Table 2.2.

3.3 Phytoplankton dynamics

Phytoplankton dynamics was examined from 1987 to 2016 and the most representative classes in terms of biomass were cyanobacteria (51.2%), chlorophyceae (38.2%), chrysophyceae (7.9%) and pyrophyceae (2.7%) (Figure 2.4). 158 species were detected along

the period. Phytoplankton biomass ranged from $0.3 \mu\text{gL}^{-1}$ to $16.1 \mu\text{gL}^{-1}$ and mean value was $3.1 \mu\text{gL}^{-1}$. From 1987 to 2005, chlorophyceae was the most dominant group among the phytoplankton population, representing 70% of total biomass. In 1991, chlorophyceae biomass reached $14.7 \mu\text{gL}^{-1}$, its greatest biomass recorded in Serra Azul reservoir.

Table 2.2 – Statistical metrics of the limnological variables. Source of data: COPASA, 2017.

	n ¹	Mean	SD ²	Range
Water temperature at 0.5 m depth (°C)	178	24.6	2.5	17.5 – 29.0
Water temperature at 15 m depth (°C)	177	22.3	2.0	16.5 – 28.0
Secchi depth (m)	76	2.3	1.0	0.5 – 5.8
Turbidity (uT) in water column	1278	6.8	8.6	0.2 – 84.9
Dissolved oxygen at 0.5 m depth (mgL ⁻¹)	224	7.0	0.9	3.5 – 10.0
Dissolved oxygen at 15 m depth (mgL ⁻¹)	202	3.7	1.9	0.3 – 8.2
Nitrate (mgL ⁻¹) in water column	276	0.4	0.3	0.2 – 2.0
Conductivity (μScm ⁻¹) in water column	1207	24.4	5.2	15.0 – 58.9

¹ Number of observations

² Standard deviation

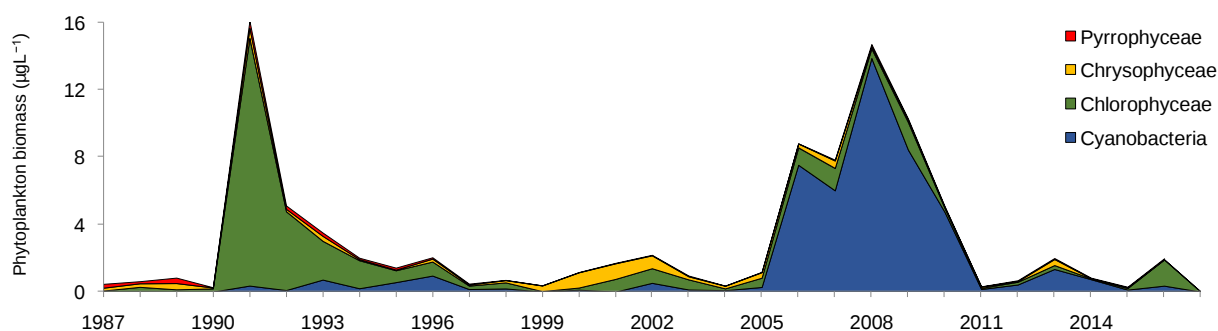


Figure 2.4 – Phytoplankton biomass of each class from 1987 to 2016. Source of data: COPASA, 2017.

From 2006 to 2015, cyanobacteria was dominant and peak densities occurred from 2006 to 2010 with biomass ranging from $4.8 \mu\text{gL}^{-1}$ (in 2010) to $13.9 \mu\text{gL}^{-1}$ (in 2008). Several factors can influence the dominance of cyanobacteria in phytoplankton communities, such as the higher stability of the water column (Calijuri *et al.*, 2002), light availability and several kinds of adaptive strategies (Dokulil and Teubner, 2000). However, it is difficult to determine which are the most important because synergistic forces are usually involved (Figueredo and Giani, 2009). In 2016, chlorophyceae became dominant again in phytoplankton community.

In tropical zones, temperature is not a limiting factor for phytoplanktonic development; hence hydrology, morphometry of water bodies, land use in catchments and the frequency, distribution and magnitude of rain, in addition to the velocity and direction of winds are the most important factors for explaining the occurrence, structure and variation of phytoplankton

(Andrade and Giroldo, 2014). Several long-term studies have been performed to establish patterns of phytoplankton fluctuations (Anneville *et al.*, 2004). In Guaíba Lake (Brazil), chlorophyll-*a* ranged from 2.7 μgL^{-1} to 88.9 μgL^{-1} and the most representative taxonomic classes in terms of richness were Chlorophyceae and Bacillariophyceae (Andrade and Giroldo, 2014). Phytoplankton of Lake Lanao (Philippines) were composed mostly of chlorophyceae and cyanobacteria (Metillo and Garcia-Hansel, 2016). The chlorophyll-*a* concentration at the surface of Lake Cote (Costa Rica) varied between 1.33 μgL^{-1} and 14.76 μgL^{-1} , with a mean of $4.86 \pm 3.07 \mu\text{gL}^{-1}$ (Umaña, 2014).

In Serra Azul reservoir, the highest phytoplankton biomass generally occurred in November and December (wet season), while the lowest biomass occurred in July and August (dry season). Monthly mean phytoplankton density by groups is presented in Figure 2.5a and shows that the peak densities of chlorophyceae and cyanobacteria occurs from September to January. Both groups presented the same seasonal pattern. Density variability of chrysophyceae and pyrrophyceae presented less sharp seasonality (Figure 2.5b).

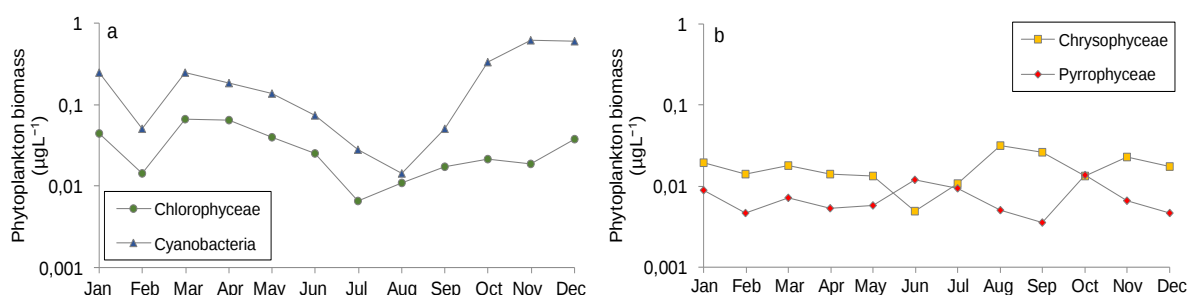


Figure 2.5 – Seasonal variability of mean phytoplankton biomass in Serra Azul reservoir for all years (from 1987 to 2016): a) chlorophyceae and cyanobacteria, and b) chrysophyceae and pyrrophyceae. Source of data: COPASA, 2017.

Phytoplankton acts as a refined sensor of environmental conditions (Alvarez-Cobelas *et al.*, 1998), hence, it is an important indicator when assessing water quality (Demir and Atay, 2002). The Trophic State Index (Lamparelli, 2004) considering only chlorophyll-*a* was calculated and revealed the high water quality of Serra Azul reservoir (Figure 2.6). In most of the long-term period, the reservoir was ultraoligotrophic or oligotrophic (thirteen years and nine years, respectively) (Figure 2.6). The mesotrophic state was observed in six years and the eutrophic condition just happened twice along thirty years.

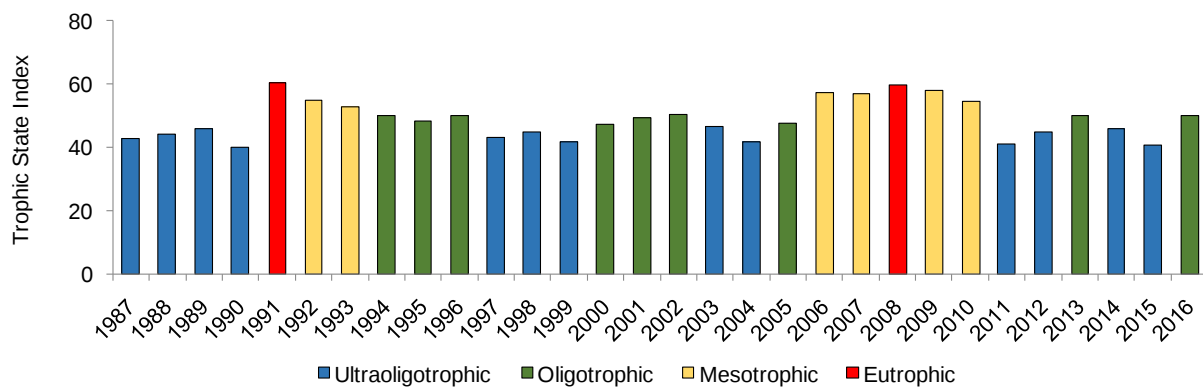


Figure 2.6 – Trophic State Index of Serra Azul reservoir from 1987 to 2016. Source of data: COPASA, 2017.

In comparison with other lakes, most of the physical and biological conditions of Serra Azul reservoir were consistent with those of oligotrophic lakes: high levels of dissolved oxygen throughout the year, low nitrate concentration, chlorophyceae dominance, low phytoplankton biomass and, as a consequence, water quality is usually good (Tundisi and Tundisi, 2008).

The present work suggests that the reservoir remains in a good condition of water quality, despite changes in land use in its catchment. It is probably due the small human interference in the catchment and a large extent of vegetation area (46% of total catchment in 2016). The results of our analysis highlighted the importance of long-term monitored data for providing new information essential for management strategies and biodiversity conservation to maintain the water quality.

This study only presents a limnological characterization of a tropical reservoir with high water quality. Numerous research questions remain and the links between environmental parameters, phytoplankton structure, land use in the catchment and meteorological conditions must be investigated to give a better understanding of the mechanisms responsible for the temporal evolution of ecological dynamics in the reservoir.

4 CONCLUSION

The long-term data provided the limnological characterization of Serra Azul reservoir and the classification of its trophic state from 1987 to 2016. The studied period of thirty years represents a relevant period of time relatively to the thirty six years of Serra Azul reservoir existence. The thermal structure was characterized as a warm-monomictic water circulation pattern and the reservoir is thermally stratified from September to April.

Mixing events were responsible for oxygenation of bottom layers, when higher concentrations were observed at 15 m depth. Hypoxic conditions were verified occasionally during stratification periods at this depth. Nitrate concentration seems to increase along the period, however it remain far below the drinking water-quality standard. Small values of conductivity were observed.

Four phytoplankton groups were detected along the study period: cyanobacteria, chlorophyceae, chrysophyceae and pyrophyceae. The dominant group was shifted from chlorophyceae to cyanobacteria from 2005 to 2015. A seasonal pattern was identified, with higher biomass in November and December (under thermal stratification condition) and lower biomass in July and August (under mixing events). In most of the long-term period, the reservoir was ultraoligotrophic or oligotrophic and the eutrophic condition just happened twice along thirty years. The results presented here suggest that the reservoir remains in a good condition of water quality, despite some changes in land use in its catchment.

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CAPÍTULO 3

Modelling thermal dynamics and phytoplankton in a tropical water supply reservoir

ABSTRACT

In this study, the one-dimensional hydrodynamic model GLM was coupled to the ecological model AED to simulate Serra Azul reservoir thermal structure and phytoplankton dynamics over a period of five years. First, a sensitivity analysis was performed to identify the most sensitive parameters in order to include them in the calibration process. Then, a manual calibration and validation of the hydrodynamic model were applied. Results of the sensitivity analysis indicated that water temperature was most sensitive to changes in inflow and wind speed; the phytoplankton biomass was most sensitive to the parameters related to growth. The hydrodynamic model reliably simulated the temperature profile of the water column over time and represented the warm-monomictic behavior of the reservoir. Although the ecological model was not calibrated, the model was able to reproduce the order of magnitude of the chlorophyll-a concentration for cyanobacteria and other functional groups and also their seasonal trends.

Keywords: hydrodynamics; ecological dynamics; 1D-V model; GLM-AED.

1 INTRODUCTION

Aquatic ecosystems provide a range of ecosystem services, in particular by being sources and sinks for natural resources and anthropogenic substances, and also attend to vast human necessities. More recently, their aesthetic and recreational value has been recognized with associated health benefits (Schallenberg et al., 2013). These anthropogenic and natural functions present specific requirements on the water quality of the aquatic ecosystem.

Endogenous factors (*e.g.* stratification, mixing, nutrients concentration, water temperature and euphotic depth) and exogenous factors (*e.g.* air temperature, solar radiation, wind speed and wind direction) induce modifications in the thermal structure in water column (Ji, 2007). It can directly and indirectly control the biological processes in the lake through successional changes in phytoplankton at species level, functional structure or size/biomass distribution (*e.g.* Padisák et al., 2009).

Changes in the algal communities can severely affect the water quality and even compromise the effectiveness of treatment processes for drinking water supply. The occurrence of blue-green algal blooms in water supply reservoirs may lead to severe clogging problems during

filtering operations; or it may lead to taste and odour in drinking water and even to health problems as a consequence of the presence of potential toxic cyanobacteria species.

Mathematical modeling may constitute a valuable tool for description of the interactions between physical and ecological processes and their impact on water quality. Since the early models development in the 1970s, hydrodynamic and ecological modelling of lake ecosystems have become more abundant and sophisticated as a result of increased demand and improved computing technology (Jorgensen, 2010). The field of aquatic ecosystem modelling has undergone major development during the past decades, and models have grown in complexity in terms of ecosystem components and processes included (Robson, 2014). One of the major challenges facing modellers is how to develop generic models that can capture the diversity of ecosystems while allowing prediction with confidence of the processes of each system (Bruce et al., 2018).

Modelling lake processes can be useful for a wide range of practical applications. For example, (i) to overcome the lack of in situ data (Read et al., 2014); (ii) to understand physical and ecological mechanisms from past monitored conditions (Gaeta et al., 2014); (iii) to conclude about the climate in the past by relating circulation patterns in lakes and climatic conditions triggering these changes (Bueche and Vetter, 2014); (iv) to test the potential effects that different management scenarios may have on the lake (Gal et al., 2009); (v) to facilitate immediate management decisions using near real-time modelling and forecasting, *e.g.* the shutdown of drinking water intakes (Silva et al., 2014); and (vi) to prognosticate the future of lakes (Hetherington et al., 2015).

Understanding the physical processes of transport and mixing that occur inside a reservoir and their driven forces is crucial for the predictability of phytoplankton composition and its temporal evolution. One-dimensional (1-D) model approach has found widespread use (*e.g.* Read et al., 2014; Vinçon-Leite et al., 2014; Weinberger and Vetter, 2012; Silva et al., 2016) due to its simplicity (minimal calibration requirement), computational efficiency and tractability relative to 3-D models (Bruce et al., 2018). Moreover, lake managers and reservoir operators prefer models having a simpler application and often rely on 1-D models for this reason (Kerimoglu and Rinke, 2013).

In this study, a 1-D hydrodynamic and ecological coupled modelling of a tropical water supply reservoir was applied in order to assess the thermal processes of stratification and

mixing of the water column, and to verify the parameters that most influence its physical, chemical and biological processes. Furthermore, this study also intends to identify the behavior of phytoplankton groups according seasonal patterns.

2 MATERIAL AND METHODS

2.1 Study area

Serra Azul reservoir (19.97° S, 44.34° W) is located in Belo Horizonte Metropolitan Region (BHMR), southeastern Brazil (Figure 3.1), at elevation 760 m. The reservoir has a surface area of 9.11 km², its volume is 8.16 x 10⁷ m³, the average depth is 9.0 m and the maximum depth is 47.3 m (Fernandes, 2012). Its hydraulic residence time is 375 days. This reservoir was built in 1981 and is exclusively used to supply drinking water to 800,000 inhabitants in the BHMR.

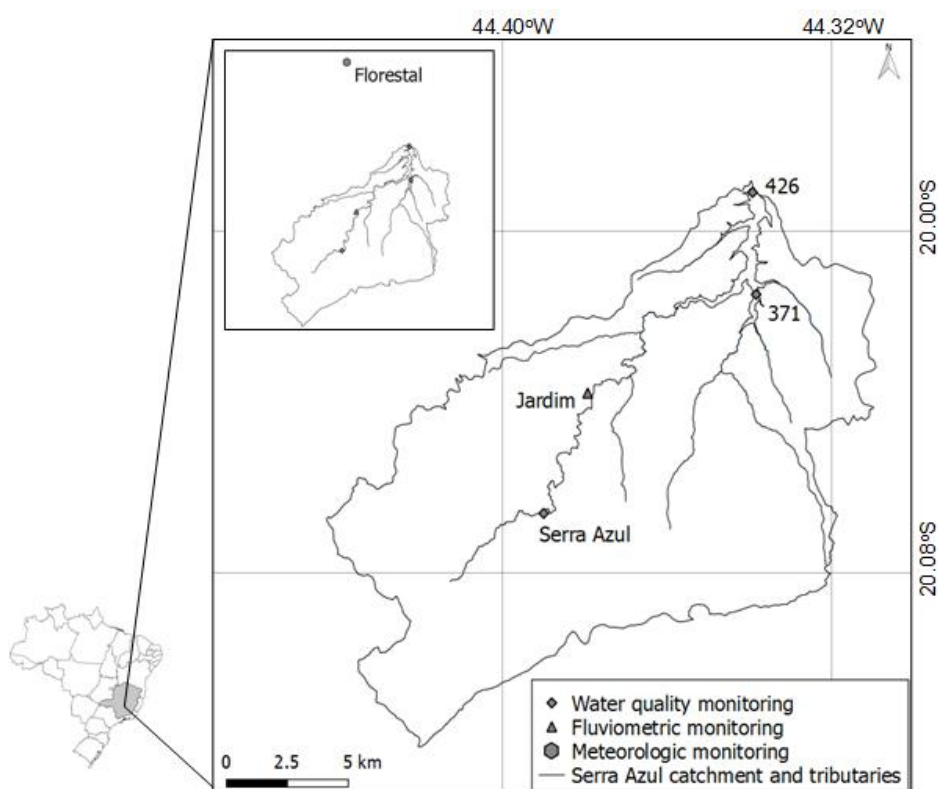


Figure 3.1 – Serra Azul reservoir, its tributaries and monitoring stations: 426, 371, Jardim, Serra Azul and Florestal.

The Serra Azul reservoir has eight tributaries and one outflow. Its catchment is a Special Protection Area of 262 km² and the land use is subdivided in vegetation (46%), farming (26%), urbanization (10%), exposed soil (7%), bedrock (5%), mining activities (2%) and the

reservoir itself (4%) (Matos *et al.*, 2017). Climate in the region is sub-humid tropical (Cwb) with a wet season from October to March and a dry season from April to September.

2.2 Monitoring stations

Two lake monitoring stations were consulted: station 426 located in the body of the reservoir, 200 m from the water intake; and station 371 located in an arm of the reservoir, in the affluence of a tributary. Those stations are operated by the Sanitation Company and monitor physicochemical parameters in the water column bimonthly since 1981 and 1990, respectively.

The main tributary (Serra Azul creek) is monitored by 2 stations: Jardim fluviometric station monitors daily discharge since 1979; and Serra Azul station monitors daily water temperature since 1987. Both Jardim and Serra Azul stations are operated by the Brazilian Geological Service (CPRM). One meteorological station was consulted: Florestal station is located 11 km far from the reservoir and it automatically monitors meteorological data every hour since 2008 by the National Institute of Meteorology (INMET). Monitoring stations are summarized in Table 3.1.

Table 3.1 – Monitoring stations.

Name	Code	Monitoring	Operated by	Latitude	Longitude
426	-	Water quality	COPASA ¹	19.97° S	44.34° W
371	-	Water quality	COPASA	20.02° S	44.35° W
Jardim	40811100	Fluviometric	CPRM ²	20.05° S	44.41° W
Serra Azul	02044054	Water quality	CPRM	20.09° S	44.43° W
Florestal	86798	Meteorologic	INMET ³	19.88° S	44.42° W

¹ Sanitation Company of Minas Gerais

² Geological Survey of Brazil

³ National Institute of Meteorology

2.3 Field data

Eight tributaries flow into Serra Azul reservoir, however, only Serra Azul creek has a fluviometric monitoring station (Jardim station). Therefore, the model was simplified to take into account one inflow contribution whose daily volume was computed from Jardim station data increased by the ratio between the reservoir and Jardim station drainage areas. Missing inflow data were filled by linear interpolation and for the period of 30 consecutive days of missing data, monthly inflow from 1979 to 2015 was adopted.

Serra Azul station provided water temperature of Serra Azul creek. Missing data were filled from linear regression with air temperature, $r^2 = 0.66$, since this was the best regression in comparison with other variables (daily maximum and minimum air temperature, daily mean, total and maximum shortwave radiation). When air temperature and water temperature were missing, mean monthly water temperature from 2009 to 2015 was used. The inflow salinity is not monitored and it was estimated from conductivity. As GLM adopts the International Equation of State for seawater (Hipsey *et al.*, 2014), it was found a conversion equation by linear regression between estimated salinity for freshwater and salinity for seawater. Considering that few conductivity data is available and the reservoir has a small content of salinity (below 0.03), it was adopted a unique value of salinity for all studied period.

Missing data from the meteorological station were rare and were replaced by linear interpolation. Meteorological and hydrological conditions near the reservoir during the study period (from 2009 to 2013) is presented in Figure 3.2. Total phytoplankton biomass per year from 2009 to 2013 monitored at the Secchi depths (from 1.2 m depth to 5 m depth) in Station 426 is presented in Figure 3.3.

2.4 Model description

The General Lake Model (GLM) has been recently developed by the Aquatic EcoDynamics Research Group team at University of Western Australia. This is an open-access model for simulating vertical profiles of temperature, salinity and density by accounting for the effect of inflows and outflows on the water balance, in addition to surface heating and cooling, energy balance and vertical mixing. The model adopts a one-dimension solution process of vertical mixing and is composed by a series of horizontal layers numbered from the lake bottom to the surface, describing the variation in the properties of the water column (Hipsey *et al.*, 2014).

The Lagrangian design assumes that each layer is a control volume that can change thickness by contracting and expanding in response to inflows, outflows, mixing with adjacent layers, and surface mass fluxes. Layer thicknesses are adjusted throughout the water column by the model in order to sufficiently resolve the vertical density gradient with fine resolution occurring in the thermocline and thicker cells where mixing is occurring. The mass balance is calculated for each layer and accounts for inflows (surface inflows, deep inflows and runoff

from the surrounding catchment), outflows (withdrawals, overflow and seepage in the bottom) and surface mass fluxes (evaporation and rainfall):

$$\frac{\partial h}{\partial t} = Q_{in} - Q_{out} - E + R \quad (3.1)$$

where h is the layer height, t is the time step, Q_{in} is the inflow, Q_{out} is the outflow, E is the evaporation mass flux and R is the rainfall.

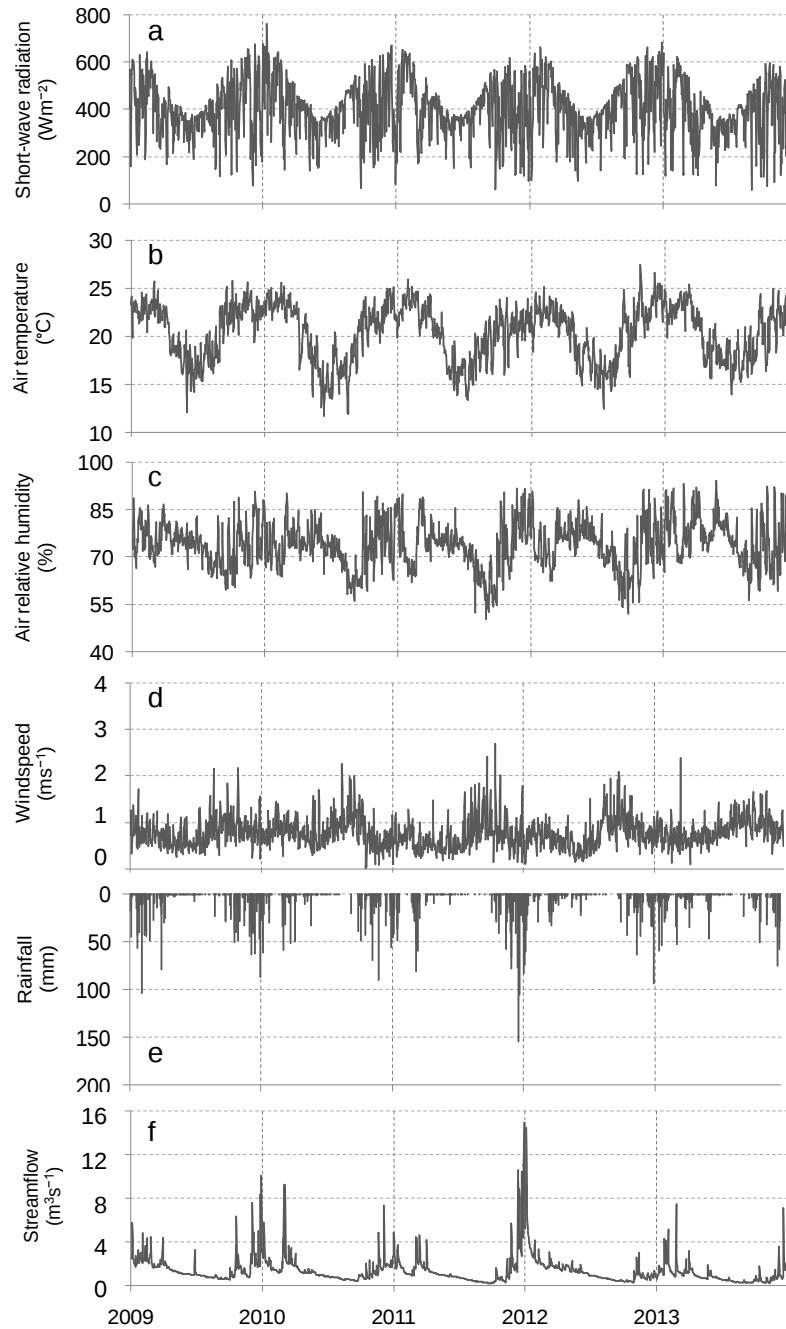


Figure 3.2 – Meteorological and hydrological variables measured at Florestal station and Jardim station: a) short-wave radiation, b) air temperature, c) air relative humidity, d) windspeed, e) rainfall and f) streamflow. Source fo data: INMET and CPRM.

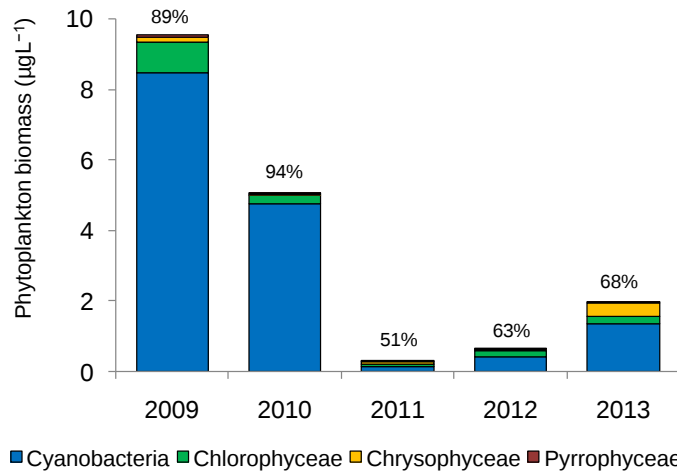


Figure 3.3 – Total phytoplankton biomass per year, distribution of groups and percentage of cyanobacteria in total biomass from 2009 to 2013. Source of data: COPASA.

The occurrence of stratification and mixing processes is calculated from energy balances. In this model, the available kinetic energy is calculated due to contributions from wind stirring, shear production between layers, convective overturn, and Kelvin-Helmholtz billowing. GLM works on the premise that the mixing occurs if the available kinetic energy (E_{TKE}) exceeds the potential energy (E_{PE}) required to keep the layers stratified:

$$E_{TKE} = 0.5C_K(w_*^3)\Delta t + 0.5C_K(C_W u_*^3)\Delta t + 0.5C_S \left[u_b^2 + \frac{u_b^2 d\varepsilon}{6 dz_{sml}} + \frac{u_b \varepsilon du_b}{3 dz_{sml}} \right] \Delta z_{k-1} \quad (3.2)$$

$$E_{PE} = \left[0.5C_T(w_*^3 + C_W u_*^3)^{2/3} + \frac{\Delta p}{\rho_0} g z_{sml} + \frac{g \varepsilon^2 d\Delta p}{24 \rho_0 dz_{sml}} + \frac{g \varepsilon \Delta p d\varepsilon}{12 \rho_0 dz_{sml}} \right] \Delta z_{k-1} \quad (3.3)$$

where C_K is the mixing efficiency of convective overturn, C_S is the mixing efficiency of shear production, C_T is the mixing efficiency of unsteady turbulence effects, C_W is the mixing efficiency of wind stirring, u_* is the wind velocity, ε is the Kelvin-Helmholtz billow length scale, g is the gravitational acceleration, $k - 1$ is the bottom of the mixed layer, ρ is the water density, t is the time step, u_b is the shear velocity at the interface of the mixed layer, w_* is the turbulent velocity scale associated with convection and z_{sml} is the thickness of the surface mixed layer.

The model can be coupled with the Aquatic EcoDynamics library (AED) for integrated simulations of temperature, stratification and vertical mixing with the dynamics of lake ecology (Hipsey *et al.*, 2013). This library consists of numerous modules that are designed as individual model components able to custom aquatic ecosystem conceptualizations – either simple or complex. Model components include the cycling of carbon, nitrogen and

phosphorus, and other relevant components such as dissolved oxygen, suspended sediment, dissolved inorganic nutrients, organic matter (dissolved and particulate), relevant fluxes at the air-water and sediment-water interface and are able to simulate organisms including different phytoplankton groups and zooplankton.

For primary production, the shortwave intensity at the surface is converted to the photosynthetically active radiation component and penetrates into the water column according to the Beer-Lambert Law. The light extinction coefficient calculated by AED (K_D) is dynamically adjusted to account for variability in the concentrations of suspended sediments, dissolved and particulate organic carbon levels and algae:

$$K_D = K_W + K_e SS + K_e DOC + K_e POC + K_e PHY \quad (3.4)$$

where K_W is the light attenuation coefficient measured with the Secchi disk, K_e is the multiplicative factor, SS is the concentration of suspended solids, DOC is the concentration of dissolved organic carbon, POC is the concentration of particulate organic carbon and PHY is the phytoplankton biomass.

Dissolved oxygen dynamics account for atmospheric exchange, sediment oxygen demand, microbial use during organic matter mineralization and nitrification, photosynthetic oxygen production and respiratory oxygen consumption. Both the inorganic and organic, and dissolved and particulate forms of carbon, nitrogen and phosphorus are modeled explicitly along the degradation pathway of particulate organic matter to dissolved organic matter and then to dissolved inorganic matter. The decomposition and mineralization process varies in response to temperature, and is additionally able to slow down under anaerobic conditions.

For each phytoplankton group, the maximum potential growth rate at 20 °C is multiplied by the minimum value of the equations accounting for limitation by light, phosphorus, nitrogen and silica (when configured). While there may be some interaction between limiting factors, a minimum expression is likely to provide a realistic representation of growth limitation:

$$f_{upt} = R_{gr}(1 - k_{pr})\Phi_t \Phi_{str} \min\{\Phi_{light}, \Phi_N, \Phi_P, \Phi_{Si}\} \quad (3.5)$$

where f_{upt} is the uptake of carbon for photosynthesis, R_{gr} is the phytoplankton growth rate at 20 °C, $1 - k_{pr}$ is the photorespiratory loss, Φ_t is the limitation by temperature, Φ_{str} is the

limitation by metabolic stress, Φ_{light} is the limitation by light, Φ_{N} is the limitation by nitrogen, Φ_{P} is the limitation by phosphorus and Φ_{Si} is the limitation by silica.

2.5 Model inputs

In this study, GLM required (i) hypsographic curve, (ii) initial profile of temperature and salinity in the water column; (iii) hourly meteorological time-series data for surface forcing, (iv) daily time series of volumetric inflow, its temperature and salinity and (v) volumetric outflow. Serra Azul reservoir morphometry (elevation - m, area - m^2 and volume - m^3) was obtained from Fernandes (2012). Initial conditions for water temperature and salinity were obtained from the field profile obtained closest to the start date (January 21th, 2009), by station 426 at 0.5 m, 5 m and 15 m depth. Hourly meteorological data (short wave radiation - Wm^{-2} , air temperature - $^{\circ}\text{C}$, wind speed - ms^{-1} , rainfall - m and air relative humidity - %) were obtained from Florestal station. Inflow volume was obtained from Jardim station and its temperature and salinity were obtained from Serra Azul station. The reservoir has a single outlet located in its northern portion. The water is withdrawal from this outlet for human supply and maintenance of the residual flow downstream of the dam. The outflow was estimated based on the monthly average flow withdrawal.

AED required (i) daily inflow concentration of dissolved oxygen (mmolm^{-3}), nitrate (mmolm^{-3}), phosphorus (mmolm^{-3}) and phytoplankton biomass (mmolm^{-3}). These data were obtained from Station 371, located in an arm of the reservoir in the affluence of a tributary, due to the absence of physico-chemical monitoring station in Serra Azul creek. For dissolved oxygen, nitrate and phosphorus concentrations it was assumed their mean values for the entire simulated period due to few available data. The concentrations of phytoplankton from the inflows were negligible; (ii) minimum and initial phytoplankton biomass (mmolm^{-3}) monitored in Station 426; and (iii) initial concentration of dissolved oxygen (mmolm^{-3}), phosphorus (mmolm^{-3}), nitrate (mmolm^{-3}), and pH in the water column. It was adopted the mean value from data monitored in different depths at January 21th, 2009 in Station 426.

2.6 Simulation

GLM-AED version 2.2 was used to model Serra Azul reservoir thermal dynamic and phytoplankton biomass from 2009 to 2013. Simulation was conducted with an 1 h time step,

and the state variables were output every 24 h. The statistical software R 3.2.5 was used to simulate and to analyze modelling results through three packages: GLM_r (Hipsey *et al.*, 2013) was used to run the model; GLMtools (Read *et al.*, 2014) was used to view simulation results; and Lake Analyzer (Winslow *et al.*, 2016) was used to calculate common metrics for lake physical states. R script for running and analyzing GLM is presented in Appendix B.

For the baseline simulation, it was adopted default parameters of GLM (Hipsey *et al.*, 2014). For AED module, the following model components were activated: sediments, dissolved oxygen, carbon, nitrogen, phosphorus, organic matter and phytoplankton. Considering the significant participation of the cyanobacteria in the phytoplankton community (see Figure 3.3), model was setup to simulate them explicitly and other groups were simulated aggregated into a single group.

Sediments and biogeochemical parameter values and phytoplankton parameters were defined based on literature (Silva *et al.*, 2016; Gal *et al.*, 2009; Schladow and Hamilton, 1997; Hipsey *et al.*, 2013). The inhibition of photosynthesis at high light intensities was taken into account.

2.7 Sensitivity analysis

Sensitivity analysis was manually performed in order to identify the parameters and input variables that most impacted hydrodynamic and ecological simulation. The analyzed parameters were defined based on literature and the one-at-a-time method was adopted.

For the hydrodynamic model, the sensitivity of water temperature was analyzed for the following parameters: minimum and maximum layer thickness, light extinction coefficient, mixing efficiency of wind stirring, mixing efficiency of shear production, streambed drag coefficient, streambed slope and stream half angle. In view of the uncertainties in wind speed and inflow data, wind factor and inflow factor were also included in the sensitivity analysis.

The range values adopted for the light extinction coefficient was determined based on the Secchi depth measured in the reservoir. For the other parameters, the range of values adopted in the sensitivity analysis (Table 2) was based on previous applications of one-dimensional lake models (*e.g.* Hetherington *et al.*, 2015; Weinberger and Vetter, 2012).

For the ecological model, the sensitivity of phytoplankton biomass was assessed to the following parameters: phytoplankton growth rate at 20 °C, half saturation constant for light limitation and loss fraction due to respiration. The parameters were tested separately for each phytoplankton group. The range values adopted for parameters (Table 3) were determined based on literature (Barbosa, 2015; Gal *et al.*, 2009; Silva *et al.*, 2016).

The Root Mean Squared Error (RMSE) and the Pearson's correlation coefficient (r) were computed from water temperature and phytoplankton biomass between simulated and observed values in order to assess model performance. RMSE and r are defined as follows:

$$RMSE = \sqrt{\frac{1}{N} \sum_{t=1}^N (O - S)^2} \quad (3.6)$$

$$r = \frac{\sum_{t=1}^N (O - \bar{O})(S - \bar{S})}{\sqrt{\sum_{t=1}^N (O - \bar{O})^2 \sum_{t=1}^N (S - \bar{S})^2}} \quad (3.7)$$

where N is the number of observations available, O is the observed value, $\bar{O}$ is the mean of observed values, S is the simulated value and $\bar{S}$ is the mean of simulated values.

Normalized sensitivity coefficients (S_c) to assess the relative sensitivity of the variable (water temperature or phytoplankton biomass) to each parameter were calculated according to:

$$S_c = \frac{\Delta Y/Y}{\Delta X/X} \quad (3.8)$$

where Y is the model performance (RMSE) for the reference value X of the parameter, which has a variation ΔX generating a model performance variation ΔY .

2.8 Calibration and validation

In order to calibrate water temperature, multiple combinations of parameters based on the sensitivity analysis were manually carried out from 2009 to 2012 through an iterative trial and error process. A greater effort was concentrated in refining the value of the parameters which presented the highest sensitivity coefficients. The validation was performed for 2013. For both model calibration and validation, water temperature data provided bimonthly (most of the time) by station 426 at 0.5, 5 and 15 m depth were used.

No attempts were made to calibrate phytoplankton biomass because it was measured only occasionally. Simulated and observed phytoplankton biomass were compared to assess the performance of the ecological model.

3 RESULTS AND DISCUSSION

3.1 Sensitivity analysis of the hydrodynamic model

The inflow factor is the variable which most affects simulated water temperature (Table 3.2). Inflows are related not only to the water surface elevation, surface area, and lake volume, but also to the heat budget since they have a relevant role in mixing processes. The inflow contributes to turbulent mixing through its own kinetic energy and through an input of buoyancy. Inflow waters are often colder and more dense than surface waters in lakes and reservoirs, then inflow waters plunge to a depth where the densities of the lake and that of the incoming waters are equal and form internal currents (Ji, 2007). Thus, inflow affects the local mixing characteristics and influences the global lake stratification. Streambed slope influences the mean inflow velocity (Ji, 2007) and the level where the new layer is created (Hipsey *et al.*, 2014). The model is less sensitive to the stream half angle and the streambed drag coefficient.

Table 3.2 – Sensitivity coefficients, calibrated parameters and range values of the hydrodynamic model.

Input variable	Symbol	S_c	Calibrated value	Range values
Inflow factor		0.298	0.85	0.40 – 1.00
Wind factor		0.279	0.80	0.50 – 3.00
Mixing efficiency of shear production	C_s	0.205	0.06	0.06 – 0.30
Streambed slope	α	0.183	1.8°	0.01 – 2.0
Mixing efficiency of wind stirring	C_w	0.073	0.23	0.23 – 0.80
Maximum layer thickness	h_{max}	0.049	0.60 m	0.60 – 3.00
Light extinction coefficient	K_w	0.012	0.25 m ⁻¹	0.20 – 2.01
Minimum layer thickness	h_{min}	0.005	0.50 m	0.25 – 1.50
Streambed half angle	β	0.002	63.95°	63.95 – 86.62
Streambed drag coefficient	C_D	0.001	0.016	0.002 – 0.050

Wind factor presents a high sensitivity coefficient. Wind velocity is responsible for supplying the kinetic energy necessary for the deepening of the surface layer and for turbulent processes through internal waves. Previous studies have presented similar conclusions about wind velocity relevance *e.g.* Hetherington *et al.* (2015) results confirmed the importance of wind speed as driver of the thermal regime of Oneida Lake (United States). The hydrodynamic simulation is also considerably sensitive to the mixing efficiencies of shear production and of

wind stirring. It is probably because these parameters control surface and hypolimnetic mixing: the vertical mixing is generally caused by wind action on the surface, through which eddy turbulence is transmitted to the lower portion of the water columns by shear stresses (Ji, 2007).

The maximum layer thickness strongly controls the spatial resolution of the model. This is because layers merge and grow in thickness until they reach the upper limit, and then become split, and merge again, especially during turnover and in the mixed surface layer. The layer thickness allows a fine resolution occurring in the thermocline. The results of the sensitivity analysis indicate a less importance of the light extinction coefficient. This is because this value is used only for the initial condition of simulation. From then on, this coefficient is dynamically adjusted by the AED module to account for variability in the concentrations of algae, inorganic and detrital particulates, and dissolved organic carbon levels. When AED module is not coupled to GLM, it is expected a strong correlation between accuracy of K_w measurements and model performance in the prediction of water temperature. According to Bruce *et al.* (2018), it emphasizes the need to include measurements of K_w in routine lake monitoring.

3.2 Sensitivity analysis of the ecological model

Phytoplankton parameters related to growth, light, respiration, phosphorus and nitrogen were tested separately for cyanobacteria and the other groups. The sensitivity coefficient of each parameter is presented in Table 3.3.

Table 3.3 – Sensitivity coefficient and range values of the ecological model.

Input variable	Symbol	Cyanobacteria		Other groups	
		S_c	Range values	S_c	Range values
Growth rate at 20 °C	R_{gr}	0.003	0.25 – 1.60	1.772	0.20 – 1.84
Half saturation for light limitation	I_K	1.451	15 – 180	0.699	20 – 250
Loss fraction due to respiration	k_{fres}	0.005	0.25 – 0.80	<0.001	0.25 – 0.80

The results showed different responses for each of the two phytoplankton groups, some of them antagonistic. The phytoplankton growth rate at 20 °C affected more other groups than cyanobacteria, which presented a low sensitivity coefficient. The algal growth rate is a complicated function of temperature, light, water movement, and nutrients and it increases

with temperature until an optimum is reached, after which further temperature increase will inhibit growth (Ji, 2007).

Cyanobacteria was most sensitive to the half saturation constant for light limitation. Other groups were also sensitive to this parameter. Light is generally regarded as one of the main factors regulating phytoplankton growth. The parameter related to respiration, loss fraction due to respiration, slightly affected cyanobacteria and no sensitivity was observed for other groups. The same was observed for lake Paranoá (Brazil) (Barbosa, 2015).

Similarly of the results presented here, a sensitivity analysis of ecological modelling in lake Paranoá verified that phytoplankton growth rate at 20 °C and light limitation of growth were sensitive for cyanobacteria and chlorophyceae (Barbosa, 2015). Phytoplankton biomass in Prospect reservoir (Australia) was highly sensitive to parameters that directly alter growth rates (such as R_{gr}) (Schladow and Hamilton, 1997).

3.3 Performance of the hydrodynamic model

The best simulation performance of the hydrodynamic model was achieved by applying the calibrated parameters shown in Table 2. Model results demonstrated a robust agreement between simulated and observed water temperature at 0.5 m depth, 5 m depth and 15 m depth (Figure 3.4). The model successfully simulated the water temperature profile and was able to reproduce the observed data (Figure 3.5).

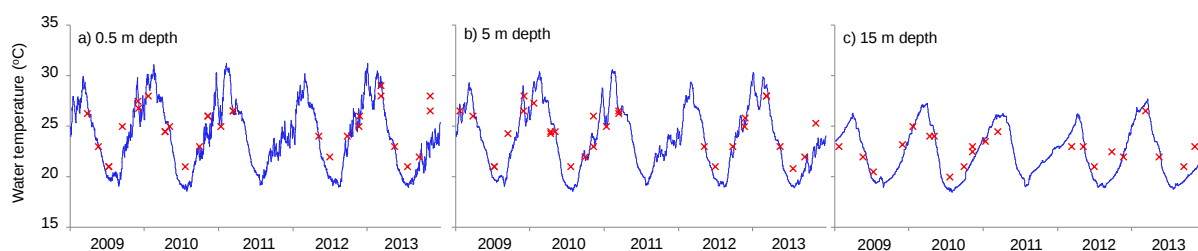


Figure 3.4 – Water temperature simulated (blue line) and observed (red marker) at a) 0.5 m depth, b) 5 m depth and c) 15 m depth.

Considering water temperature measured at different depths and simulated water temperatures, linear correlation coefficient is 0.90 ($n = 65$, $p\text{-value} < 0.05$) for the calibration period (Figure 3.6a) and 0.89 ($n = 17$, $p\text{-value} < 0.05$) for the validation period (Figure 3.6b). During calibration period (from 2009 to 2012), the RMSE between modeled and observed water temperature is 1.30 °C; and during validation period (2013) is 2.08 °C. Those values are

in accordance with RMSE reported by other studies conducted in different lakes using 1D models (Table 3.4).

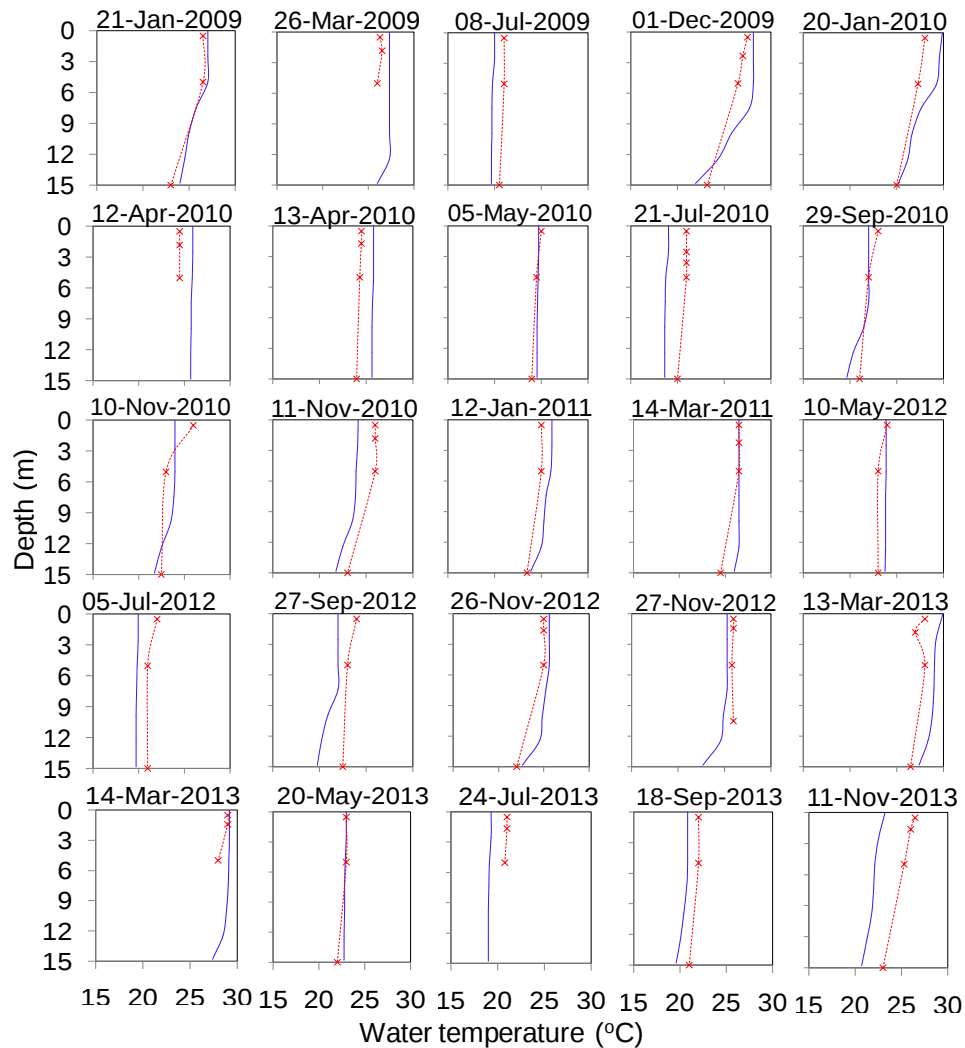


Figure 3.5 – Simulated (blue line) and observed (red markers) water temperature profiles of Serra Azul reservoir.

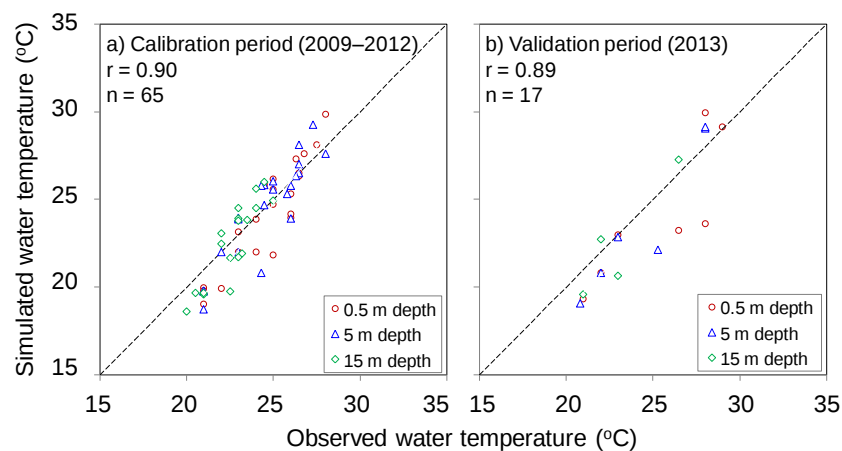


Figure 3.6 – Simulated and observed water temperature at 0.5 m, 5 m and 15 m depth for a) the calibration period (2009–2012) and for b) the validation period (2013). Dashed line represents the equality between simulated and observed values.

Table 3.4 – RMSE reported for different lakes using 1D models.

Lake	Depth	RMSE (°C)	Reference
32 lakes around the world	Profile	1.34	Bruce <i>et al.</i> , 2018
Oneida Lake (USA)	2 m; 10 m	1.51; 0.96	Hetherington <i>et al.</i> , 2015
Lake Bourget (France)	Epi-; Hypolimnion	1.5; 0.5	Vinçon-Leite <i>et al.</i> , 2014
Bautzen reservoir (Germany)	Surface; Bottom	1.25; 2.30	Kerimoglu and Rinke, 2013
Lake Ammersee (Germany)	Different depths	1.34	Weinberger and Vetter, 2012

At first, it was hypothesized that the differences between model simulation and observed measurements are caused by uncertainty in the input data. Deviations could be attributed to the simplification adopted to convert the eight tributaries into a single inflow and to meteorological data measured 11 km far from the reservoir which should lead to errors in model simulation of water temperature and evaporation rates. However, Bruce *et al.* (2018) verified that uncertainty in forcing data was not as significantly related to model performance as expected when they simulated 32 lakes using GLM. Differences between simulated and observed water temperature can be mainly attributed to (1) challenges for the parameterization; (2) the assumption of a unique value for each parameter during all the simulation period, disconsidering their seasonal variability and; (3) the 1-D nature of the model which implicitly assumes that the mixing within the lake can be constrained by processes acting in the vertical and that processes which vary in the horizontal, such as the degree of upwelling of the thermocline, have minimal impact on vertical transport (Bruce *et al.*, 2018). Concerning the reason number 2 (unique value for parameters), aiming at minimizing this type of limitation, Gaudard *et al.* (2016) proposed a method of adding a seasonal component in the parameterization that led to improved accuracy in the prediction of deep mixing in another hydrodynamic model.

3.4 Performance of the ecological model

Phytoplankton dynamics were represented reasonably well although the parameters of the ecological model were not calibrated. Despite the model overestimated cyanobacteria peaks (Figure 3.7a) and underestimated other groups biomass (Figure 3.7b), it is possible to state that it captured the order of magnitude of phytoplankton biomass and the seasonal pattern was mostly captured.

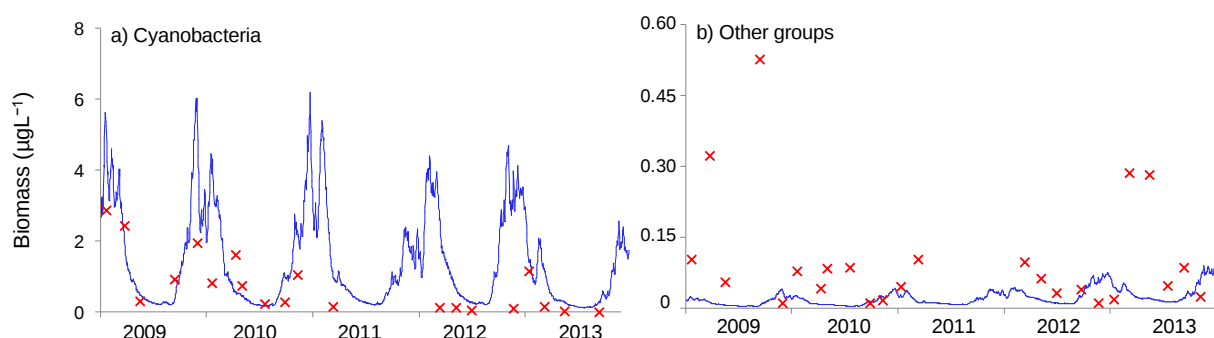


Figure 3.7 – Simulated daily mean chlorophyll-a concentration at Secchi depths (from 1.4 m depth until 5 m depth) (blue line) and observed field data (red marker) for a) cyanobacteria and b) other groups.

Linear correlation coefficient for cyanobacteria is 0.55 ($n = 20$, $p\text{-value} < 0.05$) (Figure 3.8a) and for other groups is 0.29 ($n = 24$, $p\text{-value} < 0.05$) (Figure 3.8b). For other groups, four field data is higher than $0.15 \mu\text{gL}^{-1}$ and those are not showed in Figure 3.8b in order to facilitate visualization of the data. The RMSE of cyanobacteria biomass is $1.52 \mu\text{gL}^{-1}$ and of other groups is $0.15 \mu\text{gL}^{-1}$. Those errors are high regarding the average observed biomass for each group: $0.76 \mu\text{gL}^{-1}$ for cyanobacteria and $0.10 \mu\text{gL}^{-1}$ for other groups.

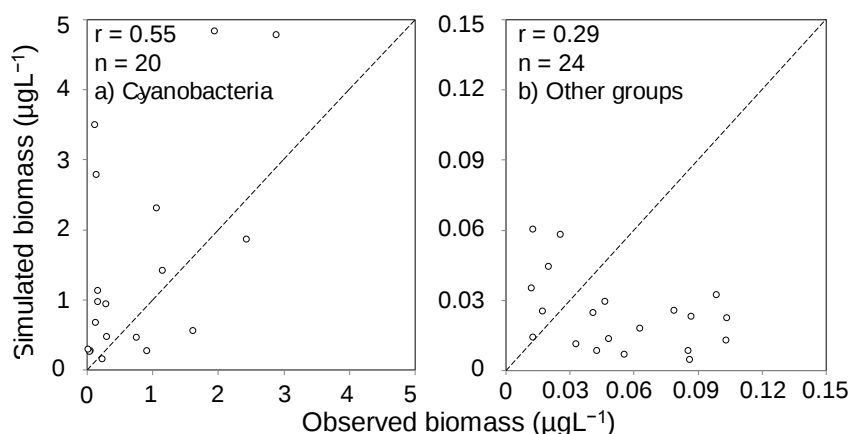


Figure 3.8 – Simulated and observed chlorophyll-a concentration at Secchi depths (from 1.4 m depth until 5m depth) from 2009 to 2013 for a) cyanobacteria and b) other groups.

Simulation of an average cyanobacteria biomass of $1.6 \mu\text{gL}^{-1}$ in lake Paranoá (Brazil) using GLM-AED found RMSE of $12 \mu\text{gL}^{-1}$ (Barbosa, 2015). Gal *et al.* (2009) found r^2 between 0.10 and 0.50 for simulated phytoplankton biomass at different depths in Lake Kinneret (Israel) (in this study, r^2 for cyanobacteria is 0.30 and r^2 for other groups is 0.08). In Prospect Reservoir (Australia), the mean difference between measured and simulated phytoplankton biomass was 24.1% of the measured chlorophyll-*a* concentration, with a standard deviation of 19.7% (Schladow and Hamilton, 1997). Silva *et al.* (2016) found r of 0.89 for the calibration

period and of 0.82 for the validation period in phytoplankton biomass simulation in Lake Pampulha (Brazil).

The outputs presented here were the result of using default parameters whose values were identified from literature. The ecological model has potential to be improved through calibration, since most of default parameters here applied have been empirically estimated in temperate regions and may present substantial variations in tropical environments. Furthermore, few observed data were available to assess model performance. Accurate simulation of the biological variables is very challenging when compared to hydrodynamic ones due to the complexity of the interactions affecting species succession and biotic dynamics, the large number of known (and unknown) processes affecting their biomass, timing of blooming, and vertical distribution of each phytoplankton group (Gal *et al.*, 2009).

3.5 Thermal dynamics

The model reproduced the seasonal variability in thermal profiles including stratification and mixing events (Figure 3.9). From simulated water temperature profiles, Serra Azul reservoir presents a warm-monomictic behavior with an annual autumn-winter water column turnover. The reservoir was thermally stratified (when the difference between surface and bottom temperature is greater than 1.0 °C) with a thermocline starting at approximately 1 m depth (Table 3.5). Mixing events occur from June to August (Table 3.5), when a homogeneous water column is characterized by lower temperatures (around 19.0 °C) (Table 3.5). The modelled thermal profile indicates a seasonal behavior similar with those reported for other water bodies in tropical regions (Valdespino-Castillo *et al.*, 2014; Curtarelli *et al.*, 2015; Barbosa *et al.*, 2013).

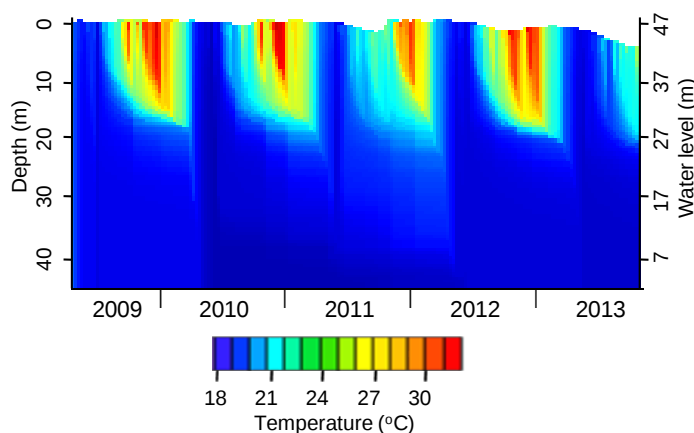


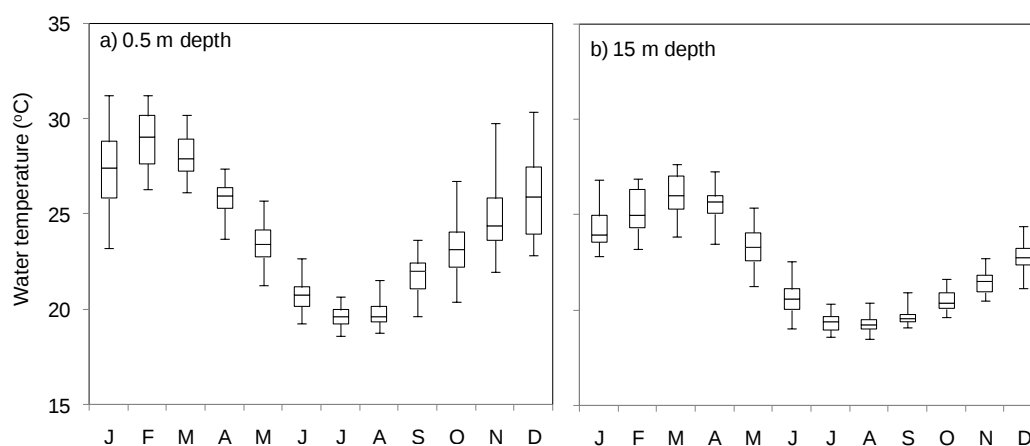
Figure 3.9 – Simulated water temperature in Serra Azul reservoir.

Table 3.5 – Thermal characteristics of the hydrodynamics in Serra Azul reservoir.

	2009	2010	2011	2012	2013
Epilimnion depth (m) ¹	10.6 (4.5)	13.5 (4.5)	16.0 (7.6)	13.9 (8.8)	14.2 (4.7)
Starting metalimnion depth (m)	3.7	3.8	1.2	1.2	1.2
Ending metalimnion depth (m)	20.5	31.0	32.1	40.8	25.9
Thermocline thickness (m) ¹	1.8 (2.7)	3.0 (3.2)	2.5 (3.0)	3.1 (3.2)	2.6 (3.0)
Mixing start date	May, 1 st	Jun, 08 th	Jul, 25 th	Jun, 25 th	Jun, 25 th
Mixing final date	Sep, 08 th	Aug, 21 st	Aug, 06 th	Aug, 31 st	Aug, 24 st
Water temperature in mixing (°C) ¹	20.8 (1.5)	19.0 (0.4)	19.1 (0.3)	19.3 (0.3)	19.1 (0.3)
Maximum water temperature (°C)	30.0	31.6	31.2	31.0	31.3
Minimum water temperature (°C)	19.0	18.5	18.5	18.7	18.8

¹ Mean value (standard deviation)

Simulated water temperature ranged from 18.6 °C in July to 31.2 °C in January at 0.5 m depth (Figure 3.10a) and from 18.5 °C in August to 27.7 °C in March at 15 m depth (Figure 3.10b). Water temperature at 0.5 m and 15 m depth followed seasonal changes, but surface water temperature is more variable than in the bottom layer. All simulated years showed a similar pattern over the study period.

**Figure 3.10 – Simulated water temperature at a) 0.5 m depth and b) 15 m depth.**

3.6 Phytoplankton dynamics

The model simulated the dynamics of cyanobacteria and other phytoplankton groups which presented a seasonal pattern over the simulated period. Phytoplankton population was dominated by cyanobacteria throughout the study period and its biomass was about 100 times greater than the other groups (Figure 3.11). It is possibly due to its higher maximum growth rate and to its several types of adaptive strategies that allow them to dominate over other phytoplankton groups.

Chlorophyll-a concentrations ranged from 0 to $8.4 \mu\text{gL}^{-1}$ during the studied period. As an overall, peaks of phytoplankton (around $6 \mu\text{gL}^{-1}$) are observed once a year (mainly from November to February), during the warm wet season, characterized by higher water temperature and thermal stratification of the water column. During this season, phytoplankton is concentrated in surface layers, until 15 m depth (Figure 11). In bottom layers, the absence of light inhibits phytoplankton growth. Mixing events do not provide favorable conditions to phytoplankton organisms since they are associated with very low biomass concentrations nearly $0.7 \mu\text{gL}^{-1}$.

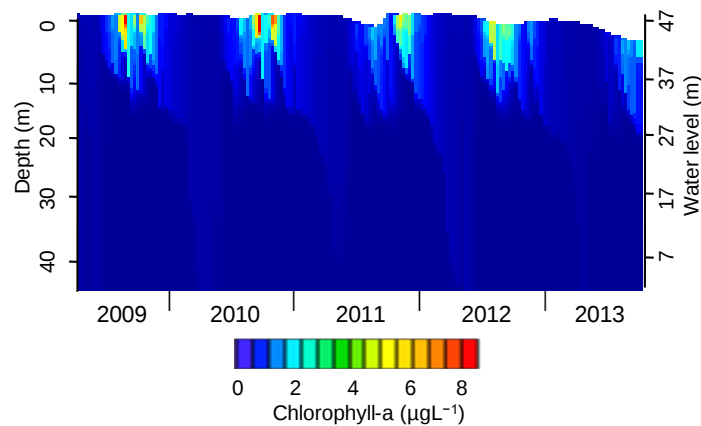


Figure 3.11 – Simulated chlorophyll-a concentration in Serra Azul reservoir.

Modelling results indicate that the factors with the greatest influence on phytoplankton biomass are the events of mixing and thermal stratification of the water column, promoting a seasonal pattern of concentrations and distribution. Field data is required to confirm this evidence and to give more confidence to the model through a robust calibration.

4 CONCLUSIONS

An one-dimensional hydrodynamic model coupled with an ecological one (GLM-AED) was used to simulate water temperature and phytoplankton dynamics in a tropical water supply reservoir for the period 2009–2013. Despite the simplifications in system description and in input data, the model describes the most important hydrodynamic and the main biological processes of the phytoplankton in an integrated way and provides reliable results for thermal simulations and represents the main aspects of the physic and ecological dynamics of phytoplankton.

GLM meets the requirements of properly representing water temperatures and reproducing thermal stratification and mixing events. As in many tropical reservoirs, Serra Azul exhibits a warm-monomictic water circulation pattern. It can be noticed that the sensitivity analysis confirms that inflow and wind speed are very sensitive input variables, highlighting the importance of high-quality input data. Furthermore, the hydrodynamic model was successfully calibrated and validated.

For the ecological model, the sensitivity analysis evidenced that the phytoplankton biomass was most sensitive to the parameters related to growth and light limitation. Phytoplankton concentrations and distribution are driven mainly by thermal stratification and mixing events, which are closely related to meteorological and hydrological patterns. This study improved the current understanding of the hydrodynamic and ecological processes in the reservoir and provides, on the local scale, an important background data for the water managers to better understand the complex dynamics of in-lake processes. Our results also contribute to the application of a useful tool for the investigation of the physical and ecological processes that occurs in lakes and reservoirs.

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CAPÍTULO 4

Modelling drought impacts on the hydrodynamics of a tropical water supply reservoir

ABSTRACT

Changes in meteorological patterns to drier conditions alter the hydrodynamic processes of reservoirs and can disrupt their thermal structure. The aim of this study was to assess the impacts of a severe drought period on a tropical reservoir. One-dimensional hydrodynamic modelling was applied from 2009 to 2016 in order to simulate the thermal dynamics of the highly drought-impacted Serra Azul reservoir, a drinking water supply reservoir located in southeast Brazil. After calibration ($r = 0.87$, RMSE = 1.36 °C) and validation of the model ($r = 0.92$, RMSE = 2.08 °C), physical indicators were calculated to evaluate the stratification and mixing processes in the water column. Results showed that the reservoir hydrodynamics was affected by the drought period through an increase in mixing frequency and duration. Negative implications on water quality are expected from drought impacts on the reservoir hydrodynamic. This study pointed out the need for managers to predict impacts of extreme meteorological events in reservoirs dynamics, particularly when they are part of drinking water systems.

Keywords: lentic water bodies; General Lake Model (GLM); thermal dynamics; physical indicators; water column mixing.

1 INTRODUCTION

Water supply systems, including those which rely on reservoir storage, are stressed by population growth, unsustainable consumption pattern and poor management practices. These systems may be further impacted by extreme hydrological and meteorological events (Raje and Mujumdar 2010), such as droughts. Drought reduces water availability, increases turbidity, causes bad taste and smell (Wright et al. 2014). Drought relationship with waterborne diseases has also been reported and is expected to increase with water quality deterioration (Cann et al. 2013). Drought impacts on water quality are, nevertheless, very variable in time and space and dependent on water body type and its catchment, geographic location, local climate and precedent meteorological events (Mosley 2015). In lentic systems such as lakes and reservoirs, water quality is highly dependent on hydrodynamic processes, such as thermal stratification and mixing conditions. Thus, in order to prevent or manage adverse impacts caused by drought on water supply reservoirs it is very important to investigate its impacts on the hydrodynamics of such systems.

Thermal stratification and mixing of the water column in lentic systems are controlled, in the one hand, by meteorological forcing at water surface (Stefan et al. 1996), including wind velocity, air temperature and rainfall, and in the other hand, by their geometry (depth, length, width), inflow and outflow discharges and hydraulic residence time (Ji 2007). In turn, the hydrodynamics of lentic water bodies highly affects the vertical distribution of nutrients, contaminants, dissolved oxygen, suspended matter and phytoplankton. Hence, reservoirs are especially sensitive to the conditions of their environment.

Over the last decades there has been an increasing amount of studies dealing with the effects of drought in lentic freshwater systems, mostly in North America, Europe, and Australia (Mosley 2015). These studies highlighted that periods of low rainfall and low discharge lead to a decrease of water level which may affect the thermal structure of the water column (Zohary and Ostrovsky 2011) and alter the heat budget and hydrodynamics in lakes and reservoirs. A water level decrease of more than 15 m in Karaoun reservoir (Lebanon) between June and October, 2012, weakened the thermal stratification and led to the mixing of the water column (Fadel et al. 2015). A drawdown of 15 m in Lake Arancio (Italy) during summer 2001 caused a collapse of the thermocline and full mixing of the water column (Naselli-Flores and Barone 2005). In Valle de Bravo reservoir (Mexico) the water level decreased 10 m during the drier years of 2006–2007 and resulted in the intensification of mixing events (Valdespino-Castillo et al. 2014). Despite the above mentioned examples, existing literature about the impacts of water level decrease has focused mainly in shallow lakes and studies of systems located outside temperate regions are particularly scarce (Zohary and Ostrovsky 2011; Mosley 2015).

Water level decrease in lentic systems may result in water quality deterioration (Naselli-Flores and Barone 2005) which is mainly related to (1) the increase of salinity, nutrient concentration (Li et al. 2017) and conductivity (Costa et al. 2016); (2) the decrease of water transparency (Jeppesen et al. 2015) through resuspension of particulate material (Zohary and Ostrovsky 2011) and; (3) trophic state changes impacting phytoplankton dynamics (Bouvy et al. 2003; Medeiros et al. 2015) and zooplankton abundances (G.-Tóth et al. 2011). In Brazil, a study involving 40 ponds in a semi-arid region showed that the reduction in water level caused by drought was responsible for eutrophication processes, leading to a higher biomass and dominance of cyanobacteria (Brasil et al. 2016). According to Mosley (2015), additional studies about drought impacts on water quality of freshwater systems in Asia, South America

and Africa would be beneficial in view of the few published literature until now. Besides, the vulnerability of developing countries to extreme weather events is very high due a lack of resources, infrastructure and disaster-preparedness programs (Watson et al. 2007). Cann et al. (2013) warn that even high-income countries are not well prepared to cope with some extreme weather events.

In Brazil, from 2013 to 2016, a severe drought occurred in the southeastern region, a very populated area with the three largest Brazilian metropolitan areas. Seventy years of hydrological monitoring revealed that the lowest rainfall volumes and the lowest streamflow were recorded during this period in the region (CPRM 2017). The year 2014 was the driest, with an annual rainfall depth return period estimated in 100 years (ANA 2014). In this context, some of the largest reservoirs reached the dead storage volume and there was a deficit in the drinking water supply in several cities.

Assuming that a drought period can disrupt the thermal behavior of a water supply reservoir located in a tropical region, this study aimed (1) to assess the impact of a severe drought on the mixing and stratification processes in a reservoir through mathematical modelling and hydrodynamics indicators and (2) to envisage possible consequences to the water quality and water supply due to changes in the reservoir hydrodynamics under drought conditions.

2 MATERIAL AND METHODS

2.1 Study site

Serra Azul reservoir is located in Belo Horizonte Metropolitan Region (BHMR), Minas Gerais, southeastern Brazil (19.97° S, 44.34° W), at elevation 760 m. The reservoir has a surface area of 9.11 km², a volume of 8.16 x 10⁷ m³, with average depth of 9.0 m and 47.3 m of maximum depth at its full capacity (Fernandes 2012). Its hydraulic residence time is 375 days. The reservoir, built in 1981, is part of the water supply system of the BHMR. It is operated by the Minas Gerais State Water Utility (COPASA) and supplies 0.8 m³s⁻¹ of drinking water to 800,000 inhabitants in the BHMR, nearly 16% of the total metropolitan population.

The Serra Azul reservoir has eight tributaries. Agriculture, mining and urbanization activities are present in its catchment, which has an area of 262 km². The surrounding area of the

reservoir is owned by COPASA, as a special protection area of 30 km², covered by native vegetation. The climate in the region is humid sub-tropical (Cwb) (Köppen 1948) with a wet season from October to March and a dry season from April to September.

The mean annual rainfall between 1961 and 1990 was 1,427 mm according to data provided by the National Institute of Meteorology (INMET). In 2014, 2015, 2016 and 2017 it was registered 953 mm, 1,209 mm, 1,372 and 1,020 mm, respectively. The reservoir volume reached 5.2 % of its total storage capacity in November 2014 (Figure 4.1).

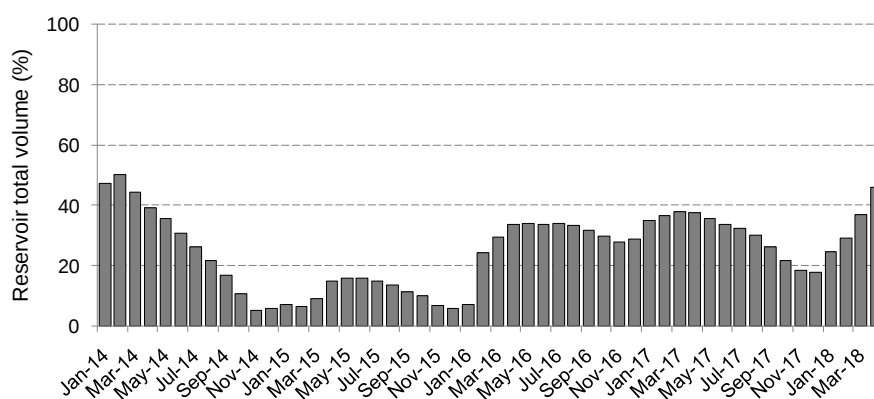


Figure 4.1 – Serra Azul volume from 2014 to 2018 in percentage of its total storage capacity ($8.16 \times 10^7 \text{m}^3$) (COPASA 2018).

2.2 Hidrological, meteorological and water quality data

The data of five monitoring stations were used in the present study (Figure 4.2, Table 4.1). The reservoir data were collected at two COPASA stations: station 426 located in the body of the reservoir, 200 m upstream from the water intake tower and; station 371 located in a reservoir arm, near a tributary entrance. A bi-monthly sampling has been performed since 1981 in station 426 and since 1990 in station 371.

The main tributary (Serra Azul creek) is monitored at 2 stations of the Brazilian Geological Service (CPRM): Jardim station and Serra Azul station. At Jardim station, daily discharge has been recorded since 1979 and at Serra Azul station, daily water temperature has been recorded since 1987. The Florestal meteorological station (INMET), located 11 km far from the reservoir, automatically measures meteorological data every hour since 2008. Hourly meteorological data and daily hydrological data measured by Florestal and Serra Azul stations from 2009 to 2016 are presented in Figure 4.3. Annual mean values and standard deviation are shown in Table 4.2.

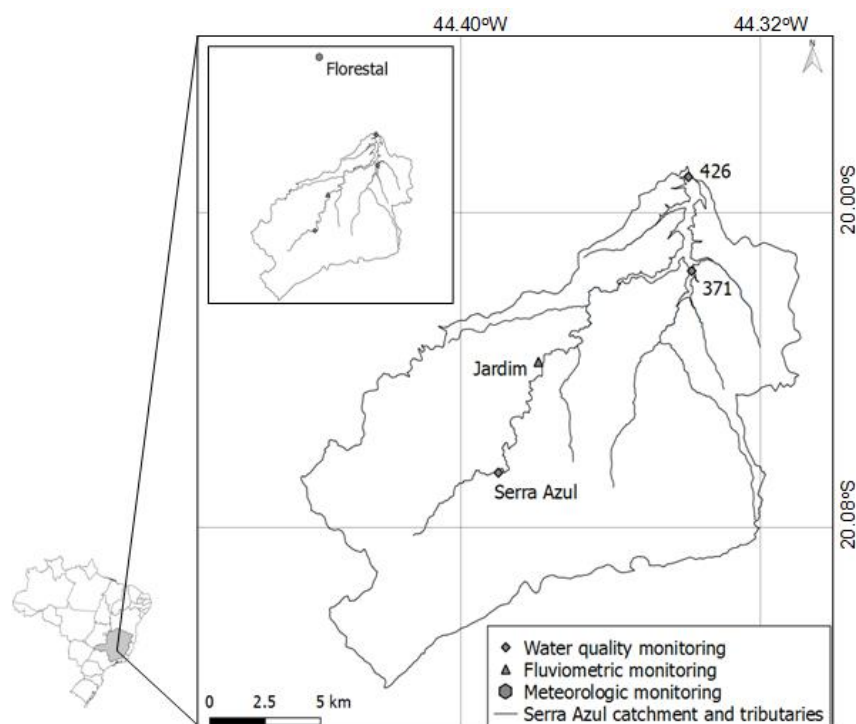


Figure 4.2 – Serra Azul reservoir, its tributaries and monitoring stations: 426, 371, Jardim, Serra Azul and Florestal.

Table 4.1 – Monitoring stations characteristics.

Name	Code	Data type	Operated by	Latitude	Longitude
426	-	Water quality	COPASA ^a	19.97° S	44.34° W
371	-	Water quality	COPASA	20.02° S	44.35° W
Jardim	40811100	Fluviometric	CPRM ^b	20.05° S	44.41° W
Serra Azul	02044054	Water quality	CPRM	20.09° S	44.43° W
Florestal	86798	Meteorologic	INMET ^c	19.88° S	44.42° W

^a Minas Gerais State Water Utility

^b Geological Survey of Brazil

^c National Institute of Meteorology

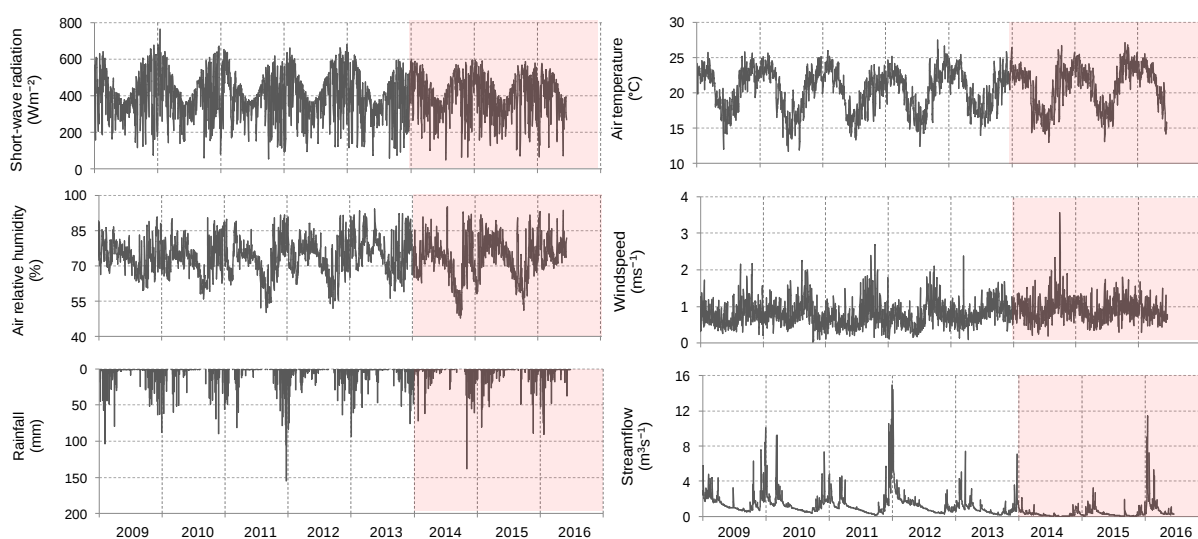


Figure 4.3 – Meteorological (Florestal station – INMET) and hydrological (Jardim station – CPRM) variables: short-wave radiation (Wm^{-2}), air temperature ($^{\circ}\text{C}$), air relative humidity (%), wind speed (ms^{-1}), rainfall (mm) and streamflow (m^3s^{-1}).

Table 4.2 – Annual mean values (standard deviation) of meteorological (Florestal - INMET) and hydrological (Serra Azul - CPRM) variables from 2009 to 2016.

	2009	2010	2011	2012	2013	2014	2015	2016
Short-wave radiation (Wm^{-2})	844 (1179)	869 (1194)	839 (1163)	879 (1213)	795 (1117)	843 (1116)	820 (1091)	769 (1042)
Air temperature ($^{\circ}\text{C}$)	20.8 (5.5)	20.2 (6.3)	20.1 (6.0)	20.4 (6.0)	20.5 (5.7)	20.4 (6.3)	21.2 (6.0)	21.0 (5.9)
Air relative humidity (%)	74 (19)	73 (20)	73 (21)	73 (21)	76 (21)	72 (23)	73 (22)	75 (21)
Windspeed (ms^{-1})	0.8 (0.9)	0.8 (0.9)	0.7 (0.9)	0.7 (0.9)	0.8 (0.8)	0.9 (0.9)	0.9 (0.8)	0.9 (0.8)
Total Rainfall (mm)	1672	1383	1652	1130	1178	953	1209	1372
Streamflow (m^3s^{-1})	1.7 (1.2)	1.4 (1.0)	1.4 (1.5)	1.5 (1.7)	0.9 (0.8)	0.3 (0.3)	0.4 (0.4)	1.1 (1.5)

2.3 Data processing

Serra Azul reservoir morphometry was obtained from Fernandes (2012): the bathymetric mapping was performed in 2009 using the SonarLite echo sounder, which detects depths between 0.3 m and 75 m with an accuracy of ± 0.025 m. Vertical profiles of temperature and salinity in the reservoir water column were obtained from station 426 data. Hourly meteorological data were obtained from Florestal station. Input missing data were replaced by linear interpolation. Of the eight Serra Azul reservoir tributaries, only Serra Azul creek has a fluviometric station (Jardim station). Daily inflow into Serra Azul reservoir was computed from Jardim station data increased by the ratio between the reservoir and Jardim station drainage area. Serra Azul station provided water temperature of Serra Azul creek.

The inflow salinity is not monitored and it was estimated from conductivity according to Bäuerle et al. (1998). Considering that few conductivity data were available and that salinity in the reservoir is very low (below 0.02 pss), its mean value (0.01 pss) was adopted during the study period.

A single outlet (Figure 4.2) is used for the water uptake and for maintenance of the residual flow downstream the reservoir dam. The outflow was estimated from the monthly water withdrawal. Due to the drought, no overflow through the reservoir spillway was observed during the study period.

Water temperature measured at different depths (mostly at 0.5, 5.0 and 15.0 m) in the reservoir was used for comparison with model results. Most of the time, bi-monthly measurements are available. Water level of Serra Azul reservoir was computed from 2014 to

2016, using the monthly data of the reservoir volume provided by COPASA. These values were used to compare with simulated water levels.

2.4 The one-dimensional hydrodynamic reservoir model

The one-dimensional-vertical General Lake Model (GLM, version 2.2), was used for modelling Serra Azul reservoir hydrodynamics. It has been developed by the Aquatic EcoDynamics Research Group team at University of Western Australia (Hipsey et al. 2014). It is an open-access model which simulates vertical profiles of temperature, salinity and density by accounting for the effect of inflows and outflows on the water balance, in addition to surface heating and cooling, energy balance and vertical mixing.

The reservoir is discretized by a series of horizontal layers numbered from the bottom to the surface, describing the variation in the water column properties. Each layer is a control volume that has homogeneous properties, including density which is computed from salinity and temperature. The layer thickness can change by contracting and expanding in response to inflows, outflows, mixing with adjacent layers, and surface mass fluxes. When sufficient energy becomes available to overcome density differences between the adjacent layers, they merge, accounting for the mixing process. Layer thickness is adjusted throughout the water column by the model, with fine resolution in the thermocline depth and thicker layers where mixing is occurring (Hipsey et al. 2014).

Stratification and mixing are calculated from energy balance. Mixing occurs if the available kinetic energy exceeds the potential energy required to keep the layers stratified. The available kinetic energy is calculated due to contributions from wind stirring, shear production between layers, convective overturn, and Kelvin-Helmholtz billowing. Mixing ends when internal kinetic energy is insufficient to overcome the stored potential energy. Then, the thermal stratification process begins (Hipsey et al. 2014). Mixing events are driven, to a large extent, by wind forcing over the reservoir. When the wind blows across the reservoir surface, a shear stress occurs on the water surface, which results in a momentum transfer from the air into the water (Ji 2007). Wind energy is converted into turbulence in the surface layer and is then transferred to the lower layer of the epilimnion by turbulent diffusion, until the thermal gradient dissipates the energy.

2.5 Modelling setup

Two simulation periods were performed: calibration period of 4 years from January 1st, 2009 to December 31st, 2012 and validation period of 3.5 years from January 1st, 2013 to June 22nd, 2016. Both periods were simulated at hourly time step, in order to compute daily water temperature which was compared to observed water temperature at the water intake tower (station 426). The initial conditions and input variables were based on available data as previously described. Model parameters were manually calibrated through an iterative trial and error process (Table 4.3) and then they were validated. Model performance was assessed by Pearson's correlation coefficient (r) and root mean squared error (RMSE) computed from simulated and observed water temperature and water level. Model performance in calibration and validation was assessed from the 7th month of simulation in order to reduce the effects of uncertainties in initial conditions.

Table 4.3 – Calibrated parameters and their range values.

Input variable	Symbol	Calibrated value	Range values
Inflow factor		0.85	0.40 – 1.00
Wind factor		0.80	0.50 – 3.00
Mixing efficiency of shear production	C_S	0.06	0.06 – 0.30
Streambed slope	α	1.8°	0.01 – 2.0
Mixing efficiency of wind stirring	C_W	0.23	0.23 – 0.80
Maximum layer thickness	$h_{\max}$	0.60 m	0.60 – 3.00
Light extinction coefficient	K_W	0.25 m ⁻¹	0.20 – 2.01
Minimum layer thickness	$h_{\min}$	0.50 m	0.25 – 1.50
Stream half angle	β	63.95°	63.95 – 86.62
Streambed drag coefficient	C_D	0.016	0.002 – 0.050

The statistical software R 3.2.5 was used to simulate and analyze modelling results through three packages: GLMr (Hipsey et al. 2014) used to run the model; GLMtools (Read et al. 2014) used to present simulation results; and Lake Analyzer (Winslow et al. 2016) used to calculate metrics of the reservoir physical state. The following reservoir hydrodynamic indicators were calculated: Schmidt stability, Wedderburn Number and Lake Number.

2.6 Hydrodynamics indicators

The Schmidt stability is a widely used indicator of the stratification process, indicating the resistance to mechanical mixing due to potential energy inherent in the stratification of the water column. It represents the required work per surface area to break the thermal

stratification and make the water column homogeneous in terms of density. The thermal stability through Schmidt's Stability Index, S_c , (Idso 1973) is calculated as:

$$S_c = \frac{g}{A_s} \int_0^{z_D} (z - z_v) \rho_z A_z \partial z \quad (4.1)$$

where g is the acceleration due to gravity, A_s is the surface area of the reservoir, A_z is the area of the reservoir at depth z , z_D is the maximum depth of the reservoir, z_v is the depth of the volume centre of the reservoir and ρ_z is the water density at depth z .

The Wedderburn Number (W_d) describes the likelihood of water upwelling under stratified conditions. For $W_d \leq 1$ there is a high probability that the thermocline will tilt to the surface at the upwind end of the reservoir and metalimnetic water will be entrained into the surface mixing layer, increasing it; otherwise ($W_d > 1$) the mixed layer will deepen slowly (Imberger and Patterson 1990). The Wedderburn number can be written as:

$$W_d = \frac{g' z_e^2}{u_*^2 L_s} \quad (4.2)$$

where $g' = g \times \Delta\rho/\rho_h$ is the reduced gravity due to the difference $\Delta\rho$ between the hypolimnion density (ρ_h) and epilimnion density, z_e is the depth of the mixed layer base, L_s is the reservoir fetch length and u_* is the water friction velocity due to wind stress.

The Lake Number (L_N) describes processes relevant to the internal mixing of lakes and reservoirs induced by wind forcing. As with W_d , lower values of L_N represent a higher potential for increased mixing, which increases the vertical flux of mass and energy across the metalimnion through the action of non-linear internal waves (Macintyre and Melack 2009). Lake Number is given by:

$$L_N = \frac{S_c(z_e + z_h)}{2\rho_h u_*^2 \sqrt{A_s z_v}} \quad (4.3)$$

where S_c is the Schmidt stability, z_e and z_h are the depths of the top and bottom of the metalimnion, respectively, ρ_h is the hypolimnion density, u_* is the water friction velocity due to wind stress, A_s is the surface area of the reservoir and z_v is the depth of the volume centre of the reservoir.

Besides the hydrodynamics indices described above, two metrics, proposed by Kerimoglu and Rinke (2013) were used to evaluate mixing events. The first metric quantifies mixing events by counting the number of days during which the difference ΔT between surface and bottom temperature is less than 0.5 °C. This metric is termed as “number of mixed days”. The second metric, which counts the number of distinct mixing events, i.e., number of episodes during a year, where homogenization ($\Delta T < 0.5$ °C) occurred following a stratified state ($\Delta T > 0.5$ °C) in daily resolution. This metric is termed as “number of mixing events”.

3 RESULTS

3.1 Model performance

The model succeeded in simulating the water level over the period when compared to available data (Figure 4.4). The RMSE between simulated and observed water levels between January 2014 and December 2016 was 1.75 m, approximately 7% of the total water level variation during this period. The reservoir was at its full capacity in June 2012. From 2014, the drought strongly affected the water level and it decreased by 25 m, reaching the lowest value (21.5 m) in January 2015, which corresponds to approximately 5% of the reservoir total volume. The water level started to rise with the wet season in 2016.

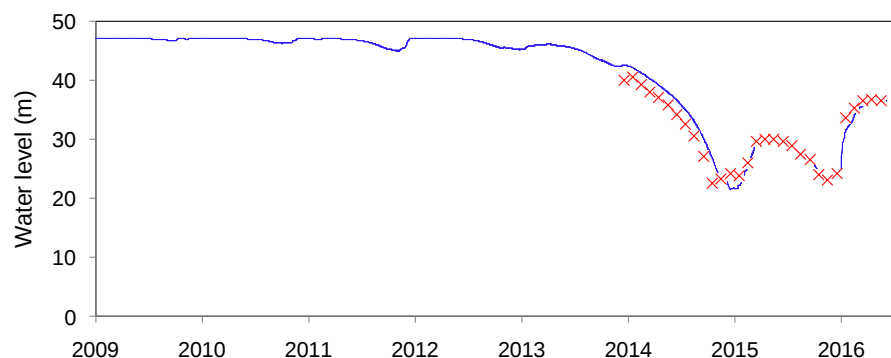


Figure 4.4 – Simulated (blue line) and observed (red marker) water levels in Serra Azul reservoir.

Model results demonstrated good agreement between simulated and observed water temperature at 0.5 m depth (surface waters) (Figure 4.5a) and 15 m depth (bottom waters) (Figure 4.5b). The RMSE between simulated and observed water temperatures was 1.36 °C for the calibration period (from January 1st, 2009 to December 31st, 2012) and 2.08°C for the validation period (from January 1st, 2013 to June 22nd, 2016). Pearson coefficient between

simulated and observed water temperatures was 0.92 ($n = 81$, $p\text{-value} < 0.05$) for the calibration period and 0.87 ($n = 47$, $p\text{-value} < 0.05$) for the validation period.

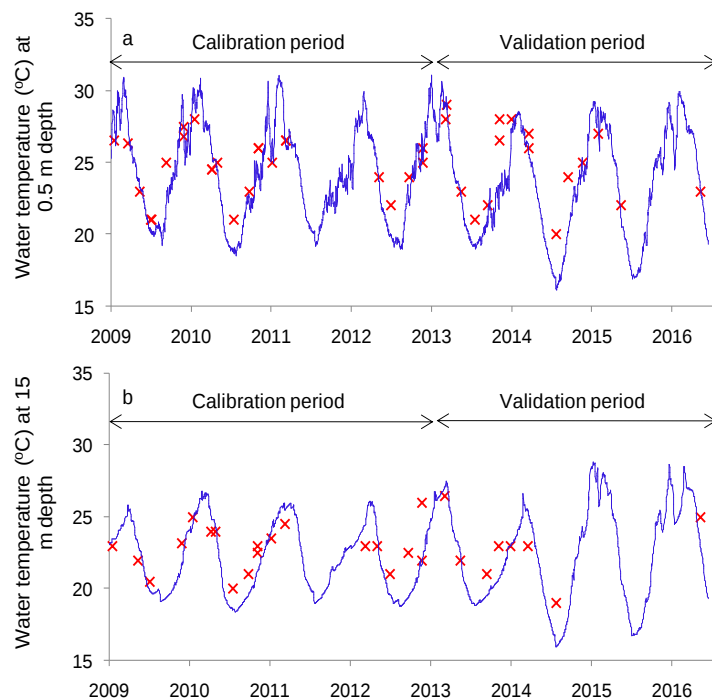


Figure 4.5 – Simulated (blue line) and observed (red marker) water temperature at (a) 0.5 m depth, and at (b) 15 m depth.

3.2 Drought impact on reservoir hydrodynamics

From 2009 to 2013, before the drought period, Serra Azul reservoir presented a warm monomictic behavior (Figure 4.6). During stratification events, the epilimnion extended from the surface to $14.3 (\pm 5.2)$ m depth (Table 4.4) and water temperature at the surface reached 31.7°C . In the metalimnion, the strongest thermal gradient region presented a mean thickness of $3.4 (\pm 2.9)$ m (Table 4.4). The temperature profile annually reached a uniform temperature from May to August (Table 4.4) and the reservoir was well mixed at a temperature of 20.1°C .

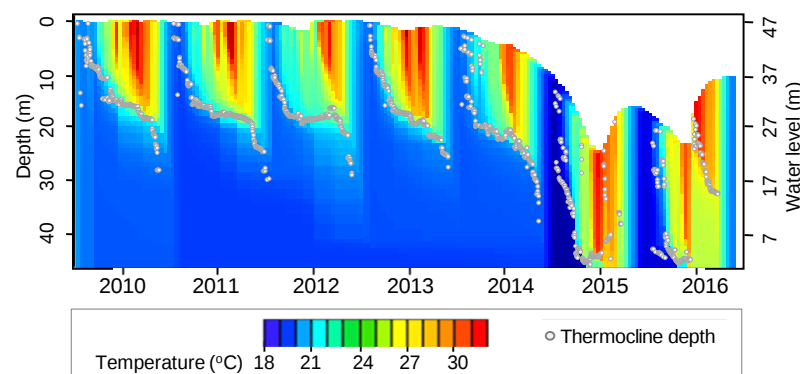


Figure 4.6 – Contour plot of simulated water temperature in Serra Azul reservoir.

Table 4.4 – Simulated hydrodynamics indicators of Serra Azul reservoir.

Hydrodynamics indicators	2010	2011	2012	2013	2014	2015
Epilimnion depth (m) ^a	13.5 (4.3)	15.6 (4.5)	14.2 (5.4)	14.8 (5.0)	15.6 (4.2)	16.6 (5.2)
Metalimnion thickness (m) ^a	4.1 (3.0)	3.1 (2.9)	4.0 (2.9)	3.3 (3.2)	3.4 (2.5)	2.1 (2.0)
Mixing start date	May, 21 st	Jun, 10 th	May, 16 th	May, 12 th	May, 27 th	Mar, 12 th
Mixing final date	Aug, 2 nd	Aug, 7 th	Aug, 25 th	Aug, 2 nd	Aug, 4 th	Aug, 16 th
Number of mixed days	71	57	86	82	79	150
Number of mixing events	2	2	6	3	4	9
Maximum water level (m)	47.0	47.0	47.0	46.1	42.6	30.7
Minimum water level (m)	46.3	45.0	45.2	42.3	21.7	21.5
Number of days in dead storage	0	0	0	0	109	365

^a Mean value for each year $\pm$ (standard deviation).

Water level (m) of dead storage is 32.0 m.

The drought period led to the disruption of the annual cycle of stratification. The thermal structure of the reservoir had shift from a monomictic behavior to a polymictic one. In 2015, the start of mixing period shifted from May to March and as a consequence with a higher number of mixing events. The number of mixed days per year increased from 75 days on average from 2010 to 2014, to 150 days in 2015 (Table 4.4). The number of mixing events was also much higher than in previous years, reaching nine events in 2015 (Table 4.4). The end of the mixing period seemed to be not affected. From 2014 to 2015, in parallel with the water level lowering, thermocline depth was deepened until the bottom of the reservoir (Figure 4.6). Thermocline deepening was a result of the increased turbulence in the surface layer and indicates upwelling of hypolimnion waters. In 2016, the reservoir started to recover its water level but the polymictic behavior remained.

In Serra Azul reservoir, there are three possible water withdrawals located at different depths of the water intake tower. Considering that the minimum level of the water withdrawal for human supply is 32 m, the water level was below this limit (dead storage), from September 2014 to February 2016 (Table 4.4). This period included a total of 513 days below the lowest withdrawal elevation, revealing the severity of the water scarcity. During this period, the water stored in the reservoir could only be withdrawn through pumping.

The drought affected the temperature difference between surface and bottom layers. According to simulated results, from 2009 to 2014, the mean difference between surface (depth 0) and 20 m depth was 4.0 °C; in 2015 the mean difference was 1.6 °C (Figure 4.7). Surface temperature was impacted in a lesser extent (reduction of 0.2 °C), while water temperature at 20 m depth increased by 2.3 °C, on average. This increase in water temperature of bottom layers indicates that heat transfer to deep layers was increased with the water level decrease.

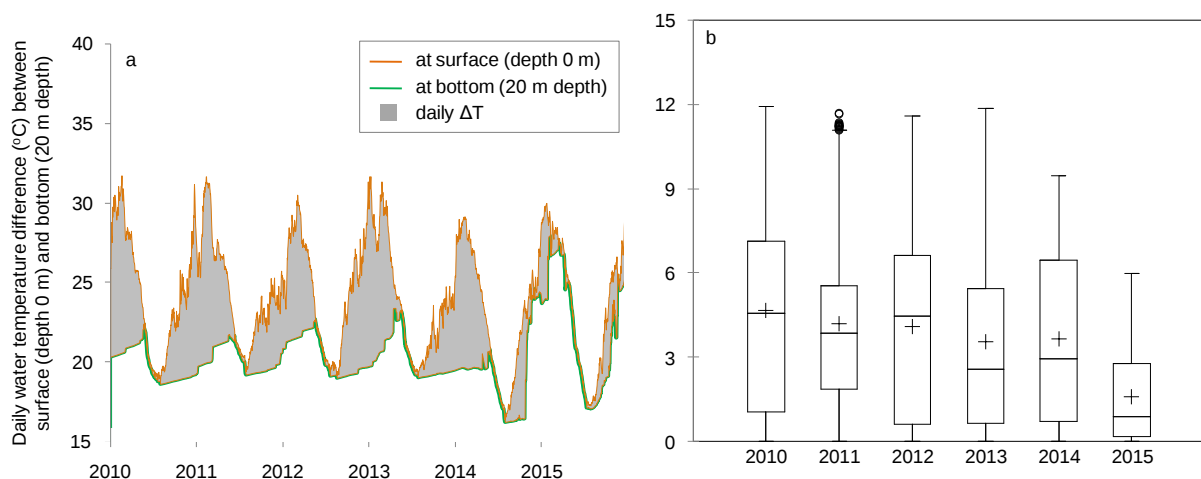


Figure 4.7 – Water temperature difference between surface (depth 0 m) and bottom (20 m depth) in Serra Azul reservoir: a) daily values and b) annual values. The horizontal line inside the box plot indicates the median, the cross marker indicates the mean, the boundaries of the box plot indicate the 25th and 75th percentiles and the circle indicates outliers. Whiskers indicate the maximum and minimum value.

The Schmidt Stability Index confirmed that when the reservoir is shallower, less work was required to mix the entire body of water, homogenizing the temperature. Schmidt Index computed with the simulated water temperature indicated that the strength of stratification decreased approximately by half during the drought period in 2015 and 2016 (Figure 4.8). Its mean annual value varied between 243 (± 178) Jm^{-2} and 270 (± 201) Jm^{-2} from 2010 to 2012 and between 120 (± 120) Jm^{-2} and 215 (± 185) Jm^{-2} from 2013 to 2015. Mean annual Schmidt Stability Index was correlated to mean annual streamflow ($r = 0.81$, $p = 0.05$, $n = 6$).

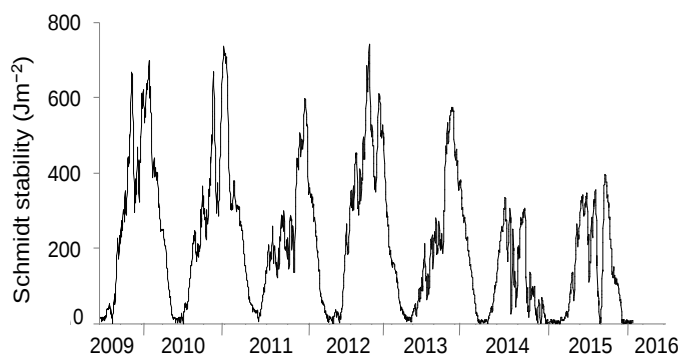


Figure 4.8 – Simulated Schmidt stability of Serra Azul reservoir.

The Wedderburn Number and Lake Number explain the potential for mixing events, predicting wind-induced internal waves. When mixing events were occurring, those indicators reached zero values. During stratification period, those indicators reached their maximum values. Mean annual Wedderburn Number was correlated to mean annual streamflow ($r = 0.99$, $p < 0.01$, $n = 6$).

Lake Number and Wedderburn Number presented a similar pattern every year, but the Wedderburn Number presents more variability when compared to the Lake Number. While W_d is dependent on the highly variable mixed layer depth, L_N relies on the depth of the metalimnion midpoint, which tends to reduce variability through averaging with the more stable base of the metalimnion (Read et al. 2011). Boxplots of the physical indicators for each year are summarized in Figure 4.9.

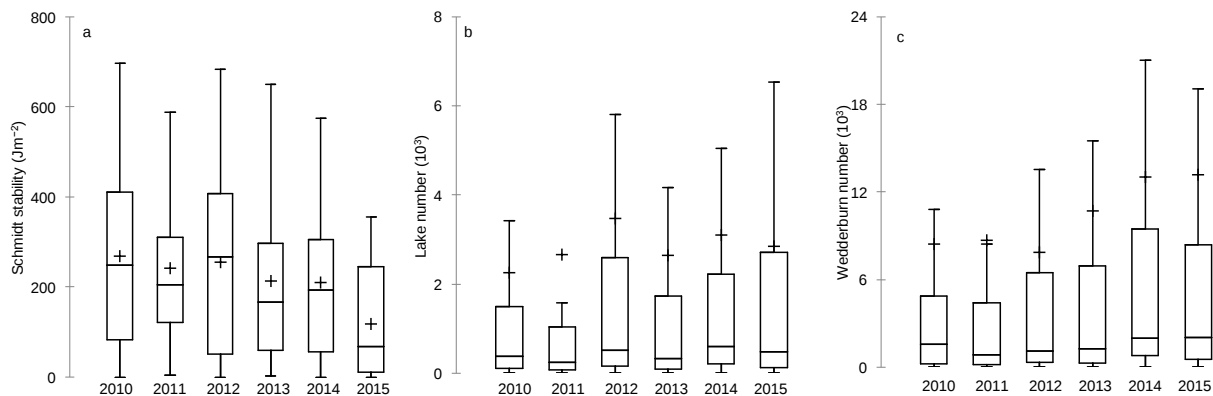


Figure 4.9 – Simulated hydrodynamics indicators of Serra Azul reservoir: a) Schmidt stability, b) Lake number, and c) Wedderburn number. The horizontal line inside the box plot indicates the median, the cross marker indicates the mean and the boundaries of the box plot indicate the 25th and 75th percentiles. Whiskers indicate the maximum and minimum value.

4 DISCUSSION

4.1 Model performance

One-dimensional vertical models have been widespread used and improved since 1970s to represent the vertical structure of lentic water bodies. They have been successfully applied to a large size range of lakes and reservoirs (Gal et al, 2003; Rigosi and Rueda 2012; Weinberger and Vetter 2012; Read et al. 2014; Vinçon-Leite et al. 2014). GLM was applied to more than 30 lakes and reservoirs and the prediction of temperature profiles demonstrated a robust fit with an average RMSE of 1.34 °C and r of 0.96 for two years of simulation (Bruce et al. 2018).

In Serra Azul reservoir, GLM performance was consistent with results reported in literature. The model successfully represented water temperature distribution across water column, as well as the mixing and stratification processes over a long continuous validation period of 42 months. The warm-monomictic behavior of Serra Azul reservoir was demonstrated

through modelling and it is in agreement with the behavior of most tropical lakes, which are warm-monomictic with a deep mixing on a seasonal cycle (Lewis 2000). Furthermore, in Serra Azul reservoir, water temperature data were quite scarce despite the long time series because monitoring campaigns were performed few times throughout a year. Hydrodynamic modelling allowed us to better understand the reservoir physical dynamics under normal meteorological and hydrological conditions and under drought conditions.

4.2 Drought impact on reservoir hydrodynamics

Water supply from Serra Azul reservoir was severely impacted during drought periods in 2014 and 2015. Drinking water to BHMR had to be supplied by other sources (other reservoirs and rivers) through the COPASA integrated water supply network and emergency measures which cost more than US\$ 32 million. Concerning drought impacts on reservoir hydrodynamics, stratification has weakened with increasing number of mixing events and mixing period duration being extended at the annual time scale; mean water temperature of the water column has risen with the heating of bottom layers through mixing. Schmidt Stability Index indicates that, when the reservoir is shallower, its thermal stability is reduced and the required energy to homogenize water temperature is smaller.

The impact of drought on reservoir hydrodynamics may be highly variable because it may be related to reservoir morphology, inflows and outflows, geographic location, local climate and precedent meteorological events. Regarding mixing regime, its impact seems consistent in the studied systems: drought increases lake polymicticity. An extreme reduction of water level may convert monomictic lakes into polymictic ones (Zohary and Ostrovsky 2011), as it was observed for Serra Azul reservoir. A 15 m decrease in the water level of Lake Arancio (Italy) resulted in lake destratification and in a conversion from a monomictic lake to a polymictic one (Naselli-Flores et al. 2003). Similarly, full mixing events were reported in Lake Hume (Australia) after its drawdown of 30 m (Baldwin et al. 2008).

On the other hand, many studies have found more intensive reservoir stratification due to reservoir water temperature increase which in turn, results from increasing on air temperatures and/or inflow temperatures during drought (Mosley 2015). In Diamond Valley Lake (United States), Schmidt Stability Index highlighted the more pronounced stratification of the lake during drought. Mean Schmidt Stability Index increased from 5173 Jm^{-2} in normal years to

6382 Jm⁻² in dry years (Gebremariam 2015), which is possibly due to the lower input of destabilizing energy from inflow.

And still there are cases where the water column temperature did not change under drought conditions. In Harlan County reservoir (United States) despite the reduction of water level caused by a reduction in precipitation during drought, water column temperature remained unaltered (Olds et al. 2011). This was presumed to be due to wind mixing and normal air temperature during summer season.

4.3 Risks on water supply due to drought

Many studies have presented common findings concerning physical-chemical variables of water quality in reservoirs under drought, despite their individual characteristics (geographic location, climate, landuse in the catchment, precedent or simultaneously events) (Mosley 2015). The more frequent and prolonged mixing events in Serra Azul reservoir may lead to higher turbidity caused by sediments resuspension. Increased turbidity was observed in Harlan County reservoir (USA) during drought years (2003–2006) caused by increased resuspension of benthic sediment by wind-driven waves (Olds et al. 2011). More frequent mixing events will probably enhance reaeration in deep layers which will be more oxygenated. On the other hand, in Serra Azul warmer hypolimnion, dissolved oxygen concentration may decrease due to its lower solubility under higher temperatures.

Dissolution and complexation of mineral and organic substances, mineralization of organic matter may be also favored due to the positive correlation between temperature and endothermic processes. As a direct effect higher concentrations of dissolved substances in water are expected. The reservoir may experience increased salinity, as observed in most lakes and reservoirs during droughts, which has been attributed to water volume reduction and to evapoconcentration (Flanagan et al. 2009; Mosley et al. 2012). An increased electrical conductivity is expected due to its significant correlation with water level reduction probably as a result of higher evaporation, as observed in Lake Doirani (Greece) (Jeppesen et al. 2015). Similarly, a 3-fold increase in mean electrical conductivity was observed in 40 man-made lakes in Brazil threatening the water supply during a dry season (Brasil et al. 2016).

Concerning nutrient concentrations, one can hypothesize that there will be an increase due to higher internal loading promoted by more frequent mixing and a concentration effect in the

smaller water volume. Nitrification is likely to reduce under low-oxygen conditions in hypolimnion and increased resuspension (Özen et al. 2010; Jeppesen et al. 2015).

Predicting drought impacts on the ecosystem is a challenging task in view of the complex interactions between physical-chemical and biological processes and the diverse influence of drought time-space scales (Costa et al. 2016). For instance, studies have been reporting contrasting dynamics of phytoplankton in lakes during extreme drought periods (Bouvy et al. 2003; Naselli-Flores and Barone 2005; Medeiros et al. 2015). More frequent vertical turbulent mixing may promote nutrient turnover in the water column, potentially enhancing primary productivity and leading to more abundant algal populations. Full mixing during a drought (2001) in Lake Arancio (Italy) triggered an immediate bloom of *Microcystis* (Naselli-Flores and Barone 2005). However, there are also examples of reduced phytoplankton biomass at low water level, such as in Lake Cruzeta (Brazil), where was observed a reduction in chlorophyll-a concentration (Medeiros et al. 2015). Considering that phytoplankton biomass is positively related to water column stability, the possible effects of drought on phytoplankton dynamics in Serra Azul reservoir are not clear.

From the above exposed, degradation of water quality may be expected in Serra Azul reservoir due to the important changes in its hydrodynamics caused by drought. Prolonged drought can result in a series of cascading consequences that present challenges to water quality management. In short and medium term, effective mitigation strategies of drought impact would include increase in water quality monitoring frequency, especially for early warning of cyanobacterial blooms whose dynamics is hard to predict. Diversifying water resource options should also be implemented (Khan et al., 2015). In long term, water utilities and regulators are recommended to perform planning reformulation; further invest in infrastructure, technology and long-term water quantity and quality monitoring. Existing time series need to be analyzed and combined to modelling approach in order to early predict drought effects on water quality and to support resilience strategies (Jeppesen et al. 2015; Mosley 2015).

5 CONCLUSIONS

Drought impacts on the hydrodynamics of a tropical reservoir, Serra Azul reservoir (Brazil), were investigated through mathematical modelling and hydrodynamic indicators derived from

modelling results. The one-dimensional vertical GLM model proved to be an efficient tool for simulating the thermal dynamics and water budget of Serra Azul reservoir. Mathematical modelling improved our understanding about the impacts of the greatest drought recorded in over 70 years on the reservoir hydrodynamics. Drought caused an extreme maximum water level drawdown (25.5 m) around 54% of the reservoir depth at full capacity and the effects on the thermal stratification and mixing processes were identified. More frequent and prolonged mixing events have shifted reservoir behavior from monomictic to polymictic. The more frequent mixing events promoted warmer bottom layers. Schmidt Stability Index corroborates stratification weakening presenting lower values during drought period.

This study had evidenced that drought period had strong implications on the thermal structure of the reservoir and it may bring to managers, beyond quantitative concerns about water availability, also concerns about water quality and water security. Considering that weather is expected to become more extreme and variable due to climate change, there is an increasing need to consider extreme hydrological and meteorological events when planning and managing water supply systems in order to meet human demands of drinking water quantity and quality. This is a great challenge for less developed countries, such as Brazil, which tend to be disproportionately affected due to higher environmental sensitivity, reduced resources, and lower adaptive capacity.

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CAPÍTULO 5

Effects of a drought period on physico-chemical variables and phytoplankton dynamics in a tropical reservoir

ABSTRACT

Changes in meteorological variables to a drought condition can disrupt hydrodynamic processes, substance fluxes, nutrients concentration and phytoplankton dynamics in lakes and reservoirs. The aim of this study was to identify the effects of a drought period on the physico-chemical variables and on the phytoplankton dynamics in a tropical water supply reservoir. Field data of physico-chemical variables and phytoplankton biomass were compared before and during the drought period. Results show that water level reduction contributed to the degradation of the water quality mainly through increasing turbidity, total solids and nutrients concentration in the water column. A large increase in conductivity was also observed during the drought. Phytoplankton composition was shifted from cyanobacteria to chlorophyceae and the total biomass decreased probably due to more frequent mixing events. This study points out that water managers have to be prepared to deal with drought periods because beyond the quantitative issues, there is also a concern regarding water quality.

Keywords: water supply reservoir; water quality, mixing.

1 INTRODUCTION

Water level of lentic water bodies may naturally vary as a result of seasonal or long-term imbalance between the amounts of water entering and leaving the lake. These natural fluctuations are inherent of lake ecosystems and they are essential for the survival and well-being of many species that have evolved to suit their life cycle to those fluctuations. Furthermore, variation in lake level is important for a range of ecosystem services (Wantzen *et al.*, 2008), such as the foodweb in littoral zones, triggering life-cycles of some species and creating spawning or refuge habitat. However, extreme water level reductions have undesirable effects for ecosystems, anthropic uses and biota, including both top-down and bottom-up effects (Zohary and Ostrovsky, 2011).

Drought events with water level reduction in lakes and reservoirs have been reported in literature: Costa *et al.* (2016) studied the drastic water level reduction of Lake Pocinhos and Lake ESEC (Brazil) during two exceptionally dry years; Fadel *et al.* (2015) verified a water level decrease of more than 15 m in Karaoun reservoir (Lebanon) due to low inflow and

continuous withdrawal; the volume stored in Cruzeta reservoir (Brazil) decreased from 90% to 35% approximately during two years of a severe drought (Medeiros *et al.*, 2015); in Valle de Bravo reservoir (Mexico) water level decreased 10 m during two drier years (Valdespino-Castillo *et al.*, 2014); the lake level of Little Rock Lake (United States) declined 17% over a prolonged drought (Gaeta *et al.*, 2012); a water level reduction was reported in Lake Alexandrina and Lake Albert (Australia) due to several years of drought (Wedderburn *et al.*, 2012); water depth at different points of Lake Balaton (Hungary) decreased by 19.5% to 35% in a extremely dry period (G.Tóth *et al.*, 2011).

Drought events are expected to reduce water level, an important driving factor for lake dynamics, affecting the thermal structure of the water column (Zohary and Ostrovsky, 2011), its physical environment and biota (Han *et al.*, 2016), and may result in a degradation of water quality (Medeiros *et al.*, 2015). The physical and chemical impacts are mainly through mixing enhancement (Valdespino-Castillo *et al.*, 2014), higher nutrient concentrations (Jeppesen *et al.*, 2015) and increased conductivity, pH, turbidity and suspended solids (Brasil *et al.*, 2016; Jeppesen *et al.*, 2015).

Water level reduction affects phytoplankton biomass and species composition through its effect on mixing and oxygen dynamics (Valdespino-Castillo *et al.*, 2014) and enhancing the proliferation and dominance of cyanobacteria (Paerl and Huisman, 2008). A study involving 40 Brazilian ponds estimated that the reduction in water level caused by droughts was responsible for eutrophication processes, leading to a higher biomass and dominance of cyanobacteria (Brasil *et al.*, 2016). The volume reduction (90%) in semi-arid Mediterranean reservoirs led to eutrophication and an increased incidence of potentially toxic cyanobacteria species such as *Microcystis* (Naselli-Flores, 2003). Moreover, drought may increase the water residence time in lakes, and longer water residence time during dry years may increase cyanobacteria biomass (Romo *et al.*, 2013). Bouvy *et al.* (2000) showed that the lack of water renewal linked to a drought event in thirty nine Brazilian reservoirs was the major determinant of both trophic conditions and cyanobacteria dominance.

On the other hand, below a critical threshold of water volume, the water level reduction may also increase sediment resuspension and therefore inorganic turbidity, decreasing light availability in the water column. As a consequence, there is an inhibition of phytoplankton growth, leading to a reduction in phytoplankton biomass (Jeppesen *et al.*, 2015). Medeiros *et*

al. (2015) verified that cyanobacteria biomass declined with ongoing reduction in water volume of Cruzeta reservoir (Brazil).

The effects of drought on chemical and biological dynamics of lakes and reservoirs are diverse and, in some cases, antagonistic. Therefore, for improving the management of these water bodies and guarantee the ecosystem services provided by them, to investigate and to understand the impacts of drought periods on the water quality of lentic water bodies are needed. In comparison to temperate regions, little is known about how water level reduction affects the ecology of reservoirs in the tropics (Brasil *et al.*, 2016). Besides that, the existing literature based on the impacts of water level decreasing in deep lakes is considerably smaller than for shallow lakes (Zohary and Ostrovsky, 2011).

It is important to consider the impacts of water scarcity on phytoplankton dynamics to adopt appropriate planning and mitigation measures for water resource systems in face of such extreme hydrological events. The aim of this study was to analyze the effects of a three exceptionally dry years (2014–2016) in southeastern Brazil on physico-chemical variables and phytoplankton dynamics in a water supply reservoir. Physico-chemical variables and phytoplankton biomass and composition were compared before and during the drought using monitored data.

2 MATERIAL AND METHODS

2.1 Study site

Serra Azul reservoir is located in a metropolitan region, southeastern Brazil (19.97° S, 44.34° W) (Figure 5.1), at elevation 760 m. The reservoir has a surface area of 9.11 km², a volume of 8.16×10^7 m³, with an average depth of 9.0 m and 47.3 m of maximum depth at its full capacity (Fernandes, 2012). Its hydraulic residence time is 375 days. This reservoir was built in 1981 and is exclusively used to supply drinking water to 800,000 inhabitants in Belo Horizonte Metropolitan Region (BHMR).

The Serra Azul reservoir has eight tributaries and one outflow. Its catchment is a Special Protection Area of 262 km² and land use is covered by vegetation (46%), farming (26%), urbanization (10%), exposed soil (7%), bedrock (5%), mining activities (2%) and the reservoir itself (4%) (Matos *et al.*, 2017). Climate in the region is sub-humid tropical (Cwb)

(Köppen, 1948) with a wet season from October to March and a dry season from April to September.

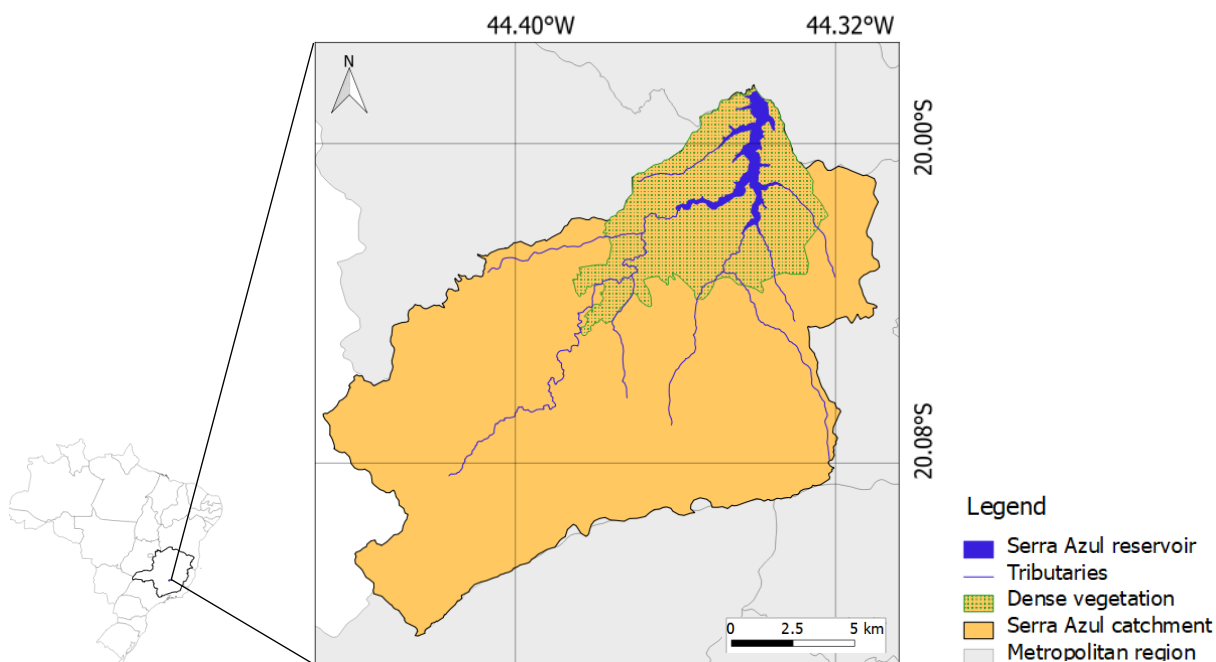


Figure 5.1 – Serra Azul reservoir, its tributaries and catchment.

Since 2013, the region has faced a severe drought period. In almost all the catchments of southeastern Brazil, the lowest rainfall volumes and the lowest streamflow of the 70-year hydrological time-series were recorded (CPRM, 2017). The year 2014 registered the worst drought in this region with an annual rainfall depth return period estimated in 100 years (ANA, 2014). Some of the largest reservoirs in the country were forced to operate the dead storage and there was a deficit in the drinking water supply system. In Serra Azul reservoir, the volume reached 5.2% of its total capacity in November 2014 (Figure 5.2). The water scarcity remained in the year 2017, during which the recorded rainfall was lower than the historical average annual precipitation.

A hydrodynamic modelling (Chapter 4) indicated that the drought period affected hydrodynamics of Serra Azul reservoir through: thermal structure had shift from a monomictic behavior to a polymictic one; mixing events were more frequent; the count of mixed days almost doubled; and water temperature at bottom layers (at 20 m depth) increased 2.1 °C, on average.

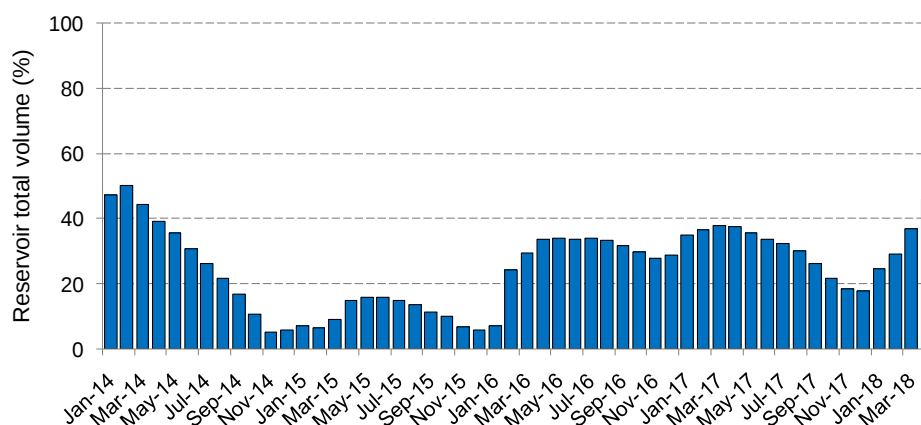


Figure 5.2 – Serra Azul volume from 2014 to 2018 in percentage of its total storage capacity ($8.16 \times 10^7 \text{ m}^3$) (COPASA, 2018).

2.2 Field data analysis

Data from one in-lake monitoring station were consulted in the present study. This station is located 200 m from the water withdrawal structure (19.97° S , 44.34° W) and is operated by the Minas Gerais Water Utility (COPASA) with sampling procedures every two months since 1981. Biomass of each phytoplankton class at Secchi depth and nine physicochemical variables measured at 0.5, 5.0 and 15.0 m depth in water column were evaluated: Secchi depth, turbidity, total solids concentration (TS), total phosphorus concentration (TP), ammonia concentration (NH_3), nitrate concentration (NO_3), dissolved oxygen concentration (DO), pH and conductivity. The water temperature was not included in this study because it is deeply studied in Chapter 4 using other approaches and other tools.

A comparative analysis was carried out between two periods through boxplots: from 2009 to 2013, period under usual meteorological and hydrological conditions; and from 2014 to 2016, the drought period. In order to examine any potential difference before and during the drought period, a non-parametric Mann–Whitney test with XLSTAT software version 2018.1 was performed.

3 RESULTS AND DISCUSSION

3.1 Physico-chemical variables

Environmental differences were observed between the period under usual meteorological and hydrological conditions (2009–2013) and the drought period (2014–2016). Man-Whitney U-

test revealed that Secchi depth during the drought period was significantly lower than before this period (Figure 5.3a) as a consequence of higher turbidity (Figure 5.3b) and higher total solids concentration (Figure 5.3c). Due to the reservoir shallower depth, sediment resuspension caused by wind and more frequent mixing events likely induced higher suspended solids concentration (and consequently higher turbidity) in the water column.

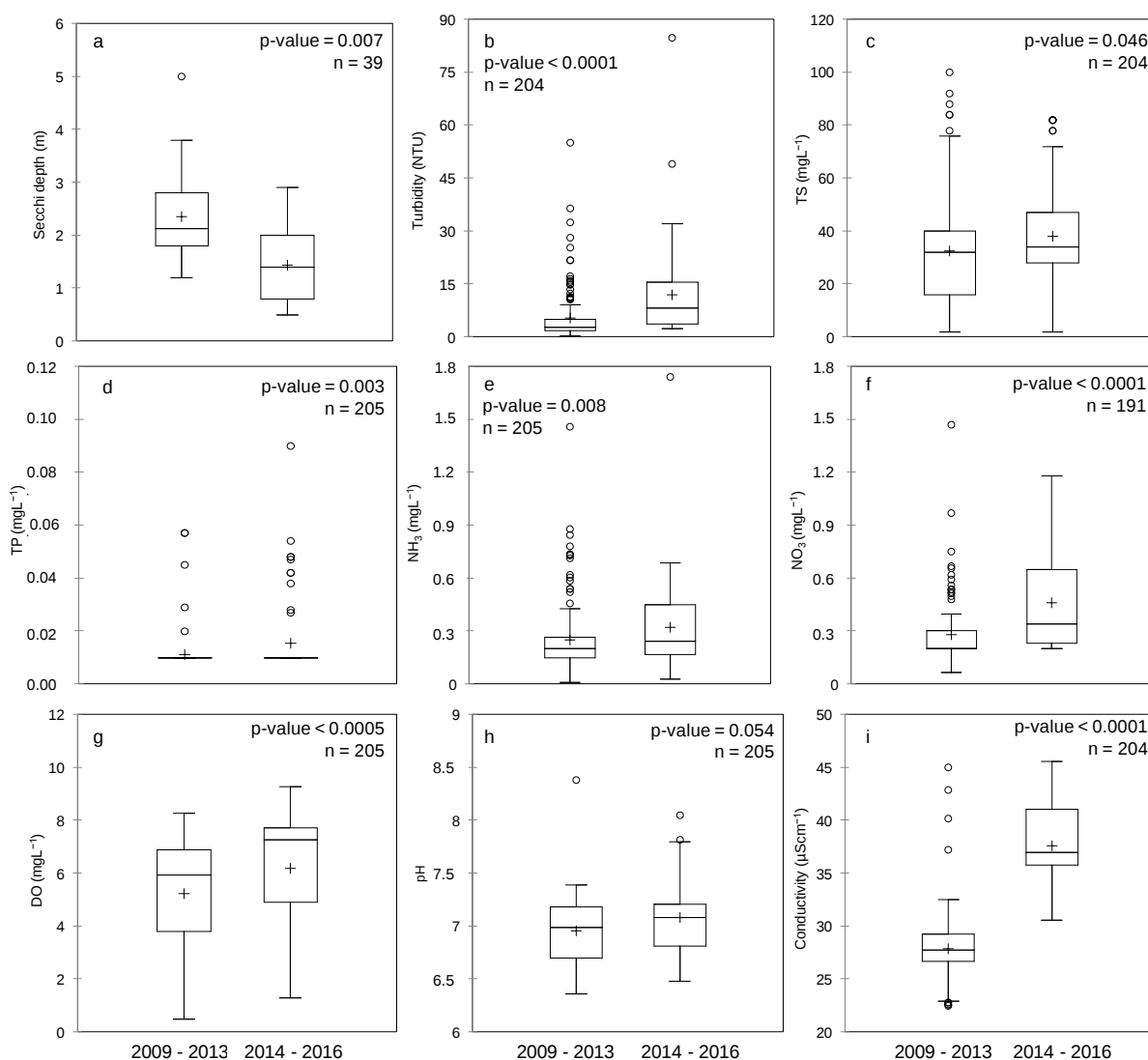


Figure 5.3 – Boxplots of monitored physico-chemical variables of Serra Azul reservoir for the period from 2009 to 2013 and the period from 2014 to 2016 at different depths: a) Secchi depth; b) turbidity; c) total solids concentration; d) total phosphorus concentration; e) ammonia concentration; f) nitrate concentration; g) dissolved oxygen concentration; h) pH and i) conductivity. The horizontal line inside the boxplot indicates the median, the cross marker indicates the mean, the boundaries of the box plot indicate the 25th and 75th percentiles and the circle indicates the outliers. Whiskers indicate the maximum and minimum value. Source of data: COPASA, 2017.

As a consequence of the water level reduction, TP, NH₃ and NO₃ concentrations were higher during the drought period (Figure 5.3d, Figure 5.3e, Figure 5.3f, respectively). Despite lower

inflow volume, nutrients higher concentrations in the reservoir are potentially related to internal processes such as decomposition and internal loading. Özen *et al.* (2010) also found higher nutrient concentrations during dry years in two Mediterranean lakes.

Dissolved oxygen concentration increased in the drought period (Figure 5.3g). It is possibly due to the more frequent turbulent mixing events promoting reaeration and vertically transferring DO from surface to deep waters. There was no difference, however, in pH (Figure 5.3h) between the periods. Significantly higher values of electrical conductivity occurred in the drought period (Figure 5.3i). It was also observed by Brasil *et al.* (2016) in Brazilian reservoirs under dry conditions. Similarly, in Lake Doirani (Greece), electrical conductivity increased from 0.49 to 1.24 mScm⁻¹ during a drought and showed a significant correlation with the water-level reduction (Jeppesen *et al.*, 2015).

3.2 Phytoplankton dynamics

Four phytoplankton classes were identified during the studied period: cyanobacteria, chlorophyceae, chrysophyceae and pyrrophyceae (Figure 5.4). From 2009 to 2014, cyanobacteria was dominant likely due to its higher maximum growth rate and to its several types of adaptive strategies which allow it to dominate over other phytoplankton groups. From 2014 to 2016, cyanobacteria biomass decreased and its relative contribution to total phytoplankton biomass reduced from 90% to 18%. The lower water level and more frequent mixing events induced sediment resuspension, which may have negatively affected cyanobacteria through increased turbidity (total solids). In 2016, the phytoplankton dominance shifted from cyanobacteria to chlorophyceae, which reached 79% of the total composition.

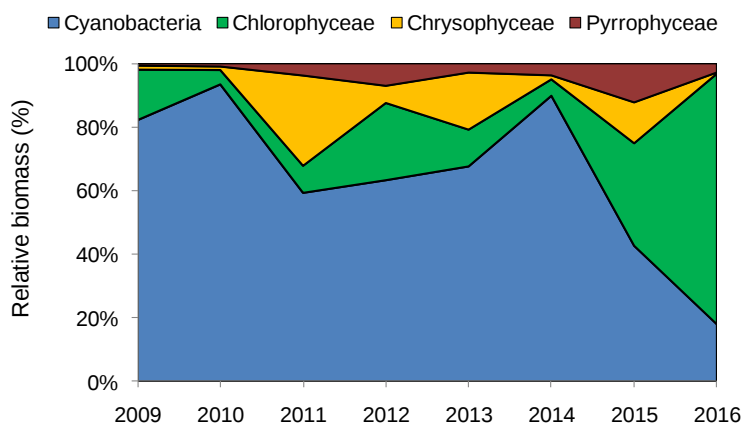


Figure 5.4 – Relative biomass of each phytoplankton class. Source of data: COPASA, 2017.

From 2009 to 2016, chlorophyceae contributed with the highest number of species (50) followed by pyrrophyceae (31), chrysophyceae (24) and cyanobacteria (23). Chlorophyceae dominant species during the drought period were *Coelastrum spp.*, *Chlamydomonas sp.* and *Botryococcus sp.* It suggests that the high nutrient concentrations associated with turbidity and mixing intensification could have created favorable environmental conditions for the chlorophyceae.

Monitored data evidences the reduction on phytoplankton biomass during the drought period. Median biomass decreased from 0.14 $\mu\text{g L}^{-1}$ to 0.06 $\mu\text{g L}^{-1}$ for cyanobacteria (Figure 5.5a), from 0.007 $\mu\text{g L}^{-1}$ to 0.002 $\mu\text{g L}^{-1}$ for chlorophyceae (Figure 5.5b), from 0.006 $\mu\text{g L}^{-1}$ to 0.0009 $\mu\text{g L}^{-1}$ for chrysophyceae (Figure 5.5c) and from 0.005 $\mu\text{g L}^{-1}$ to 0.001 $\mu\text{g L}^{-1}$ for pyrrophyceae (Figure 5.5d).

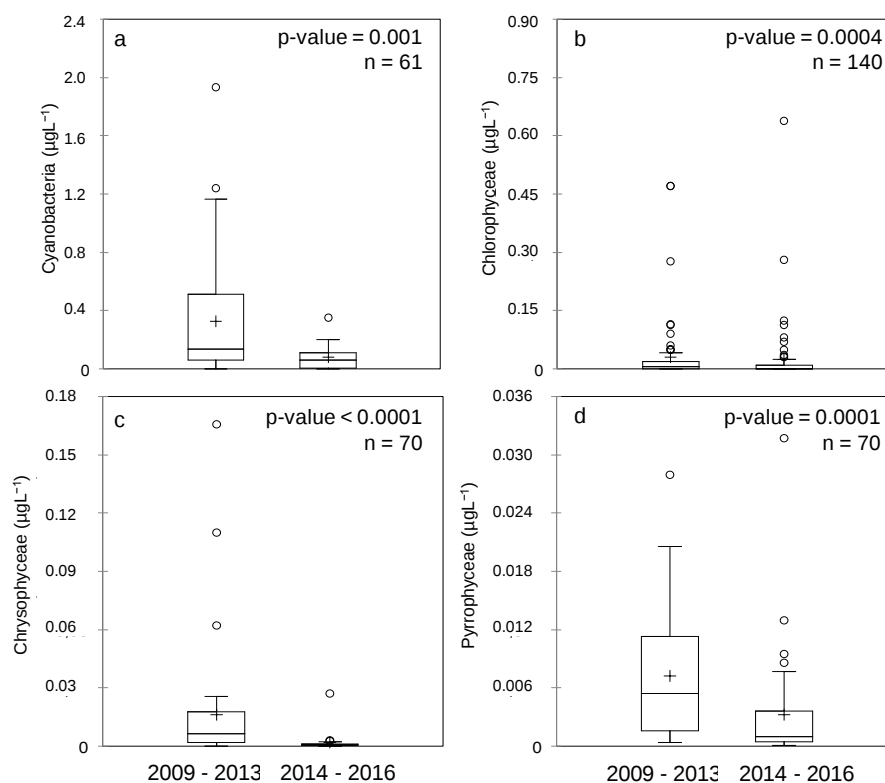


Figure 5.5 – Boxplots of monitored phytoplankton classes of Serra Azul reservoir from 2009 to 2013 and from 2014 to 2016: a) cyanobacteria, b) chlorophyceae, c) chrysophyceae and d) pyrrophyceae. The horizontal line inside the box plot indicates the median, and the boundaries of the box plot indicate the 25th and 75th percentiles. Whiskers indicate the maximum and minimum value. Source of data: COPASA, 2017.

Although some studies reported higher phytoplankton biomass during dry periods (Bouvy et al., 2000; Elliott, 2012; Brasil et al., 2016), antagonistic phytoplankton dynamic may occur during an extreme drought period. During two exceptionally dry years in the Brazilian semi-

arid region, water level reduction of two reservoirs led to phytoplankton biomass increase and cyanobacteria dominance and phytoplankton biomass decrease in the shallower lake (Costa *et al.*, 2016). In Cruzeta reservoir (Brazil), the lowest phytoplankton biomass occurred in the period of severe drought (Medeiros *et al.*, 2015). In Serra Azul reservoir, turbidity increased in the drought period, limiting light intensity in the water column (lower Secchi depth) which seemed to be a decisive factor in reducing phytoplankton biomass. The results suggest that drought have impacted chemical, physical and biological characteristics the reservoir, affecting phytoplankton composition and biomass.

4 CONCLUSIONS

This study shows the effects of the drought period on the physico-chemical variables and phytoplankton dynamics of a tropical reservoir. As an overall, the drought promoted a degradation of water quality in terms of physico-chemical variables mostly by reducing the transparency and increasing nutrients concentration, despite lower inflow volume. Turbidity and total solids significantly increased in water column during the drought possibly due to more frequent mixing events and, as a consequence, Secchi depth reduced. Significant higher values of dissolved oxygen concentration and conductivity were observed during drought period.

Low water levels and hydrodynamic alterations caused by the drought affected the system also through a significant decrease in the biomass of each phytoplankton class. Besides that, phytoplankton composition was also affected and its dominance shifted from cyanobacteria to chlorophyceae.

The results presented in this study improve the knowledge about phytoplankton dynamics and physico-chemical variables in tropical reservoirs under water scarcity. Since the effects of a drought on phytoplankton dynamics and physico-chemical variables are not entirely intuitive, this study evidences that water managers need to be prepared to face not only quantitative issues during water scarcity periods, but also the degradation of water quality.

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CAPÍTULO 6

Considerações finais

1 CONCLUSÕES

Modelagem hidrodinâmica e ecológica unidimensional foi empregada utilizando-se o GLM-AED para simular a temperatura da água e a dinâmica do fitoplâncton durante o período de 2009 a 2016 em um reservatório destinado ao abastecimento de água. O modelo mostrou-se uma eficiente ferramenta capaz de capturar os processos térmicos e a dinâmica do fitoplâncton na coluna d'água. Devido à abordagem unidimensional, o modelo desconsidera os processos horizontais que podem ser capazes de intervir na dinâmica vertical. A limitação deste estudo se deve, em grande parte, à parametrização do modelo, o que comumente se constitui no principal desafio da modelagem ambiental.

Os resultados evidenciam a influência das condições meteorológicas e hidrológicas sobre os processos internos em corpos d'água lânticos. Os eventos de estratificação térmica e mistura da coluna d'água acompanham a sazonalidade do período chuvoso e de seca. A temperatura da água mostrou maior sensibilidade à vazão afluente ao reservatórios e à ação do vento na camada superficial. O modelo hidrodinâmico, após ter sido calibrado, representou o comportamento monomítico da estrutura térmica do reservatório, típico de reservatórios tropicais profundos. O modelo ecológico representou a comunidade de fitoplâncton em dois grupos: cianobactérias e os demais grupos. A biomassa e composição do fitoplâncton são governadas, em grande parte, pelos eventos térmicos, que estão intimamente relacionados aos padrões meteorológicos e hidrológicos locais.

A modelagem matemática revela que o período de escassez hídrica que atingiu a região sudeste do país entre 2013 e 2016 afetou os processos hidrodinâmicos no reservatório Serra Azul por meio do enfraquecimento da estabilidade térmica e da maior frequência dos eventos de mistura. Em consequência, o reservatório passou a apresentar comportamento polimítico. Ademais, houve um aumento da temperatura da água nas camadas profundas. Dados de campo indicam alterações significativas nas variáveis físico-químicas: redução da transparência da água e aumento da condutividade elétrica, da concentração de nutrientes e da concentração de oxigênio dissolvido. O fitoplâncton foi sensível às mudanças ambientais e a biomassa de cada grupo reduziu significativamente durante o período de escassez hídrica. Ademais, o ambiente deixou de ser dominado por cianobactérias e as clorofíceas tornaram-se dominantes em 2016.

Os resultados desse estudo evidenciam que os efeitos da escassez hídrica sobre reservatórios tropicais envolvem não apenas aspectos quantitativos de disponibilidade hídrica, mas também aspectos qualitativos devido a alterações nos processos térmicos, nas características físico-químicas e na dinâmica ecológica do corpo d'água. Acredita-se que esse estudo pode auxiliar a gestão de recursos hídricos ao elucidar a resposta de reservatórios tropicais à condição de escassez hídrica.

2 RECOMENDAÇÕES PARA ESTUDOS FUTUROS

Visando ao aprimoramento da modelagem hidrodinâmica e ecológica do reservatório Serra Azul, seria interessante a realização, em estudos futuros, de análise de incerteza para conferir maior grau de confiabilidade aos resultados obtidos. Permanecem ainda questões concernentes ao comportamento do fitoplâncton, sobretudo a alteração do grupo dominante ocorrida durante o período de escassez hídrica. Avanços no entendimento dos gatilhos que conduziram a esta alteração são necessários no intuito de aprofundar a compreensão dos impactos de eventos extremos sobre a comunidade fitoplanctônica.

Salienta-se que este estudo estava inserido em um projeto de pesquisa de abrangência maior (Projeto MoMa-SE), que conta com a modelagem hidrológica da bacia hidrográfica do reservatório Serra Azul. Pretende-se, em estudos futuros, integrar os modelos hidrológico e hidrodinâmico-ecológico buscando-se superar a falta de monitoramento do regime hidrológico nos cursos d'água afluentes ao reservatório.

Sugere-se o estabelecimento de campanhas periódicas de monitoramento, incluindo-se variáveis estratégicas para a modelagem dos processos térmicos e da dinâmica do fitoplâncton. Ademais das variáveis usualmente monitoradas, devem ser incluídos o monitoramento *in loco* da velocidade do vento devido ao elevado coeficiente de sensibilidade do modelo hidrodinâmico, maior frequência da amostragem da profundidade de Secchi, do perfil vertical da temperatura da água e da biomassa do fitoplâncton. A modelagem dos demais mananciais de abastecimento de água da RMBH é encorajada, pois permitirá um prognóstico da condição atual dos reservatórios na região. Para tanto, ressalta-se a necessidade da incorporação do monitoramento hidrológico nas bacias hidrográficas de cada manancial.

Apêndices

APÊNDICE A

Registro fotográfico das visitas técnicas ao reservatório Serra Azul e sua bacia hidrográfica.



Figura 1A.1 – Corpo d'água, torre de tomada d'água, barragem, vertedouro de soleira livre e rebaixamento do nível d'água no reservatório Serra Azul.



Figura 1A.2 – Tributários do reservatório Serra Azul: (a) Curralinho; (b) Estiva; (c) Diogo; (d) Ribeirão Serra Azul; (e) Jacu; e (f) Pedreira.



Figura 1A.3 – Aproveitamento agrícola na bacia hidrográfica do reservatório Serra Azul.

APPENDIX B

R script for running and analyzing GLM.

```
# Loading packages
library(GLMr)
library(glmtools)
library(rLakeAnalyzer)

# Defining work folder and files
work_folder <- "." # Define work folder
nc_file <- file.path(work_folder, 'output.nc') # Indicate model output file
field_file <- file.path(work_folder, 'field_data.tsv') # Indicate field data file

# Getting model information
print(work_folder) # Get work folder
dir(work_folder) # Get all files in the work folder
glm_version() # Get the current version of GLM
citation('GLMr') # Get the current citation for GLMr (or glmtools or rLakeAnalyzer)

# Defining variables
num_cells <- 100 # Number of layers for data interpolation
var_name <- "temp" # Variable (water temperature = temp; density = rho; salinity=salt)
metric <- "water.temperature"

# Running simulation
run_glm(work_folder) # Run the model
sim_vars(file = nc_file) # Find simulation variables, their abbreviations and units

# Getting simulation results
water_temp <- get_temp(nc_file, reference = 'surface') # Get water temperature
write.table(water_temp, file="temp.csv") # Save file water_temp.csv in the work folder
temp_wtr <- file.path(work_folder, 'temp.wtr') # Indicate simulated water temperature file
water.temp = load.ts(temp_wtr) # Load simulated water temperature file
```

```

surface_height <- get_surface_height(file = nc_file) # Get surface height
write.table(surface_height, file="surface_height.csv") # Save file surface_height.csv in the
work folder

variable <- get_var(file = nc_file, '___', "surface") # Get values for specific variable
write.table(variable, file="variable.csv") # Save file variable.csv in the work folder


# Model performance

rmse <- compare_to_field(nc_file, field_file, metric = 'water.temperature', as_value =
FALSE) # Compare simulated and observed data
print(paste(rmse,'deg C RMSE')) # Print RMSE


# Calculating lake physical metrics

meta_depth = ts.meta.depths(water.temp, slope = 0.1, na.rm=TRUE) # Get metalimnion depth
thermo_depth = ts.thermo.depth(water.temp, na.rm=TRUE) # Get thermocline depth


# Save file .csv in the work folder

write.table(meta_depth, file="meta_depth.csv")
write.table(thermo_depth, file="thermo_depth.csv")


# Plots

plot_temp(file = nc_file, reference = 'surface') # Plot water temperature profile
plot_var(nc_file, '___') # Plot a specific variable
plot_temp_compare(nc_file, field_file) # Plot matching heatmap for simulated and observed
data
plot_validate_profiles(nc_file, field_file, fig_path = FALSE, method = 'interp') # Plot water
temperature profiles comparing simulated and observed data
plot(meta_depth$datetime, meta_depth$top, type='l', ylab='Metalimnion Depths (m)',
xlab='Date') # Plot the hypolimnion top
lines(meta_depth$datetime, meta_depth$bottom) # Add the epilimnion bottom to the previous
plot
plot(thermo_depth$datetime, thermo_depth$thermo.depth, type='l', ylab='Thermocline Depth
(m)', xlab='Date') # Plot the thermocline depth

```