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Linking ecosystem services and damages from bauxite mining in an LCA context

A case study from Hydro and a movement
towards no net loss of ecosystem services

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Master in Industrial Ecology
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Department of Energy and Process Engineering

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NTNU
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MASTER THESIS

for

Student
Alya Francesca Bolowich

Spring 2016

Linking ecosystem services and damages in an LCA context: the case of aluminium mining

Background and objective

During the Masters thesis, the student will assess the current value of the ecosystem services under current mining operations, as part of a collaboration with Norsk Hydro ASA. Baselines for comparisons of the loss of these services are set after the land acquisition by Hydro from Vale, as well as the natural state. The student will build on research methods gathered in Part I (the summer job) and Part II (the Masters project) to find the best way of evaluating an on-site net zero loss of ecosystem services. Hydro aims to mitigate losses regarding biodiversity and ecosystem services with an ambitious reforesting program. The thesis will therefore also aim to have a comparative overview of how much of the ecosystem services is gained or lost due to Hydro after reforesting. In the evaluation of ecosystem service losses, different stages of Hydro's aluminum life cycle up until primary production shall be considered. The overall aim is to further methods for ecosystem service characterization factors within LCIA. The ultimate goal is to attempt to create a characterization factor for ecosystem service incorporation in LCIA as related to this Hydro case study.

The following tasks are to be considered:

1. Based on the findings of Parts I & II, where is the largest possibility for ecosystem service improvement in the Brazilian value chain (e.g. through reforestation)?
2. How does ecosystem service provisioning change over time? When can Hydro expect to see fully restored ecosystem services? What defines fully restored ecosystem services?

3. What are current practices for addressing ecosystem services in LCIA? What are the knowledge gaps, challenges, and limitations for incorporating ecosystem services in LCIA?
4. How can researchers strengthen current LCIA methodology when addressing ecosystem services?
5. How can a characterization factor to include ecosystem services be created?

-- " --

Within 14 days of receiving the written text on the master thesis, the candidate shall submit a research plan for his project to the department.

When the thesis is evaluated, emphasis is put on processing of the results, and that they are presented in tabular and/or graphic form in a clear manner, and that they are analyzed carefully.

The thesis should be formulated as a research report with summary both in English and Norwegian, conclusion, literature references, table of contents etc. During the preparation of the text, the candidate should make an effort to produce a well-structured and easily readable report. In order to ease the evaluation of the thesis, it is important that the cross-references are correct. In the making of the report, strong emphasis should be placed on both a thorough discussion of the results and an orderly presentation.

The candidate is requested to initiate and keep close contact with his/her academic supervisor(s) throughout the working period. The candidate must follow the rules and regulations of NTNU as well as passive directions given by the Department of Energy and Process Engineering.

Risk assessment of the candidate's work shall be carried out according to the department's procedures. The risk assessment must be documented and included as part of the final report. Events related to the candidate's work adversely affecting the health, safety or security, must be documented and included as part of the final report. If the documentation on risk assessment represents a large number of pages, the full version is to be submitted electronically to the supervisor and an excerpt is included in the report.

Pursuant to "Regulations concerning the supplementary provisions to the technology study program/Master of Science" at NTNU §20, the Department reserves the permission to utilize all the results and data for teaching and research purposes as well as in future publications.

The final report is to be submitted digitally in DAIM. An executive summary of the thesis including title, student's name, supervisor's name, year, department name, and NTNU's logo and name, shall be submitted to the department as a separate pdf file. Based on an agreement with the supervisor, the final report and other material and documents may be given to the supervisor in digital format.

☐ Work to be done in lab (Water power lab, Fluids engineering lab, Thermal engineering lab)
☐ Field work

Department of Energy and Process Engineering, 13. January 2016

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ABSTRACT

This study is part of a yearlong study with Norsk Hydro ASA addressing the impacts on ecosystem services in primary aluminum value chains. Here, the focus is on the impacts of bauxite mining at two locations in northern Brazil: Trombetas and Paragominas. Although increasingly used as an impact assessment method, life cycle assessment (LCA) has yet to incorporate ecosystem services as an area of protection, mainly due to region-specific data requirements and the lack of a cohesive agreement as to how they should be covered in LCA. To solve these problems, I propose a region-specific method to account for the potentially lost fraction of ecosystem services (PLES) at an endpoint level. This study is based on aluminum, although the PLES method is applicable in many different cases. The PLES system relies on a literature review, expert knowledge, and a scoring system corresponding to land cover to evaluate the potential presence of ecosystem services. Because ecosystem services are highly site- and area-dependent, this study addresses discrepancies between modeled land cover and expert knowledge on land cover. I found that using modeled land cover data leads to a 27% increase in the perceived loss of ecosystem services when compared to data based on expert knowledge. Trombetas had a lesser impact on ecosystem services than Paragominas using the PLES methodology. However, the PLES does not account for cultural ecosystem services. This would likely yield higher results on overall ecosystem service impacts in Trombetas, especially since it is located where many Quilombolas are living.

SAMMENDRAG

Denne studien er en del av et ettårig prosjekt med Norsk Hydro ASA for å vurdere påvirkninger på økosystemtjenester fra aluminiumsproduksjon. Fokus har her vært på påvirkninger fra bauxittutvinning ved to verk i det nordlige Brasil: Trombetas og Paragominas. Livsløpsanalyse (LCA) har i økende grad vokst frem som den viktigste metoden for miljøkonsekvensutredninger, men vern av økosystemtjenester har til nå ikke vært inkludert i slike vurderinger. To viktige grunner til dette er for det første store datakrav som følge av meget stor geografisk variasjon i økosystemtjenester, og dessuten en mangel på vitenskapelig konsensus om hvordan slik påvirkning skal analyseres i livsløpsanalyser. I denne rapporten presenterer jeg en region-spesifikk metode for å estimere potensielt tapt andel økosystemtjenester. Denne studien fokuserer på aluminium, men metoden kan også benyttes for andre studier. Metoden bygger på eksisterende forskningslitteratur, ekspertvurderinger, og et poengbasert system for vurdering av arealdekke for å estimere graden av eksisterende økosystemtjenester. Siden økosystemtjenester varierer mye fra område til område, har jeg her analysert avvik mellom modellert arealdekke og ekspertvurderinger, og funnet at bruk av modellert arealdekke førte til en 27% øke i oppfattet tap av økosystemtjenester. Trombetas-anlegget funnet å føre til lavere kvantitativ påvirkning på økosystemtjenester enn Paragominas. Dersom metoden også hadde inkludert kulturelle økosystemtjenester ville trolig resultatene vist noe større påvirkning fra Trombetas-anlegget siden dette ligger hvor mange Quilombolas lever.

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ABBREVIATIONS

ADA – Area directly affected

ADI – Area of direct influence

AoP – Area of Protection

CF — Characterization Factor

EF — Effect Factor

ESR – Ecosystem services review tool

FF — Fate Factor

GLC – Global land cover

LCA – Life cycle assessment

LCI – Life cycle inventory

LCIA – Life cycle impact assessment

MRN – Mineração do Norte

PLES – Potentially lost fraction of ecosystem services

SI – Supporting Information

DISCLAIMER

This report is the third and final part of a Norsk Hydro ASA (Hydro) sponsored study designed to examine and quantify ecosystem services impacted by the production of primary aluminum. Hydro is a multinational aluminum company concerned with their impacts on ecosystem services from the value chain of primary aluminum. In this yearlong project, I have been addressing which ecosystem services were and are present on the mining and processing sites during Hydro's bauxite mining, alumina refining, and primary aluminum production operations. Any information that has already been covered in previous reports and is vital for understanding this report is referenced in the Supporting Information (SI). The Table of Contents from the two prior reports have also been included in the SI. These serve as a brief overview of the topics that have already been addressed, such as the connections between biodiversity and ecosystem services (SI-S2), natural capital and ecosystem services (SI-S2), and the monetization of ecosystem services, to name a few examples.

1 INTRODUCTION

1.1 MOTIVATION

Aluminum comes second to steel as the most used metal in society, and there is an increasing trend in its use in the transportation, electronics, and building sectors, to list a few (Liu and Müller 2012). Aluminum is fabricated from bauxite, an amalgamation of aluminum oxides, water, and natural material found up to 20 meters below the earth's surface (Tan and Khoo 2005; Hydro 2012). Bauxite is primarily found around the equatorial belt—an area known for tropical rainforests—and is harvested via strip mining (Hydro 2013). Addressing how the extraction of bauxite impacts ecosystem services is critical for maintaining healthy rainforest ecosystems and the many benefits humans can enjoy from these rainforests.

Ecosystem services, by definition, fundamentally support human life as they are "[…] *the benefits people obtain from ecosystems*" (MA 2003). Ecosystem services comprise four main categories: provisioning, regulating, cultural, and supporting (MA 2003). Some examples of ecosystem services under these four umbrella categories are found in Figure 1. Definitions and examples of ecosystem services in rainforests have been covered in detail in earlier stages; thus, I will not go into further detail of the benefits of ecosystem services.

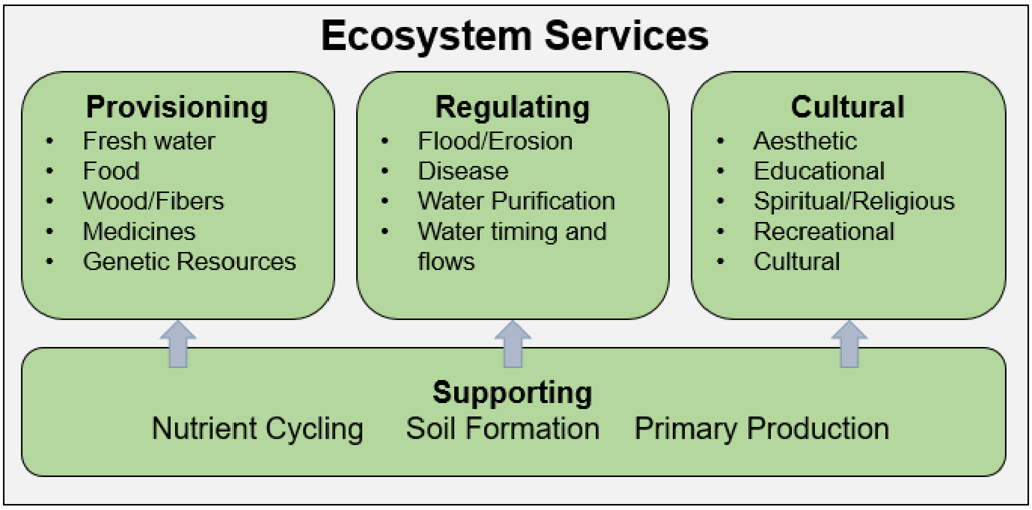

Figure 1. The four main categories of ecosystem services represented here comprise specific, individual ecosystem services. The supporting services serve as a function of others, hence why the supporting services are on the bottom. Interactions between and among ecosystem services are no included in this figure. Figure adapted from the MA (2005).

As of 2010, most published life cycle assessments (LCA) did not detail the mining and processing procedures for many different metals, mostly because of data constraints (Norgate and Haque 2010). Recently, there has been an increase in LCAs of the primary aluminum production chain, mostly related to electricity consumption and greenhouse gas emissions without much emphasis on land use or ecosystem services (Tan and Khoo 2005; Nunez and Jones 2015). Furthermore, ecosystem services are currently not uniquely considered in LCA methodologies (Othoniel et al. 2016). Globally, around 70% of aluminum produced is from primary material, and although this number is slightly decreasing, this highlights its potential impact on places of extraction (IAI 2009b). Here, I aim to fill the methodological gap of incorporating impacts on ecosystem services from aluminum production into LCA.

1.2 GOAL AND SCOPE

The objective of this report is two-fold: (1) to address the feasibility of reaching a no-net loss of ecosystem services scenario for Hydro's bauxite mine in Paragominas, and (2) to summarize and further contribute to the efforts of incorporating ecosystem services into life cycle impact assessment (LCIA). This report highlights the largest possibility for ecosystem service improvement in the Hydro value chain while addressing how ecosystem services change over time and when can Hydro expect to see fully restored ecosystem services. This report also examines how ecosystem services are currently included in LCA, pinpointing the gaps Using the Paragominas and Trombetas mines as a case study, this report answers the questions of what literature exists regarding ecosystem services and LCA, what are the limitations, and how this can be strengthened. This study then goes further to suggest a possible methodology for incorporation of ecosystem services within LCA, including a case study application.

2 BACKGROUND

2.1 ENVIRONMENTAL CONSIDERATIONS FOR PRIMARY ALUMINUM PRODUCTION

To understand impacts on ecosystem services, it is important to know the threat to ecosystem services at each step in the life cycle of aluminum: bauxite mining, alumina refining, and primary aluminum production. The Master's project addressed impacts at the latter two stages, and this study will focus primarily on the mining of bauxite.

Bauxite is the material extracted for aluminum fabrication and is mined via surface (or strip) mining, where the entire surface area gets removed (IAI 2009a). After extraction, bauxite is washed and crushed, removing excess dirt, clay, or other material - called "tailings" - and this excess is put into a tailings pond (Hydro 2012). Environmental concerns related to bauxite mining include displacement and/or removal of biodiversity; disruption of local hydrologic cycle; dust/noise pollution; erosion and run off; and the storage and the containment of tailings (Hydro 2012).

2.2 MINING AND REFINING LOCATIONS IN BRAZIL

Hydro's primary aluminum production value chain extends from Paragominas to Barcarena within the Pará state of northeastern Brazil (Figure 2). Bauxite is mined in Paragominas and then transferred northwest to Alunorte via a 244-kilometer long pipeline (Hydro 2015a). Impacts from this pipeline are not included in this study. A part of that alumina is sent across the road, to the Albras plant, for primary aluminum production. Alumina from Alunorte is also sent to Sunndalsøra, Norway for primary production. Impacts on ecosystem services at Alunorte, Albras, and Sunndalsøra have been addressed in the earlier phases of this project and will not be compared here.

For the purposes of comparison, I will be evaluating the impacts from operations in Paragominas to bauxite mining operations in Trombetas, operated by Mineração do Norte (MRN), another aluminum company. Hydro holds a 5% share in Trombetas and receives 40% of the bauxite extracted (Personal communication Bernt Malme 2016). Trombetas is located

900 kilometers east of Alunorte (Boulangé 2013)—deeper into the Amazon rainforest than Paragominas and just north of the Amazon River (Figure 2). Bauxite from Trombetas is transported via cargo ship down the Amazon River to Alunorte for alumina refining. I do not address aquatic impacts in this assessment nor the transport of bauxite from Trombetas to Alunorte.

Figure 2. Location of mines and refinery in Brazil. The mines are in orange (labeled T and P for Trombetas and Paragominas, respectively) and the alumina refinery, Alunorte (A), is in blue. The scale of the Brazil map is 1:40,000,000 (ESRI) and the zoomed map is 1:10,000,000 (ESRI). The maps were provided by ArcGIS®, the intellectual property of ESRI® (ESRI 2014).

2.3 RECAPITULATION OF IMPACTED AREAS

Before proceeding further in this report, it is important to know the coverage of information from the previous parts of this project in determining impacts on ecosystem services. Part I resulted in findings about the impact on the provisioning of freshwater at Paragominas and the impact on a nearby, protected area for local, indigenous people. More on freshwater provisioning is explained in Section 4.1 and in the SI-S6. During Part II, I developed a method for evaluating the potential loss of ecosystem services. This method measures terrestrial impacts based on mining activities within the respective geographic boundaries.

This methodology to finding potentially lost fraction of ecosystem services is explained further in Section 3.2. Parts I and II focused only on Hydro's value chain; Trombetas is introduced in this report for a comparison of mining impacts.

Both Paragominas and Trombetas each have two different geographical boundaries: an area directly affected (ADA) and an area of direct influence (ADI). The ADA corresponds to the areas that are to be occupied by and restricted to the area used for mining (Barbosa 2015). In this case, the ADA relates to areas owned by Hydro (for Paragominas) (Figure 3). The ADI is the geographical area that could be directly influenced by positive or negative significant impacts from the mining (Barbosa 2015). For Paragominas, the ADA and ADI data were provided.

The Trombetas ADA and ADI were created for this study based on the definitions provided by Barbosa (2015), as no ADI or ADA information was provided. The Trombetas ADA and ADI hold roughly the same ratios as the Paragominas ADA and ADI to the physically mined area.

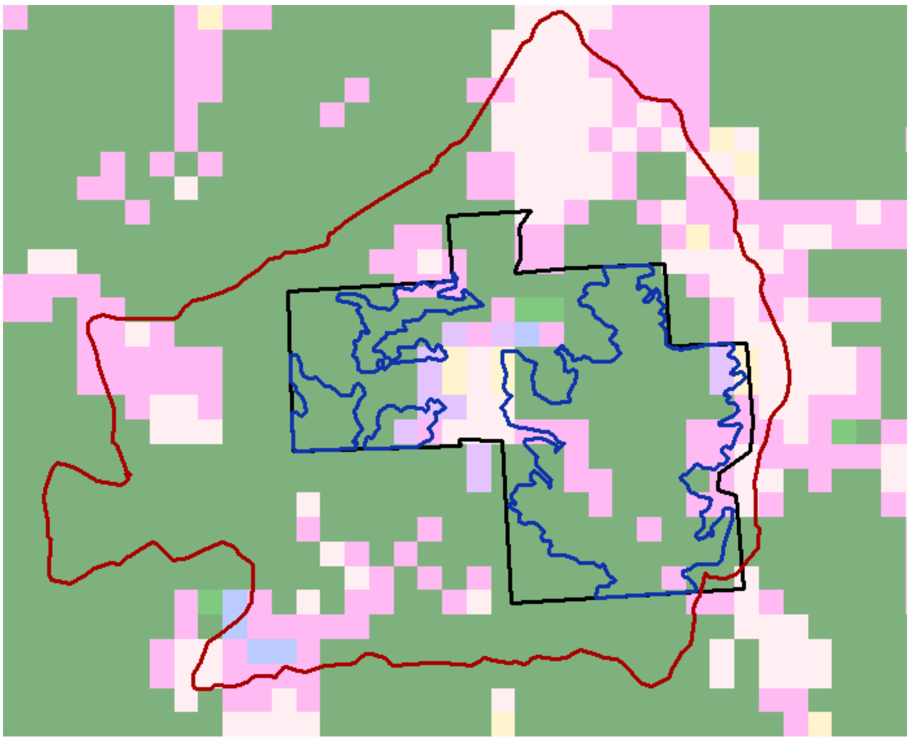

Figure 3. Paragominas ADA. The red boundary is the Hydro ADI, the black is the ADA, and the blue is the boundary for what is projected to be mined in the ADA by 2030. The background is the Global land cover 2000 (GLC) data from Bartholomé and Belward (2005). Green pixels represent forested areas and the pinks and yellow human modified areas. Scale: 1:200 000.

Trombetas has two key differences from Paragominas: (1) it is located in a national forest (Figure 4) and (2) mining is expected to occur on lands claimed by Quilombolas—ethnic minority groups of people who descend from Africa and hold the same rights to land as indigenous peoples (Adams et al. 2013; Steward 2013). The Saracá-Taquera National Forest (*Floresta Nacional Saracá-Taquera*) was established in 1989 and has an area of 429,600 hectares (ha) (4296 km^2) (MRN 2012b). Parrotta and Knowles (2001) report that the natural forests in Trombetas have remained largely undisturbed from harmful human activity for the past 200–300 years. The national forest was established the same time the mining commenced (Hydro 2016b).

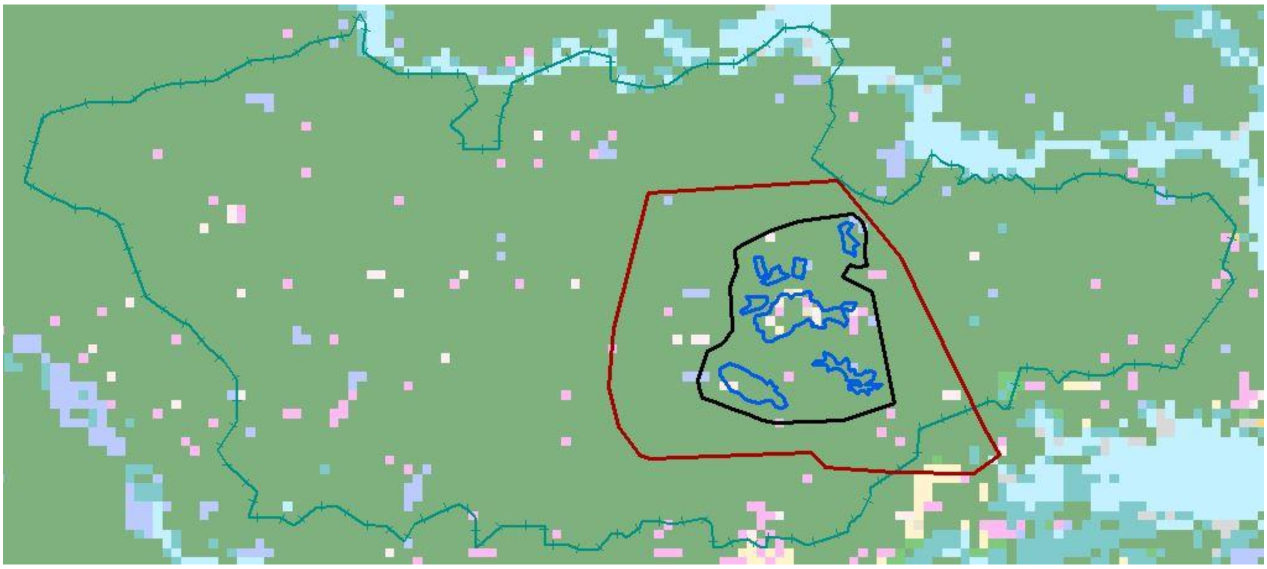

Figure 4. Trombetas within the Sacará-Taquera National Forest The national forest boundary is represented by the tick-marked blue line. The ADA, ADI, and mined areas are marked in their red, black, and blue, respectively. Background data is provided by Bartholomé and Belward (2005) and the national forest data from Oliveira et al. (2009) and cross-checked with Comissão Pró-Índio de São Paulo (2013a). Scale: 1:500,000.

Upon a brief comparison between the areas occupied by the Paragominas and Trombetas mines, the land cover in Trombetas comprises more rainforest (green pixels) than Paragominas, which has more pink and yellow pixels denoting, in this case, human modified areas. It is important to mention that the rainforested areas in Paragominas are secondary forests (Personal communication Bernt Malme 2016) and the implications of this on ecosystem services are discussed in Section 3.2.1.

2.4 REHABILITATION AND ECOSYSTEM SERVICES

Hydro aims to have a 1:1 ratio of rehabilitated land to open mining areas by 2017 with a rolling, two-year gap from clearing and mining to replanting (Persson Hager 2014; Hydro 2015b). Rehabilitation of the forested areas means that Hydro aims to restore the forest to a state that can deliver ecosystem services (FAO 2016). Forest restoration is to get the site to its potential natural vegetation (FAO 2016), which Hydro would ultimately like to achieve.

Hydro has experimented with three different methods to find the best rehabilitation result: natural regeneration, nucleation, and traditional planting, (Hydro 2015b, 2015a). Natural regeneration allows the seeds in the soil regrow on their own without human assistance (Hydro 2015b). Nucleation is a technique that uses piles of organic matter to increase biodiversity habitats for insects, small mammals, plants, and fungi (Persson Hager 2014). The traditional planting method used by Hydro replants in gridded rows 3 m by 3 m apart, using 50-70 different species (Hydro 2015b). Deciding which rehabilitation technique has yet to be decided at Paragominas (Personal communication Bernt Malme 2016).

Time lags are conceptually important to rehabilitation because of their influence on how and when the selected areas will reprovide the lost ecosystem services. In contrast to the natural ecosystem, the rehabilitated ecosystem is likely to have a different species composition, but may have a comparable species richness (Marin-Spiotta et al. 2007). Because of this, the rehabilitated areas in Paragominas and Trombetas may never provide the exact quantity or quality of ecosystem services as the original, primary forest (Chazdon 2013). How ecosystem services change and develop over time is addressed in Section 4.5.3.

2.5 LIFE CYCLE ASSESSMENT

Life cycle assessment (LCA) is a way of determining the environmental damage caused by a product during its lifetime—from resource extraction to recycling and/or disposal (ISO 2006). LCAs identify environmental impacts, both direct and indirect, from a product's value chain (ISO 2006). An LCA comprises four objectives: (1) goal and scope (2) life cycle inventory (LCI) (3) impact assessment (LCIA) and (4) interpretation (Figure 5) (ISO 2006). The goal and scope set the system boundary and the functional unit on which the LCA is based allowing for the comparison of different goods and services (Rebitzer et al. 2004). The LCI accounts for all data requirements and the respective emissions or stressors needed to fulfill the functional unit of the study (ISO 2006; Hellweg and Milà i Canals 2014). LCIA characterizes the LCI data through impact indicators to address the environmental implications from the use of the LCI components (Rebitzer et al. 2004; ISO 2006). LCA and LCIA interpretation should be done at every step to show, for example, differences in materials used in the LCI or the changes among impact categories between products (Rebitzer et al. 2004).

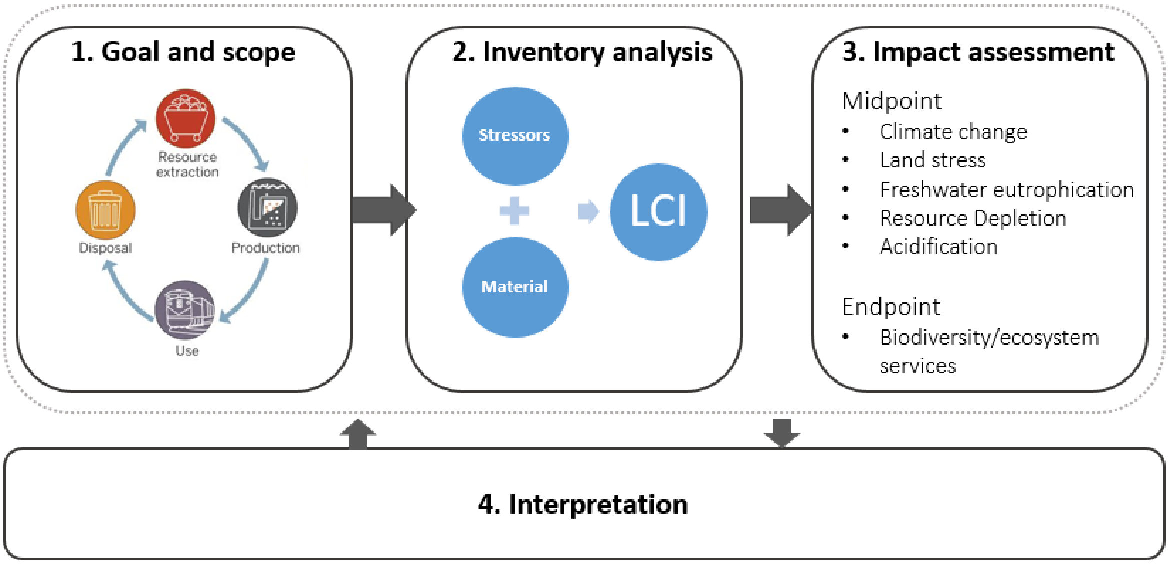

Figure 5. The steps of an LCA include (1) goal and scope, (2) inventory analysis (LCI), (3) impact assessment (LCIA), and (4) interpretation along each step (ISO 2006; Hellweg and Milà i Canals 2014) Figure adapted from Hellweg and Milà i Canals (2014).

Characterization factors (CF) are ways of expressing environmental impacts from emissions or human activities within specific impact categories, such as climate change potential or terrestrial acidification, for example (Bare 2000; Brentrup et al. 2004; Goedkoop et al. 2009a; Boulay et al. 2011). A simple example is characterizing methane (CH_4) emissions into kilograms of CO_2 equivalents, in order to express impacts on climate change from greenhouse gas emissions, at the midpoint level (Goedkoop et al. 2009a).

Impacts can be calculated either at the midpoint or endpoint level, depending on which stage of the impact chain one is wanting to observe (Bare 2000; Brilhuis-Meijer 2014; Raugei et al. 2014). The midpoint level indicates probable stress of an emission on a certain impact category, and the endpoint level indicates the projected damage (Brilhuis-Meijer 2014; Raugei et al. 2014). Endpoint indicators aggregate impacts into three different areas of protection (AoP)—ecosystem quality, human health, and resources—quantitatively suggesting how severe an impact will affect one of the AoPs (Goedkoop et al. 2009a). Reaching these mid- and endpoint levels is done through a series of characterization factors (CFs) (Bare 2000).

CFs identify the cause-effect pathway within a certain impact category through fate, effect, and exposure models (Bare 2000; Boulay et al. 2011). Fate factors (FF), as explained by Huijbregts (2000), tell practitioners into which environmental medium, or compartment, an emission will travel after release (e.g. fertilizer can be released into the air and can travel to water either as spray drift or runoff). The effect factor (EF) indicates the effect that a certain emission can have on the environment or humans (e.g. additional algae growth per kg of fertilizer released) (Huijbregts 2000). Endpoint level CFs connect to the three aforementioned AoPs via endpoint indicators (Goedkoop et al. 2009a) and express "[…] *the relative importance of an emission* […]" (Bare 2000). This can be done by incorporating the damage (via a damage factor) within the EF to show the extent of damage to the ecosystem quality or human health (Pfister et al. 2009). Exposure factors are also components of a CF (Boulay et al. 2011), but in this report, only the FF and EF are most relevant. The framework for CFs is always used this way in LCA In simple terms, the CF can be read as:

$$\mathrm{CF} = \mathrm{FF} \times \mathrm{EF}$$

The environmental impact of the functional unit can then be seen by multiplying the LCI with the CF.

2.6 CURRENT RESEARCH REGARDING ECOSYSTEM SERVICES IN LCA

One challenge method developers face when including ecosystem services into LCIA is the multi-functionality and interconnected nature of ecosystem services (MA 2003; UNEP 2009). Ecosystem services can overlap, underpin, complement, and support one another in non-linear relationships (Carpenter et al. 2009; Koch et al. 2009; Geyer et al. 2010; Mace et al.

2012). Additionally, ecosystem services can lead both directly and indirectly to societal benefits (Zhang et al. 2010b; Othoniel et al. 2016), thus, using a hierarchy of metrics (like in LCA) to match impacts on ecosystem services may not accurately capture all of the occurring impacts (Zhang et al. 2010b; Othoniel et al. 2016). Because ecosystem services are so complex, they can be evaluated in different ways with different data units (Zhang et al. 2010b).

Othoniel et al. (2016) mention three methods for evaluating ecosystem services: (1) proxy-based (such as land use or soil quality), (2) process-modeling based, and (3) primary data-based (such as collecting field samples). Each method of assessment has different strengths and weaknesses. Proxy-based methods are often used, but these can yield LCA results in units which don't necessarily capture the precise service generated from the ecosystem; this is due to the challenge of addressing all mechanisms that create an ecosystem service (Othoniel et al. 2016). Proxy-based methods have lower data requirements, lower level of detail, but a higher level of uncertainty when compared to the other two methods, thus limiting their abilities to weed out interconnections and complex cause-effect chains (Othoniel et al. 2016). Contrary to proxy-based methods, process modeling-based methods have higher data requirements and levels of detail, but may have a lower uncertainty than proxy-based methods (Othoniel et al. 2016). Primary data methods would yield the lowest uncertainty, however they would require the highest data requirements and level of detail; as such, they have not yet been used for development in LCA (Othoniel et al. 2016).

Several studies have attempted to provide characterization factors for anthropogenic impacts on ecosystem services within LCA (Table 1). As noted in their review of ecosystem service inclusion in LCA, Othoniel et al. (2016) mention that addressing the physical flows of ecosystem services does not necessarily translate to the tangible benefits provided by the ecosystem to humans. Cao et al. (2015) did generate a characterization factor (CF) that contributes to tangible human benefits using the physical CFs developed by Müller-Wenk and Brandão (2010), Brandão and Milà i Canals (2012), and Saad et al. (2013) through monetary valuation. Arguments for/against the monetization of ecosystem services do exist (Gómez-

Baggethun et al. 2010), but are not considered in this case study. Part I of this project addressed the complexities and pros and cons of monetization and will not be discussed here.

Table 1. A brief overview of incorporated ecosystem services in LCA is listed in this table, although this is not an extensive list of all ecosystem services listed by either the Millennium Ecosystem Assessment (MA) or the Common International Classification of Ecosystem Services (CICES). Othoniel et al. (2016) developed a table per the CICES structure with corresponding LCA inclusion.

Author	Ecosystem service	Characterization level	CF Unit
Brandão and Milà i Canals (2012)	biotic production potential	midpoint	(kg C·yr)/m^2
Saad et al. (2013)	erosion control	midpoint Ɨ	centimoles of cation/kg soil
	freshwater regulation	midpoint Ɨ	ton/(ha·yr)
	water purification	midpoint Ɨ	mm/yr (for groundwater recharge)
Arbault et al. (2014)	gas regulation	midpoint	kg bioavailable C/kg gross primary production
	climate regulation	midpoint	°C/Gt biomass
	disturbance regulation (erosion and water reg.)	midpoint	Gt biomass
	soil formation	midpoint	Gt dead organic matter
	nutrient cycling	midpoint	Gt nutrients in organic matter
	waste treatment	midpoint	Gt (capacity for waste assimilation)
	recreation	midpoint	Gt biomass/ social capital index
Cao et al. (2015)	social cost of compensation and/or adaptation to ecosystem services loss from land use	endpoint	$/(ha·yr)

Ɨ denotes assumed midpoint

In their study aiming to create an indictor to attribute ecosystem service impacts from land use, Cao et al. (2015) use monetary valuation of ecosystem services and soil functions as their basis to express these impacts at the end-point level. Cao et al. (2015) translate biophysical flows into ecosystem services (climate regulation, biotic production, groundwater recharge, erosion control, and water purification), which are then converted and measured in monetary units for the further development of an AoP for ecosystem services. In doing so, Cao et al. (2015) integrate societal impacts on ecosystem services within an LCIA framework (Othoniel et al. 2016). This is represented in Equation 2, which shows an economic impact value for damages on ecosystem services (Cao et al. 2015).

$$I_i = A \times t_{occ} \times CF'_i \tag{2}$$

Where:

I = impact on service *i*

A = land area occupied (ha)

t_{occ} = length of occupation (yr)

and

$$CF'_i \left[\frac{\$}{(ha \cdot yr)}\right] = ECF(CFi) \times XF_i \times AC \tag{3}$$

Where:

CF'_i = damage score for damages on ecosystem services

ECF_i = economic conversion factor

XF = exposure factor

AC = adaptation capacity

The ECF_i accounts for the social cost for the loss of an ecosystem function (what underlies the ecosystem service) (*i*); The XF_i is the "[…] *fraction of the ecosystem function used as an ES by society*", and the AC is the economic cost for a society to compensate for the loss of an ecosystem service (Cao et al. 2015). The AC is built upon research from Boulay et al. (2011) and data from the World Bank (Cao et al. 2015).

Zhang et al. (2010a) also develop a method for quantifying impacts on ecosystem services in life cycle assessment (LCA) through monetary valuation. They use environmentally extended input-output analysis coupled with LCA (known as hybrid LCA (Treloar et al. 2000)) to do so (Zhang et al. 2010a; Othoniel et al. 2016). This is more an extension of LCA as it uses different and additional information and inputs not typically required for traditional LCA (Othoniel et al. 2016). The hybrid input-output analysis approach, which compares country-wide changes through economic supply-use tables (Suh 2004), as is not a relevant approach to this case study and not considered here.

2.7 RESEARCH GAPS REGARDING ECOSYSTEM SERVICES IN LCA

Although Cao et al. (2015) developed a successful characterization factor using country-wide economic data, the persons affected by the impacts on ecosystem services are not spatially defined (Cao et al. 2015; Othoniel et al. 2016). Spatial specificity is another challenge when incorporating ecosystem services in LCIA, and regionalization must be considered (Othoniel et al. 2016). Regionalization is a multi-faceted subject accommodating the differences in energy and material flows through different spatial scales (Mutel et al. 2012). Spatial scales extend from generic (global), to site-dependent (country, ecoregion, watershed, etc.), to site-specific (localized) (Hauschild 2006). Ecosystem services vary depending upon the region, the land use, the land cover, and neighboring land use/cover among other criteria (Othoniel et al. 2016). For example, the ecosystem service "pollination" may be present in Oslo, Norway, but may have a stronger presence in warmer climates, such as Seville, Spain, and regionalization would better account for the different strengths of this ecosystem service.

Even with country-specific data, some countries, such as the United States, China, and Brazil, have dramatic differences in their ecological structure (Hellweg and Milà i Canals 2014), so one, country-wide mid- or endpoint score will not necessarily be the most accurate in depicting impacts on ecosystem services. In this study, land cover is used as a proxy-method for evaluating ecosystem services, and according to ReCiPe 2008 methodology (Goedkoop et al. 2009a), the connection between the midpoint and the endpoint is highly region specific. Using a CF for the entire country of Brazil may yield less accurate LCA results than using an ecoregion CF for the specific area under examination. This is because Brazil is a large country with several different land covers each with a different strength of ecosystem service provisioning.

Biodiversity, natural resources, weather patterns, seasons, and edge effects can all play different spatial and temporal roles within an ecosystem, thus adding to the difficulty of accurately incorporating ecosystem services in LCA (Othoniel et al. 2016). The connection between biodiversity and ecosystem services, as well as natural resources and ecosystem services, was addressed in the earlier stages of this project and can be seen in SI-S2. Othoniel et al. (2016) argue that the spatial and temporal variations require a certain level of detail to remain accurate, but must also contain a certain breadth to be applicable within an LCA framework. This spatialization is important for two reasons. First, land going from forest or to forest from another state will cancel out each land covers' impacts if the differences in land cover are not already accounted for (Othoniel et al. 2016). Second, looking at location alone, such as in the same ecoregion, may not differentiate between different land cover types, so converting agricultural land to pasture, or vice versa, will not reflect the same changes in ecosystem services when evaluated at an ecoregion scale (Othoniel et al. 2016). Addressing these ecosystem specific challenges from an LCA context that can be applied in a wide variety of studies and ecosystems is a large obstacle for method developers.

When evaluating ecosystem services from an LCA perspective, land use, land cover, and soil quality are usually used as proxies for ecosystem service quality (Othoniel et al. 2016). Impacts on ecosystem services largely stem from land cover changes and land use (Cao et al. 2015). Simply put, all ecosystem services need land in order to function, and the viability of all land depends on the soil quality (Cao et al. 2015). One crucial aspect to evaluating ecosystem services in LCA is that the typical "cause-effect" chain desired for LCA is very simple when compared to the complexities of ecosystem services (Othoniel et al. 2016). In essence, ecological systems are highly complex and dynamic, and capturing that within an LCA framework has proven to be a difficult undertaking for practitioners (Othoniel et al. 2016). This complexity makes finding a common ground between impact assessment and the pertinence of these impacts in applied, real-world situations, very difficult (Cao et al. 2015).

3 METHODOLOGY FOR COMPARISON

3.1 ECOSYSTEM SERVICE REVIEW TOOL

Part I included a qualitative review of ecosystem services using the Corporate Ecosystem Service Review (ESR) tool developed by the World Resources Institute (Hanson et al. 2012). The ESR offers corporations and businesses to identify the ecosystem services on which they depend and potentially have an impact (Hanson et al. 2012; Bagstad et al. 2013). The output data is purely qualitative, indicating positive and/or negative impacts along with a low, medium, or high company dependence on the ecosystem service (Hanson et al. 2012). The ESR questionnaire presents the results in a summary matrix of 29 ecosystem services (Hanson et al. 2012). During Part I of the project, I collaborated with Hydro to complete the ESR for both Paragominas and Alunorte, and we found that the majority of ecosystem services impacted were regulating services. The results of the ESR for Paragominas and Alunorte can be found in the SI-S3. Cultural services at Paragominas did not seem to be impacted, likely because of the absence of nearby communities of people, and only one provisioning service was heavily impacted—freshwater provisioning. Freshwater provisioning in terms of quantity was addressed in the summer report and the results showed a returned flow rate of 99% at the uppermost corner of the ADI (SI-S6). Freshwater provisioning data is not available for Trombetas, and will not be discussed in this study any further. The results of the ESR conducted for Trombetas is in Section 4.1.

3.2 Accounting for Potential Loss of Ecosystem Services

By using land cover data as a proxy for present ecosystem services, I developed an approach to calculate potential ecosystem service loss based on each land cover present within the respective geographic boundaries (from Part II of this project). Similar approaches have also been used by Comino et al. (2014) and Barnett et al. (2016). For example, in their study in northern Italy, Comino et al. (2014) used relative weighting to address the naturalness of and pressures on land areas for evaluating environmental quality. They used land cover data and a team of experts to create these weights (Comino et al. 2014).

Other studies, such as Jiang and Eastman (2000), Malczewski (2006), Valente and Vettorazzi (2008), and Ferretti and Pomarico (2013) use fuzzy weighting and the ordered weighted average (OWA) approach (Yager 1988) in their respective environmental analyses to aid in spatial planning decisions, such as where to build a housing development (Malczewski 2006). The OWA approach provides more accurate reporting of impacts in comparison to Boolean methods (an "all-or-nothing" approach) (Malczewski 2006). Metzger et al. (2006) directly addressed ecosystem service vulnerability to land use change in Europe as a part of the ATEAM project (Advanced Terrestrial Ecosystem Analysis and Modelling). They focused on the vulnerability and adaptation capacity of humans to cope with changes to ecosystem services based on changes in land use (Metzger et al. 2006). They used fuzzy logic to create one adaptation capacity for twelve indicators for societal welfare (Metzger et al. 2006). Here, I use the ideology behind the OWA to address the ecosystem service provisioning potential of the lands on which Hydro and MRN are mining.

The potentially lost fraction of ecosystem services (PLES), Table 2, was developed from a literature review of different land cover types' ability to provide ecosystem services—it has been strengthened since the project. INFRAS (1998), a research institution in Zürich, also conducted a sustainability assessment using similar approach that weighted sustainability practices from the forestry sector with impact severity. The literature review for the PLES is specific to the rainforest biome, and thus not comparable to other biomes. Additionally, part of the scoring of each land cover type is based on whether the descriptions from Bartholomé and Belward (2005) match the land cover typically found in a broadleaf, evergreen rainforest. The scores are general guidelines for what ecosystem service provisioning should be expected from certain land covers, which is an important assumption to understand when interpreting the results.

The PLES accounts for land cover change impacts from human activity prior to the occupation of Hydro and Vale—the previous owners of the Paragominas mine. The PLES represents the total amount of ecosystem services potentially lost within the geographic boundaries based on the Global Land Cover 2000 (GLC) data from Bartholomé and Belward (2005). Depending on interpretation, the PLES will show the impact of mining in each area while considering ecosystem service damage that occurred prior to Hydro and MRN's land occupation and transformation or the remaining ecosystem services that will be lost because of mining. Once the land is mined, it is assumed to have no ecosystem service provisioning potential until it is rehabilitated.

To use an example based on literature review, regularly flooded shrub/herbaceous cover (Bartholomé and Belward 2005) provides 75% of the ecosystem services found in a typical, evergreen rainforest (Table 2). This is interpreted as 0.75 PLES once Hydro mines on this land. From an LCA perspective, it is important to understand that the PLES score accounts for what exists now and potentially *will* be gone, not the 25% of potential ecosystem services that have already been depleted prior to Hydro's occupation.

Once an area been assigned a score, it will represent the full (100%) yield of ecosystem services. For example, an area comprising 2 km^2, where each km^2 provides 50% of ecosystem services, will yield a 1 km^2 of full (100%) ecosystem service provisioning. I acknowledge that this does not include the problems incurred by edge effects, forest fragmentation, and other ecological concerns. I emphasize that this is purely a way of calculating the general loss of ecosystem services within a given area.

The reference state on which to quantify impacts from Hydro and MRN is based on GLC 2000 data. This means, the current state of ecosystem services as of 2000 is the reference, however, the ambitious no net loss scenario set by Hydro will work towards a state of potential natural vegetation (SI-S4). More information on the literature for the PLES in each land cover type is found in the SI-S5. All geographic boundaries were created in ArcGIS® (ESRI 2014).

Table 2. The PLES scoring table quantitatively explains how each land use type is scored according to the literature review. It is important to understand that these are not value-based scores, but objective valuations of which land cover types found within the mining areas hold a certain percentage of ecosystem services. These represent the potentially lost fraction of ecosystem services (PLES) when Hydro or MRN mine one unit of a specific land cover type. Detailed explanations of each land cover types' score can be found in SI-S5.

GIS ID[6]	Land Cover Type[6]	Site	Score
1	Tree cover, broadleaved, evergreen	Paragominas, Trombetas	1[1]
2	Tree cover, broadleaved, deciduous, closed	Paragominas, Trombetas	1[1]
7	Tree cover, regularly flooded, fresh and brackish water	Paragominas, Trombetas	1[1, 9]
13	Herbaceous cover, closed-open	Paragominas, Trombetas	0.3[6,]
14	Sparse herbaceous or sparse shrub cover	Trombetas	0
15	Regularly flooded shrub and/or herbaceous cover	Trombetas	0.75[4]
16	Cultivated and managed areas	Paragominas, Trombetas	0.5[5, 7, 8]
17	Mosaic: cropland/tree cover/other natural vegetation	Paragominas, Trombetas	0.75[3, 2]
18	Mosaic: cropland/shrub or grass cover	Paragominas	0.6[3, 2, 8]
20	Water bodies	Trombetas	0

[1]Balmford et al. (2002), [2]Felipe-Lucia and Comín (2015), [3]Fritz (2003), [4]Williams (2006), [5]Grossman (2015), [6]Bartholomé and Belward (2005), [7]Costa et al. (2003), [8]Rodrigues et al. (2013), [9]Tockner and Stanford (2002)

3.2.1 Sensitivity of land cover data

A crucial element when dealing with land cover data in the context of ecosystem services is the distinction between primary and secondary forests (Guariguata and Ostertag 2001; Brandon 2014). The GLC data does not distinguish between primary and secondary forests (Bartholomé and Belward 2005). Trombetas is composed of primary forest (Parrotta and Knowles 2001) and the Paragominas forests are secondary (Personal communication Bernt Malme 2016). To account for the impact of secondary forests on ecosystem services at Paragominas, all land cover types that are scored as 1 to represent full ecosystem service provisioning are given a score of 0.75. The 0.75 score is based on literature from Guariguata and Ostertag (2001), Thompson et al. (2011), Brandon (2014), Nahuelhual et al. (2014), and Grossman (2015). I compare the differences between the 1.0 and 0.75 land cover values to see how this influences the mining impacts on ecosystem services in Paragominas.

3.3 DEVELOPMENT OF AN LCIA CHARACTERIZATION FACTOR FOR ECOSYSTEM SERVICES

Cao et al. (2015) comment that a large research gap exists in establishing commonalities that connect inventory databases with ecosystem services in LCA. In this report, I attempt to develop a characterization factor (CF) that includes ecosystem services in LCA at an endpoint level for the specific case of primary aluminum production in Brazil. This study addresses all ecosystem services combined within an area, contrary to the evaluations of select, individual ecosystem services as listed earlier in Table 1. I evaluate the damage to ecosystem services at an endpoint level with a biophysical unit—as opposed to monetary in the case of Cao et al. (2015).

The aforementioned PLES system accounts for part of the need for a common ground since it considers the region-specific ecosystem quality before intervention, the type of land use impact (mining), and the change in ecosystem service provisioning over time. The unit of the endpoint level CF for ecosystem services is PLES $\times$ ton_aluminum^{-1} $\times$ yr^{-1} based on the potentially disappeared fraction of species structure (PDF $\times$ $(m^{x})^{-1}$ $\times$ yr^{-1}) developed by Curran et al. (2011). I also calculated the EF based on the ecoregion data to yield compatible results to other LCIA studies, such as Koellner et al. (2013), de Baan et al. (2013), and Chaudhary et al. (2015).

The concept of PLES could be applied as an additional, separate endpoint indicator to the areas of protection (AoP) endpoint categories: the *potentially disappeared fraction of species* (PDF), which is the current impact category for ecosystem quality; the *disability adjusted life years* (DALY) for human health; and *damage to natural resources*, measured by the increase in resource cost (Goedkoop et al. 2009a). PDF addresses "ecosystem quality," although this only looks at species richness, which is not always an accurate indicator for ecosystem health (Naeem 2008). Incorporating ecosystem services as a separate AoP in LCIA is a proposed method by the UNEP/SETAC Life Cycle Initiative (Personal communication, Verones (2016)).

The FF developed for this study accounts for the yield of square meters (m^2) of cleared land needed to produce one ton of aluminum (Equation 4). The FF was created by examining the m^2 needed for 1 ton of bauxite multiplied with the ton of bauxite needed for 1 ton of aluminum.

$$\mathrm{FF}\left[\frac{\mathrm{m}^2}{\mathrm{ton}_{\mathrm{aluminum}}}\right] = \frac{\mathrm{m}^2_{\mathrm{land}}}{\mathrm{ton}_{\mathrm{bauxite}}} \times \frac{\mathrm{ton}_{\mathrm{bauxite}}}{\mathrm{ton}_{\mathrm{aluminum}}} \tag{4}$$

In this study, the FF does not account for the volume (in m^3) of land mined for two reasons. First, the ecosystem services are assumed to have a higher value in the top soil versus 20 m below the surface. I acknowledge that there are varying levels of impact based on the depth of the mining and how this can affect reforestation; however, here I assume that the overwhelming majority of the ecosystem service benefits are coming from the land surface. Second, I do not have data regarding the bauxite ore grade concentrations at different depths in Paragominas or Trombetas, thus I cannot include a definite volume of mined earth needed to produce 1 kg of aluminum.

The EF quantifies the damage to ecosystem services for each land cover type mined per year, accounting for PLES (Equation 5). When multiplying the PLES per land use i with the land cover mined each year (a_i), we find the total amount of PLES per year. This is then divided by the total area of the region in question (A) to yield the effect that mining will have per year. Here, this method assumes a steady-state, where the same proportion of land is mined each year. However, practitioners with more complete data sets can accurately account for PLES based on the variations of land cover types per year.

$$\mathrm{EF_i}\left[\frac{\mathrm{PLES}\cdot\mathrm{yr}}{\mathrm{m^2}}\right] = \frac{(\mathrm{a_i}\cdot\mathrm{PLES_i})}{\mathrm{A}} \tag{5}$$

Unless created for each year, the EF will assume the same ratio of land use i mined per year for the lifetime of the mine, as is the case here. This is because I do not know exactly which land cover are getting mined in each year, I only know the total land that will be mined by the end of the project. If land cover data is available on a yearly basis, an LCA practitioner could certainly use this method to differentiate impacts on ecosystem services within select years of a project. The resulting endpoint characterization factor for Paragominas is shown in Equation 6. This shows the PLES per ton of aluminum produced each year for the respective mines.

$$\mathrm{CF}\left[\frac{\mathrm{PLES}\cdot\mathrm{yr}}{\mathrm{ton_{aluminum}}}\right] = \mathrm{FF}\cdot\sum_{\mathrm{i=1}}^{\mathrm{N}}\mathrm{EF_i} \tag{6}$$

Although the PLES can account for provisioning services, a study focusing on these may be more accurately captured by the methodology provided by Cao et al. (2015) should the required data exist. An aggregation of the ecosystem services present in each land cover type is the main target of the PLES concept to attempt to bridge the gap between the results derived from impact assessment and their pertinence to users and policy (Cao et al. 2015). This aggregation serves to simplify the complexities and inherent rebound effects (Othoniel et al. 2016) found between and among various ecosystem services.

Most important when evaluating ecosystem services by PLES is that this method does not contain the cultural and intrinsic values of nature. I excluded cultural value considerations in the quantitative analysis of this study because of data restrictions and the deviation from environmental impact analysis towards Social LCA (Zhang et al. 2010b).

To my knowledge, there is no LCA for an aluminum can directly addressing land use. Niero et al. (2016) conducted a cradle-to-cradle LCA with an aluminum can as a functional unit; however, their study focuses more on recycling than inventory and land use. PE Americas (2010) developed an LCIA of aluminum beverage cans for the Aluminum Association, Inc. in Washington, D.C. Although the report addresses five mid-point level impact categories, including global warming potential and eutrophication potential, land use impacts are not evaluated. This makes the comparisons of our impacts to existing studies difficult, if not impossible. In the case study, I used the PLES method and PDF values addressing marginal transformation impacts from intensive forestry using the regional, countryside SAR model from Chaudhary et al. (2015).

3.4 REPLANTING FOR A NO-NET LOSS SCENARIO

I selected the year 1970 as the reference year for the start of logging/grazing due to its use as the baseline year in the Brazil Agriculture and Livestock Census (1970/2006) from the Brazilian Institute of Geography and Statistics (IBGE) (as cited in (FAO 2009)) and in the Living Planet Index (McRae et al. 2014). The reference state is potential natural vegetation for the ecoregions used, which in this case is land cover types pertaining to broadleaf, evergreen rainforest. The reference state was addressed in further detail during the project and will not be repeated here. More information can be found in the SI-S4.

From 1970 until the mine start dates, 1979 and 2007 for Trombetas and Paragominas, respectively, the PLES decreased linearly, based on the anthropogenic changes to land cover outside of the mined areas as seen in the GLC 2000 data (Bartholomé and Belward 2005). Because I do not have historical data of Trombetas' yearly area of forest mined, I averaged the entire area of the mine to date (2014) and divided by the total years of operation (35). I used this same technique for the Paragominas mine, which will include the impacts of Vale. The results of yearly land lost were cross-referenced with Röhrlich et al. (2001) for Trombetas, and

Hydro (2015b) for Paragominas. The results for when Hydro and MRN can expect a no-net loss of ecosystem services is found in Section 4.5.3.

Mining at Trombetas and Paragominas is expected, for the purposes of this study, to continue until 2025 and 2030, respectively. It is possible that the duration of mining will extend several decades beyond those years. The average yearly loss over each mine's lifetime was used to model the average yearly loss of future land. This was to ensure an even comparison between both mines, but it allows for a large uncertainty because this assumes future production remains the same. Restoration times are also linear in approach. The applied estimated time for the new plantings to reach the state of a mature forest is 40 years, based on the literature review in Table 6 in Section 4.5.3.

All land, whether it is mined or used as a tailings dam, is included in the calculations regarding the loss of ecosystem services. This was due to data restrictions since I did not know precisely where the Trombetas tailings dams were located nor the yearly expansion of the dam at either location.

4 RESULTS AND DISCUSSION

4.1 ESR

The Trombetas ESR (Table 3) revealed greater questionable impacts than the Paragominas ESR, primarily for three main reasons: (1) the area is located within a national forest, (2) it is in direct contact with the Amazon River, and (3) its close proximity to the Quilombolas' land. Because the Amazon River is a globally known icon of Brazil, it may hold intrinsic, indirect benefits to people beyond the local scale. The close proximity of Trombetas to land claimed by Quilombolas is shown below in Figure 6.

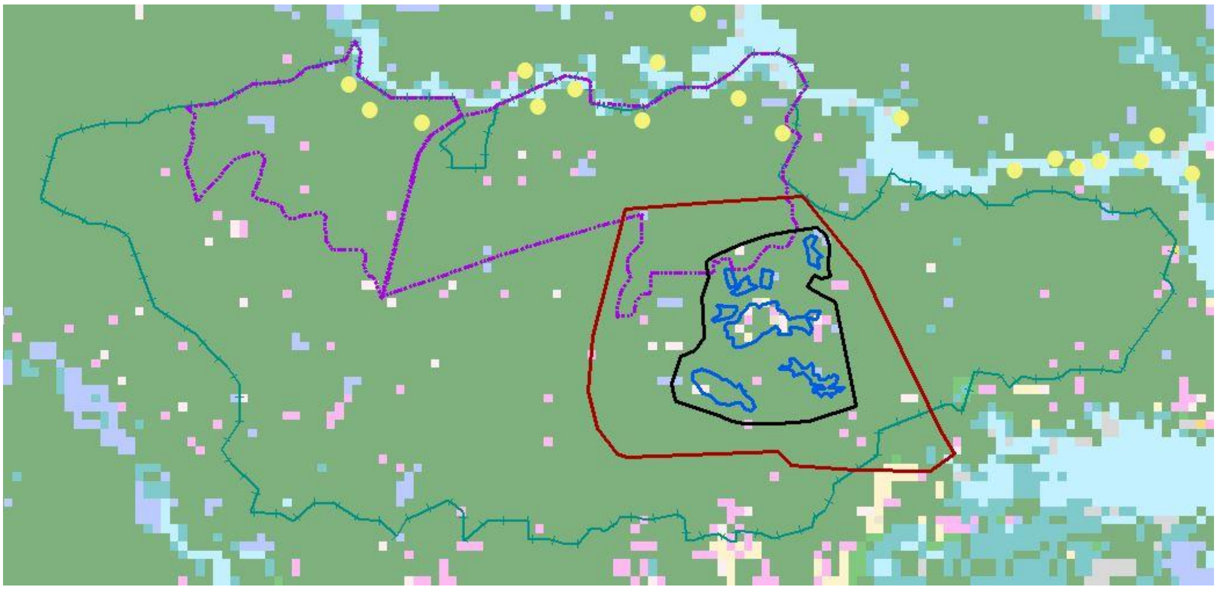

Figure 6. Quilombolas land within the national forest, indicated with purple dash marks, lies just beyond the mine borders, drawn in blue. The Quilombolas land extends within both the ADA and the ADI. Quilombolas land extends beyond the national forest, but for the purposes of this study, I only included Quilombolas land within the FNST. The yellow dots along the river indicate Quilombolas communities in the area (Comissão Pró-Índio de São Paulo 2013b).

Overall, the influence of indigenous peoples, cultural and provisioning resources yielded stronger impacts at Trombetas than Paragominas. The local people collect Brazil nuts, known locally as *castanha-do-brasil*, which is also listed as a vulnerable species according to the IUCN Red List (ICMBio n.d.; IUCN 1998). The local people use hardwoods from the forest as construction materials (Thorkildsen 2014), in addition to harvesting *copaiba*, a medicinal plant used as an anti-inflammatory and anti-bacterial agent (ICMBio n.d.; Veiga Junior et al. 2007). Because of this, cultural factors play an important role in ecosystem services found in Trombetas, especially when considering the anthropocentric nature of what defines an ecosystem service.

Table 3. The ESR for Trombetas reveals only the ecosystem services impact by MRNs mining operations. In Paragominas, there was a visible absence of cultural ecosystem service impacts and far fewer provisioning service impacts. The only ecosystem service impact in Paragominas was freshwater provisioning.

ECOSYSTEM SERVICES DEPENDENCE AND IMPACT MATRIX

Key
● High + Positive impact
○ Low - Negative impact
? Don't know

	TROMBETAS		PARAGOMINAS	
Ecosystem services	**Dependence**	**Impact**	**Dependence**	**Impact**
Provisioning				
Wild foods		● -		○ -
Timber and other wood fibers		?		?
Fibers and resins		?		
Ornamental resources		?		
Biomass fuel			?	
Freshwater	●	● -	●	● -
Genetic Resources		?		
Biochemicals and natural medicines		● -		
Regulating				
Maintenance of air quality	?	?	?	? -
Regional/local climate regulation	?		?	
Regulation of water timing and flows	●	● -	●	● -
Erosion control	?	● -	?	●
Water purification and waste treatment	?	● -	○	?
Cultural				
Ethical and spiritual values		● -		
Educational and inspirational values	?	?		
Supporting				
Habitat quality	?	? -	?	? -

4.2 AREA MINED PER YEAR

The average yearly loss of land at Paragominas is 4 km^2 which produces an average yearly output of 1×10^7 tons of bauxite (Hydro 2015b, 2016a; Personal communication Bernt Malme 2016). To my knowledge, the land use calculations from Hydro represented here include permanent and temporary infrastructure. At Trombetas, the average yearly loss of land based on my calculations is 1.8×10^6 m^2 per year. This is 0.1 km^2 higher than data provided by Röhrlich et al. (2001), who state the yearly loss is 1.70 km^2.

The bauxite output at Trombetas is 45% higher than Paragominas—1.8×10^7 tons annually. This is because the layer of earth containing bauxite is 4+/-1 meters thick in Trombetas, compared to 1.5+/-0.5 meters thick in Paragominas (Personal communication Bernt Malme 2016). Additionally, Trombetas has been reforesting for several decades, so the calculated yearly loss of land used here may be lower now than it was in previous years.

Figure 7, below, indicates the loss of land area from Trombetas and Paragominas. This figure shows the human-induced impacts on ecosystem services from 1970-1979 in the case of Trombetas, and from 1970-2007 for Paragominas. The loss of land is in green and purple for Trombetas and Paragominas, respectively. The yellow line shows the average loss of land per year at Paragominas if secondary forests had not been taken into account. This shows how fragile the results of ecosystem service depletion can be based on modeled land cover data (GLC).

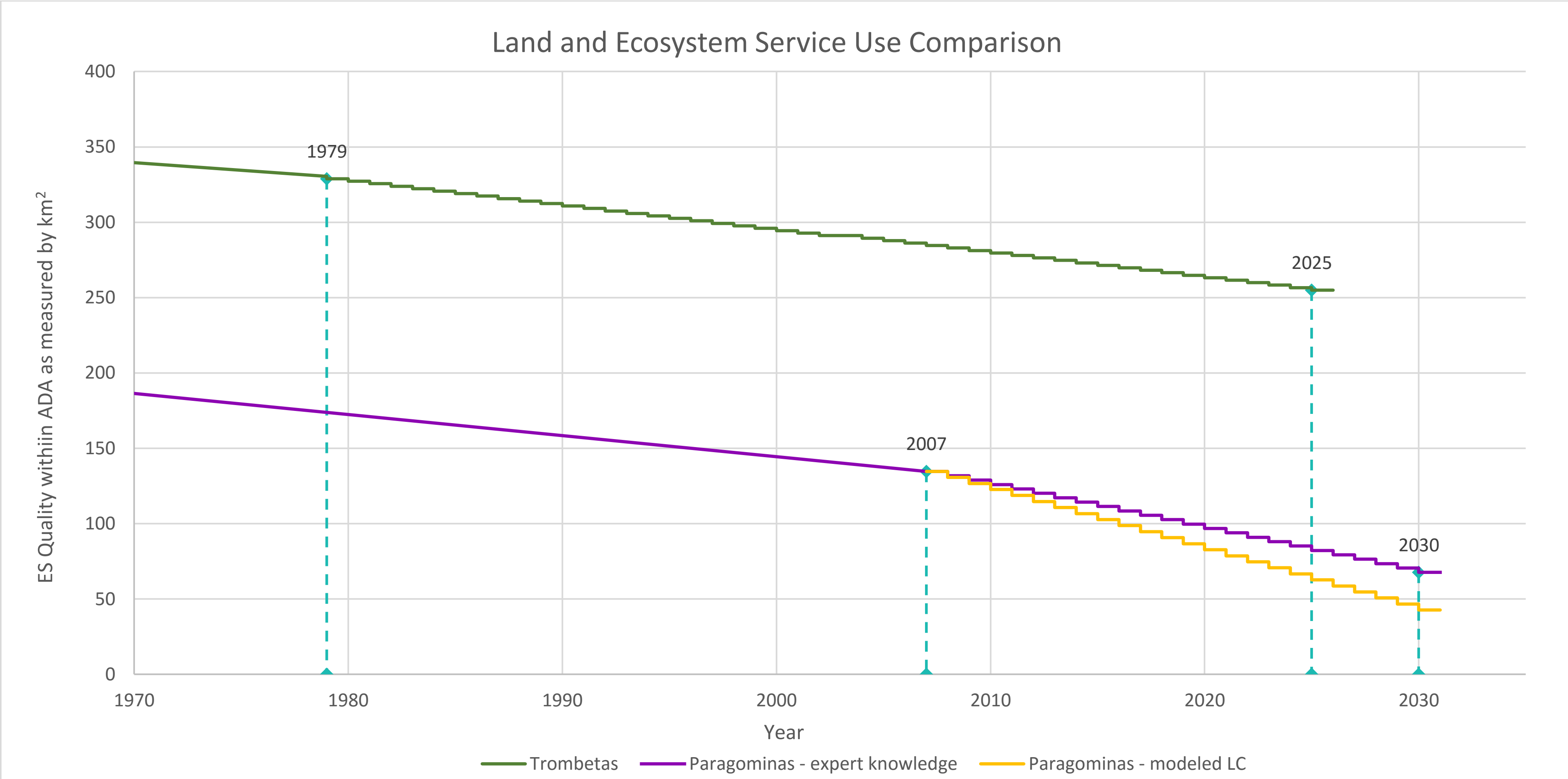

Figure 7. Average yearly loss of land at Paragominas and Trombetas. The green line represents the land clearing at Trombetas, and the yellow line is for Paragominas. The green and yellow lines indicate the ecosystem service loss as estimated by the land cover loss. The yellow line for Paragominas denotes GLC and visually shows the differences between that and expert knowledge (used to graph the purple line). The "steps" in this figure indicate the average yearly loss of mined area within the ADAs. Essentially, every "step" is a portion of land that is depleted. The blue dashed lines indicate the time mining began and will end at Trombetas and Paragominas. This graph does not show the benefit of rehabilitation.

In Figure 7, the impacts at Paragominas are steeper than at Trombetas because Paragominas is mining roughly twice as much land area per year. By the start of mining in Trombetas, there had already been a depletion of 9 km^2 comprising full ecosystem services due to previous anthropogenic changes within the ADA. At Paragominas, 52 km^2 of full ecosystem services have already been depleted (Figure 7) within the ADA. The prior impacts are seen in the linear decrease from 1970 until the respective mining start times.

From 2007 to 2030 and based on the 4 km^2 per year clearing rate, the mining at Paragominas will have disturbed 92 km^2 of area within the ADA of which there are a combined 67 km^2 containing 100% ecosystem services (accounting for the 0.75 PLES in secondary forests). From 1979-2025, Trombetas will have mined 81 km^2 of land of which the combined 100% ecosystem service value is 75 km^2. Although Trombetas is mining less land overall, their impact on ecosystem services per area mined is much greater than Paragominas. In Trombetas, 93% of the land that is mined is considered to have full ecosystem service provisioning. At Paragominas, 73% of the land mined has full ecosystem service provisioning.

If the calculated impact were based on remote sensing data at a global scale, such as the GLC, the impact at Paragominas would be 92 km^2 of lost ecosystem services for the lifetime of the mine. This would result in an additional 25 km^2 of lost ecosystem services over the mine's lifetime. That equates to a 27% increase in damage to ecosystem services when compared to the secondary forest values. Comparing the raw data of land cleared with the PLES data shows how sensitive ecosystem services are to accurate land use data.

4.3 Ecosystem service LCIA values

To calculate the FF, I assumed an average of 5.2 tons of bauxite per one ton aluminum based on data from PE Americas (2010). Based on that value, approximately 2.1 m^2 of land are needed per ton of aluminum at Paragominas (Table 4), while only 0.5 m^2 of land is needed at Trombetas (Table 5). As stated earlier, the area mined per year at Trombetas is lower than Paragominas', although with a higher yield. This explains why Trombetas has a much lower

FF than Paragominas'. It is important to understand that this difference will propagate throughout the impact assessment.

The EF calculations used the PLES from Section 3.2 and either the ADA data or ecoregion data from Olson et al. (2001) to compare impacts. I used the ADA data because it may be valuable for Hydro to see their impacts based on the geographic boundaries in which they operate and own (particularly for the case of Paragominas) (Table 4). I use ecoregion data for the case study following in line with other LCA studies. The characterization factor for primary aluminum production using Trombetas data is one order of magnitude different for both the ADA and ecoregion when compared to Paragominas. Again, this is due to the lower ratio of mined land to bauxite output at Trombetas.

Table 4. Paragominas FF, EF, and CF based on two geographic boundaries. The effect factor (EF) changes according to the geographic boundary used (ADA or ecoregion). The fate factor (FF) remains the same in both situations, as this is a constant value of m2 per ton aluminum.

	Within ADA	Within Ecoregion	Unit
FF	2.08	2.08	m^2/t al
EF	6.51×10^{-2}	1.51×10^{-5}	PLES/m^2
CF	0.14	3.14×10^{-5}	PLES × yr / t Al

Table 5. Trombetas FF, EF, and CF based on two geographic boundaries. The EF changes according to the geographic boundary used (ADA or ecoregion). The FF remains the same in both situations, as this is a constant value of m^2 per ton aluminum.

	Within ADA	Within Ecoregion	Unit
FF	0.52	0.52	m^2/t al
EF	2.61×10^{-2}	3.48×10^{-6}	PLES/m^2
CF	1.35×10^{-2}	1.81×10^{-6}	PLES × yr / t Al

The calculations for Paragominas assume the 4 km^2 yearly rate of land loss and account for the adjusted land cover data based on expert knowledge. When incorporating the expert knowledge into the calculations, the CF at the ecoregion level was 3.14×10^{-5} PLES. Had the GLC data been used at Paragominas, this would have resulted in 4.05×10^{-5} PLES at the ecoregion level (a 27% increase from the site-specific data). As with any model, the output data will only be as strong as the input data. For the case of ecosystem services based on a land use proxy, it is of crucial importance to use the most up-to-date and accurate data as possible.

4.4 CASE STUDY: COMPARING THE IMPACT OF AN ALUMINUM CAN

For this case study, I chose a functional unit of 1000 cans to compare the impacts on ecosystem services between Paragominas and Trombetas. The production of 1000 cans requires 23.31 kg of aluminum ingot (PE Americas 2010). This means 122.3 kg of bauxite is needed to produce 1000 cans when based on a 5.2:1 ratio of bauxite to aluminum, including the lid and tab aluminum alloys (PE Americas 2010).

The impact of producing 1000 aluminum cans results in 7.21×10^{-7} PLES for 1000 cans in the Paragominas ecoregion (NT0170) and 4.21×10^{-8} PLES in the Trombetas ecoregion (NT0173) (Olson et al. 2001) (Figure 8). The higher impact at Paragominas in both cases is due to the larger area mined per year as expected from previous results.

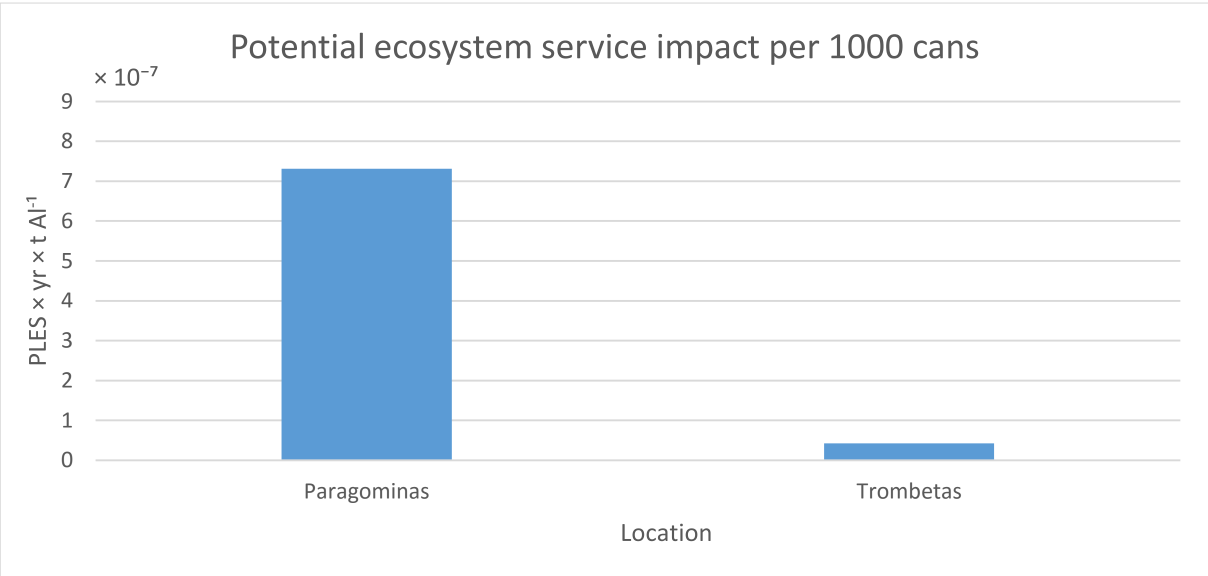

Figure 8. Potential ecosystem service loss per 1000 cans visually represented shows the higher impact at Paragominas than Trombetas. Paragominas exhibits a higher impact per ton aluminum on ecosystem services likely due to the rate of mining per year coupled with a lower bauxite output despite that Paragominas has more human modified land than Trombetas within the ADA.

To compare the results to current research, I multiplied the LCI data for 1000 cans with the CFs for each ecoregion based on the ecosystem quality metric used in LCA (PDF) calculated by Chaudhary et al. (2015). The CF values for Trombetas and Paragominas are 2.89×10^{-12} and 7.05×10^{-12} PDF, respectively. After multiplying with the LCI, these values yield the same trend as the PLES metric for ecosystem services (Figure 9). Trombetas has an impact of 3.51×10^{-14} PDF per 1000 cans, which is an order of magnitude lower than the

3.42×10^{-13} PDF at Paragominas. Again, this is because Trombetas is excavating less area per ton of aluminum due to a thicker bauxite layer than Paragominas, despite the CF for Paragominas being only slightly larger than Trombetas (Chaudhary et al. 2015).

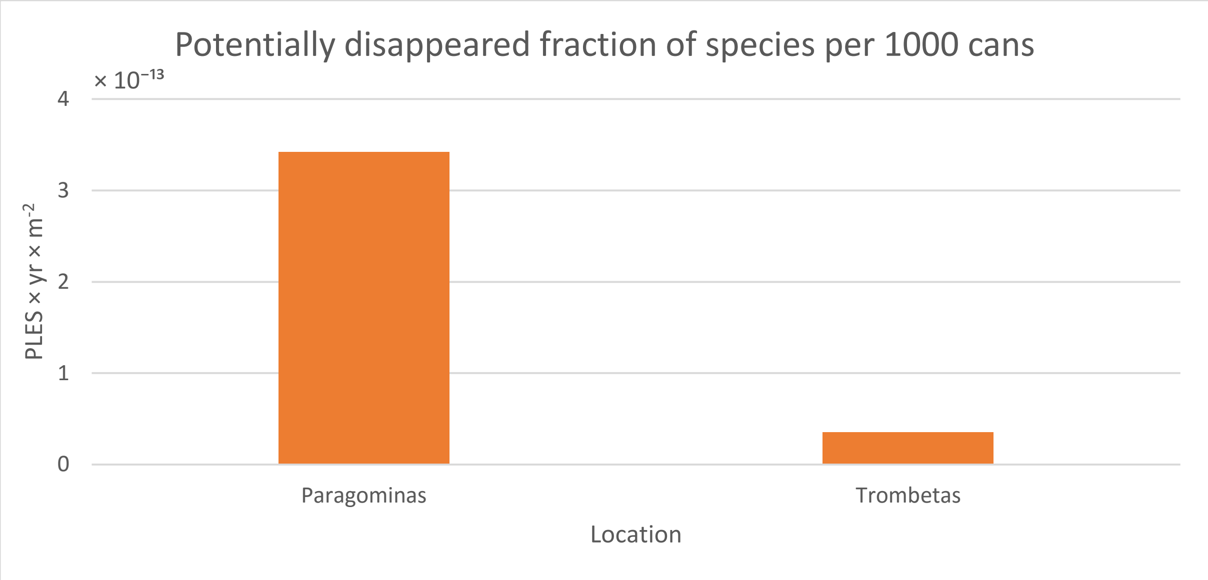

Figure 9. Potentially disappeared fraction of species per 1000 cans yields the same patterns as potentially lost ecosystem services within the mines' respective ecoregions. Paragominas has a higher impact on the potentially disappeared fraction of species per 1000 cans than Trombetas, likely because of higher yearly mining rate coupled with a lower bauxite output than Trombetas.

4.5 Reforestation

Replanting the mining area is required by Brazilian law (Hydro 2015b). Based on the yearly rate of land cleared for mining, Paragominas and Trombetas will need to replant 92 km^2 and 83 km^2 in total, respectively, on the path to no net loss of ecosystem services. These values represent the total area of land lost at a yearly loss rate of 4 km^2 and 1.8 km^2 for each site, respectively. Rehabilitating beyond the aforementioned areas would lead to net positive gain of forest cover and ecosystem services. To reach the 1970 baseline target, Trombetas and Paragominas would have to rehabilitate 90 km^2 and 144 km^2, respectively. The estimated times to reach states similar to primary forest varies for flora, fauna, and ecosystem services.

4.5.1 Biomass

The estimated reforestation times for biomass in Paragominas is 50 years, according to Dr. Fridtjof Mehlum of the Natural History Museum, University of Oslo (Persson Hager 2014). Evidence from Jones and Schmitz (2009) support this recovery time. In their meta-analysis of literature on recovered ecosystems, they found an average 42-year recovery time for forest ecosystems (both boreal and tropical) (Jones and Schmitz 2009). A similar study by Aide et al. (2000) found that floral species richness, aboveground biomass, and basal area in Puerto Rican secondary forests were similar to old growth forests (above 80 years old) after 40 years. Based on a small literature review (Table 6) and the estimates from Dr. Fridtjof Mehlum, Hydro can expect to reach a no-net loss state of biomass after 40 years of regrowth, provided there are no die-outs, blights, flooding, or other hazards to hinder forest growth. I assume the 40-year time horizon to also hold true for Trombetas barring any potential hazard.

Table 6. A small literature review of forest chronosequencing studies show the median time for a newly planted area of reforestation to match the species richness composition of a mature forest is 40 years.

Author	Ecosystem studied	Location	Prior Activity	Recovery time (year)
Jones and Schmitz (2009)	boreal and tropical forest	varied	logging/agriculture	42
Aide et al. (2000)	tropical forest	Puerto Rico	abandoned pasture	40
Saldarriaga (1985)	tropical forest	Venezuela, Colombia	slash/burn	40
Brown and Lugo (1990)	tropical forest	varied	varied	35
Zimmerman et al (2007), as cited in (Chazdon 2008)	temperate forest	Czech Republic	pasture	40
Vesk et al. (2008)	temperate forest	SE Australia	agriculture	25
Parrotta et al. (1997)	tropical forest	Trombetas, Brazil	mining	10*
Bullock et al. (2011)	varied	varied	varied	30-40
Dunn (2004)	tropical forest	varied	varied	30-40

*(50% of theoretical maximum)

Lamb et al. (2005) show before and after photographs from Trombetas after 10 years of replanting. Dr. Parrotta, a co-author on that study, reported in 1997 that after 10 years of regrowth, the forests reached roughly 50% of the theoretical maximum species richness found in the primary forest—visible in Figure 10 (Parrotta et al. 1997). Additionally, species that were not planted but native to the primary forest were found in the replanted areas, indicating wind and faunal seed dispersal (Parrotta et al. 1997).

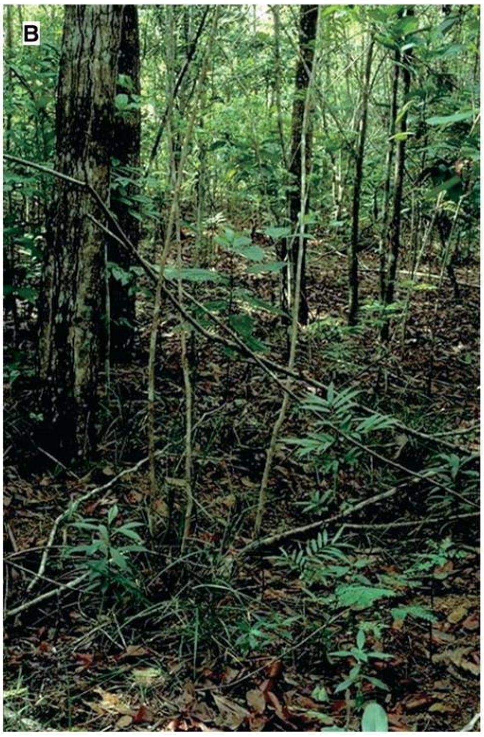

Figure 10. An aerial and ground view of Trombetas (Picture A and B, respectively) show the recovery of biomass (pictures from Lamb et al. (2005)). Picture A shows a part of the bauxite mine, and visible in the middle are small patches of regrowth. Picture B is a shot from a plot of land 10 years after replanting.

For the time being, the planned monitoring of the area is for 30 years after the reforestation projects have ended, to ensure the success of the program (Hydro 2016b). The program may fall short as a sufficient time to assess the progression of reforestation efforts (Aide et al. 2000; Vesk et al. 2008; Jones and Schmitz 2009; Cunningham et al. 2015). Based on research (Table 6), it is advisable that Hydro continues to monitor the progression of reforestation up to 40 years at least. The first years of reforestation are fairly indicative of survival success (Saldarriaga 1985; Breugel et al. 2011)—a crucial time period for Hydro to ensure successful reforestation.

4.5.2 Fauna

Dr. Fridtjof Mehlum also stated that the recovery of fauna species could take up to 150 years at the Paragominas location (Persson Hager 2014). Data from Liebsch et al. (2008) Vesk et al. (2008), Curran et al. (2014), and Cunningham et al. (2015) support Dr. Mehlum's statement. Vesk et al. (2008) remarked that mature trees typically do not bear boughs or hollows until about 100 years of age. This means that a revival of arboreal mammals and selected bird, bat, reptile, and amphibian species could take at least a century in the reforested area (Eyre et al. 2010; Cunningham et al. 2015). Observations by Parrotta et al. (1997) support this statement. Ten years after replanting in Trombetas, they found a scattering of individuals from species of tapir, armadillo, and deer, to name a few, that had returned to the area, but no confirmation of the two troops of red howler monkeys (*Alouatta seniculus*) that used to occupy the area prior to mining (Parrotta et al. 1997).

In a study by Cunningham et al. (2015) reforested areas not designed for harvest are more probable to yield greater environmental benefits than those that are. Both the Paragominas and Trombetas mines are being reforested without the intention of future harvesting. Cunningham et al. (2015) graphically represent the revival time of different taxonomic groups in actively reforested areas in Figure 11.

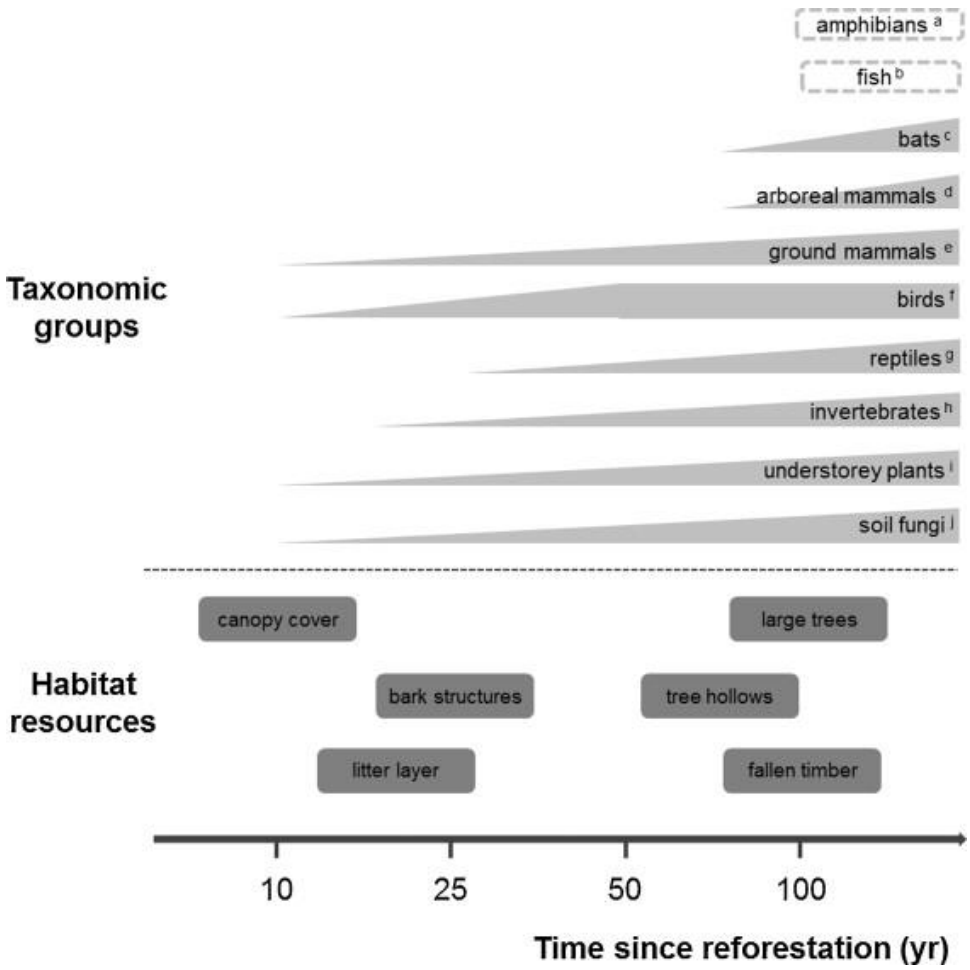

Figure 11. Taxonomic rehabilitation timeline adapted from Cunningham et al. (2015).

Fauna recovery in the two ecoregions where Trombetas and Paragominas are located is anticipated to take an average of 171 years over three different taxa—herpetofauna, mammals, and birds (Curran et al. (2014) as cited in Chaudhary et al. (2015)). Liebsch et al. (2008) studied reforestation efforts on 18 different forest areas in the Brazilian Atlantic Forest. They also concluded similar results to (Curran et al. 2014): 167 years are needed to reach a mature forest (Liebsch et al. 2008).

Other important considerations for reforestation, such as soil fertility and basal diameter (Breugel et al. 2011), and the application of lime, nitrogen, phosphorous, and potassium (Evans et al. 2013), to name a few, are not included in this study. Soil organic matter, magnesium, and manganese are also important nutrients and elements for replanting on areas mined for bauxite (Ferraz 1993). Parrotta and Knowles (2001) explain that the handling of topsoil is an important aspect for reforestation, and the seedlings need a certain amount (unspecified) of topsoil to reach healthy growth (measured in tree basal area, crown cover, and height of canopy). Currently, topsoil at both Paragominas and Trombetas is removed and stored before being reused for reforestation purposes, although the topsoil may not retain its full amount of nutrients or an active seedbank at the time of replanting (two years after clearing for Paragominas) (Lamb et al. 2005; Hydro 2015b).

4.5.3 Ecosystem services

Both flora and fauna play an integral role in maintaining and underpinning ecosystem services, which adds to the complexity of when ecosystem services should return (MA 2003; Rey Benayas et al. 2009; Chazdon 2013; Science for Environment Policy 2015b). A "full" recovery may not be a copy of the original, primary forest and little research exists on a no net loss scenario of ecosystem services as such (Marin-Spiotta et al. 2007; Chazdon 2013). To my knowledge, there are no studies explicitly stating the time needed for a reforested area to fully provide the ecosystem services as it did in its original state.

Based on the 40-year and 170-year time horizons for flora and fauna recovery, respectively, a full recovery of ecosystem services will likely occur between these two dates.

The secondary forests in this study were scored to hold 0.75 ecosystem services which was a value built upon literature. With an assumed primary forest in 1970, I do not know when the secondary forests had been modified, how old they were, or at what stage in their development they were in when modeled as rainforest land cover. Even so, a 50 year-old forest is still on the younger end of the 40–170 year spectrum for a return to ecosystem services.

Because ecosystems are so complex, the precautionary principle should be adopted when addressing impacts to ecosystem services. Some ecosystem services may be present within 40 years, but some may take longer depending on location, extent of damage, presence of certain species of flora and fauna, and other factors (Alexander et al. 2016). Based on the information presented in this report, I assert that in order to provide the same ecosystem services as the original, deciduous, evergreen rainforest, the biomass of the reforested area would need to be mature and possess evidence of increasing faunal diversity.

4.5.4 Rehabilitating areas containing tailings

The rehabilitation of tailing ponds began in 1989 at the Trombetas mine (MRN 2012a). Excess water is siphoned out of the tailing ponds (assumed to be reused for further mining) once they have reached a solid content of 35–40% (MRN 2012a). Nitrogen-fixing plants, such as legumes, are first planted via a technique called hydroseeding (MRN 2012a). Native vegetation is then planted once the soil is suitable. Despite the efforts, however, rehabilitating tailings dams is still a very large challenge for retired mines and the hydroseeding efforts do not always work as well as anticipated (Personal communication Bernt Malme 2016).

Despite no clear rehabilitation plan for tailing ponds, there are still ways to reduce potential impacts on ecosystem services. Ensuring the mechanical stability of the pond will help prevent breakage and leaks, excess rainwater or flooding runoff, leaching of toxic materials, and the spread of wind borne particles (Hansen n.d.). Reforesting on tailings dams requires special consideration due to potentially higher heavy metal content and other contaminants (Environmental Law Alliance Worldwide 2010).

5 CONCLUSION

5.1 HYDRO'S IMPACT

The quantitative results showed that Paragominas has a much higher impact on ecosystem services than Trombetas for several reasons. First, Paragominas has a higher ratio of mined area to bauxite output per year than Trombetas. This could be attributed to discrepancies in the data, such as not knowing the mining trends and outputs at Trombetas. However, the quantitative evaluation of ecosystem services provisioning in this study are purely environmental. Based on the ESR results, Trombetas has a larger cultural impact than Paragominas. This is especially important to consider alongside the quantitative results, as ecosystem services are anthropocentric in nature. Trombetas, to my knowledge, has more Quilombolas groups actively using the ecosystem services in the area than Paragominas.

5.2 GREATEST AREA OF IMPROVEMENT AND NO NET LOSS OF ECOSYSTEM SERVICES

Hydro's inquiry into their impact on ecosystem services is to discover what ecosystem services were present, how they are being impacted, and how this can be improved. For the Hydro value chain (addressed previously), from Paragominas, to Alunorte, to Albras or Sunndalsøra, by and large the greatest area to improve ecosystem service provisioning is at Paragominas. A successful rehabilitation program for tailing ponds has yet to come to fruition (Personal communication Bernt Malme 2016). Because of this, a plan for no net loss of ecosystem services should focus not only on rehabilitation of mined lands, but could also look for alternative ways to further reduce the amount of waste that enters the tailing ponds.

5.3 LCIA AND THE INCLUSION OF PLES

Inherent to the CFs produced in this study are the differences in FFs from each mine. The Trombetas FF accounts for the 0.5 m^2 of land needed per ton aluminum, whereas the Paragominas FF value is 3.3 m^2 per ton aluminum. Using a global, average value of land per ton aluminum, such as 1 m^2 per ton aluminum as suggested by IAI (2009a) may be more applicable on a broader scale for a global FF. However, because this study is for, and specific to, Hydro, I opted for the most accurate values based on the data I calculated.

Ideally, the concept behind the incorporation of ecosystem services into LCA should be the takeaway message. Here, I grouped the likely prevalence of all ecosystem services within a certain land cover instead of pulling apart select ones. Integrating ecosystem services in LCA at an ecoregion level, like in the case study, would be a remarkable challenge to apply to all terrestrial ecoregions around the world and would require tremendous amounts of region-specific data. When introducing ecosystem services into LCA in a feasible manner, I suggest using the fourteen biomes that Olson et al. (2001) used to formulate the ecoregion data used for this study. The scoring of different land cover types should remain consist within each biome, as each reference state should be angled towards the reference state of that biome. For example, in this study, the reference state is the quantity of ecosystem services in a rainforest under a potential natural vegetation scenario, but for a grassland or tundra, the reference state on which to score the different land cover types should be relative to grassland or tundra.

The PLES method does not address different time horizons, such as ReCiPe's hierarchist or egalitarian perspectives (Goedkoop et al. 2009a), nor does it address the differences in land transformation and/or occupation impacts from an LCA perspective. As stated in a literature review by Othoniel et al. (2016), these are important considerations for developing a characterization factor for ecosystem services. The PLES also does not include seasonal variation, which may be important for coastal areas (Othoniel et al. 2016), but instead provides an average estimate on the state of the ecosystem services as a whole.

As mentioned by Zhang et al. (2010b), a current knowledge gap exists in finding a method that reflects all ecosystem services. Despite its limitations, the PLES method may provide some further insight into the creation of region-specific characterization factors for ecosystem services. The PLES in this case study is spatially explicit for a rainforest biome, and different EF values would have to be created for other terrestrial biomes, such as coastal, grassland, and tundra, for example, to implement on a global scale. This would provide applicable and comparable results in LCA across the globe and would likely be the most consistent way of evaluating impacts to ecosystem services overall. However, the methodology could be finely tuned to region- or site-specific cases, as it was for Paragominas.

Site-specific data, as seen in the case study, is especially important to consider when evaluating ecosystem services. Here, the damages to ecosystem services were overestimated using the modeled land cover data compared to expert knowledge. In some cases, the damages may be underestimated depending on the accuracy of the model used. Regionalization is highly important to consider when evaluating ecosystem-related impacts Pfister et al. (2009); (de Baan et al. 2013; Hellweg and Milà i Canals 2014).

Addressing impacts from a reference state or from the current state would be up to the discretion of the practitioner. In this case study, the reference state to which Hydro would like to return is natural vegetation cover (rainforest). However, the PLES encompasses the impacts created before Hydro's operations (i.e. the change from rainforest cover to pasture lands several decades ago). For consistent and comparable results among other studies, it would be advisable to use the potential natural vegetation as the reference state. In an ever-changing world, the impacts that humans cause now will have different effects than if those impacts were caused 25, 50, or 100 years ago—a difference that the PLES methodology can account for.

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SUPPORTING INFORMATION

Linking ecosystem services and damages from bauxite mining in an LCA context

A case study from Hydro and a movement towards no net loss of ecosystem services

Supporting Information

11 Pages

1 Figure

3 Tables

1 Equation

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Master's Thesis

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S1. TABLES OF CONTENTS FROM PRIOR REPORTS

Table S1. Table of contents from the summer report (Part I), submitted August 18 2015. The tables of contents for Part I and Part II are to provide the reader with information that has already been addressed in this yearlong project.

Table S2. Table of contents from the Master's project (Part II), submitted December 18, 2015.

S2. BIODIVERSITY, NATURAL CAPITAL, AND ECOSYSTEM SERVICES

Two aspects that are highly interconnected to the concept of ecosystem services are natural capital and biodiversity. Natural capital is a natural stock delivering a flow of renewable and non-renewable goods and services (Costanza and Daly 1992). However, most studies and classifications of ecosystem services do not include abiotic, non-renewable capital as an ecosystem service (Costanza and Daly 1992; Costanza et al. 1997; Turner and Daily 2007; Haines-Young and Potschin 2013; Science for Environment Policy 2015a).

Researchers argue that biodiversity should be excluded as an ecosystem service because it under pins and aids in ecosystem service evaluation rather than being a stand-alone ecosystem service (Daily 1997; Hanson et al. 2012; Mace et al. 2012; Schröter et al. 2014; Bartkowski et al. 2015; Science for Environment Policy 2015a). In their meta-analysis between the connection of the benefits of ecosystem restoration to ecosystem services and biodiversity, Rey Benayas et al. (2009) state that "[…] *biodiversity is positively related to the ecological functions that underpin the provision of ecosystem services*", despite that the mechanisms and relationships of these underpinnings are still "[…] *poorly defined.*" The exclusion of biodiversity as an ecosystem service in international documents (MA 2005; Sukhdev et al. 2010; Haines-Young and Potschin 2011; Science for Environment Policy 2015a) go along with recent literature on the subject of how it should be classified alongside ecosystem services (Hanson et al. 2012; Mace et al. 2012; Schröter et al. 2014; Bartkowski et al. 2015). As a result, I have chosen to exclude biodiversity as an ecosystem service in this assessment.

S3. ESR RESULTS FOR PARAGOMINAS AND ALUNORTE

Table S3. ESR in Brazil. This table comprises all ecosystem services that face a direct impact either at Paragominas and/or Alunorte. These results were derived from the ESR conducted during the summer of 2015. More information on the ESR is found in the SI-S1. The table format and listed ecosystem services are from the ESR, developed by Hanson et al. (2012).

ECOSYSTEM SERVICES DEPENDENCE AND IMPACT MATRIX				
Key ● High ○ Low + Positive impact - Negative impact ? Don't know	**PARAGOMINAS**		**ALUNORTE**	
Ecosystem services	**Dependence**	**Impact**	**Dependence**	**Impact**
Provisioning				
Wild foods		○ -		
Timber and other wood fibers		?		
Fibers and resins				
Animal skins				
Biomass fuel	?			
Genetic Resources				
Biochemicals and natural medicines				
Freshwater	●	● -	●	● -
Regulating				
Maintenance of air quality	?	? -		● -
Global climate regulation				● -
Regional/local climate regulation	?			
Regulation of water timing and flows	●	● -	●	
Erosion control	?	●	?	● -
Water purification and waste treatment	○	?	?	● -
Cultural				
Recreation and ecotourism				?
Ethical and spiritual values				?
Supporting				
Habitat quality	?	? -		● -

Freshwater provisioning is undoubtedly affecting the local water supply due to the extensive amount that Hydro uses. Based on the ESR, this is the main provisioning service that needs investigation to see how far downstream the water use at Paragominas has on the ecosystem. This would also be important to consider throughout the entire value chain. Water purification and waste treatment indicated a low level of company dependence and questionable impact because of the lack of knowledge regarding the purity of the water upon its return to the ecosystem. Due to the extensive use of electricity at Alunorte (including the primary production plant, Albras), there is likely to be an impact on air quality and global climate regulation.

S4. Reference state: potential natural vegetation

The natural state of ecosystem services is based on suggested LCIA land use reference states in Goedkoop et al. (2009b), Koellner et al. (2013), and Chaudhary et al. (2015). In the ReCiPe life cycle impact assessment (LCIA) methodology developed by Goedkoop et al. (2009b), they suggest using potential natural vegetation (PNV). A concept promoted by Ramankutty and Foley (1999), PNV reflects the "[…] *vegetation that would most likely exist now in the absence of human activities* […]". Chiarucci et al. (2010) argue that the dynamism of ecosystems cannot be captured by PNV because of the different synergies between large mammals, soils, and biological invasions that occurred before humans. Humans have also managed forests and wild fires which has limited the ability of practitioners to predict whether the current forest and vegetation states would actually have existed without human interference (Chiarucci et al. 2010).

S5. PLES Literature Review

The following is an excerpt from the Master's project that details the literature behind the PLES scoring system. Some additions have been made since then to strengthen the literature review supporting the PLES.

In a study by Felipe-Lucia and Comín (2015) of ecosystem service provisioning and biodiversity in relation to different land use types in a riparian forest in north-east Spain. They not only found evidence that supports Balmford et al. (2002)'s conclusion of forests providing higher amounts of ecosystem services, but also gave greater detail on ecosystem service provisioning in mosaic landscapes. When looking at ecosystem services such as CO_2 sequestration, nutrient regulation, and habitat provisioning, Felipe-Lucia and Comín (2015) found that cropped and/or mosaic areas provided some ecosystem service benefits, but not as much as the riparian forests. Because cropped areas did provide ecosystems services, but not always to the full extent of forests (Felipe-Lucia and Comín 2015), these are weighted lower than riparian forests (Table 2). Additionally, in all the ecosystem services assessed, urban areas, which are most in line with Bartholomé and Belward (2005)'s "*artificial surfaces and associated areas*", supported none of the ecosystem services mentioned above (Felipe-Lucia and Comín 2015).

Defining an appropriate weight for mosaic and cultivated land cover types is not easy, as these land cover types can vary in ecosystem service provisioning depending on geographic location, topographic location, amount of natural forest coverage remaining, and type of cultivation occurring (Felipe-Lucia and Comín 2015). Felipe-Lucia and Comín (2015) found that different types of agricultural use, such as fruit groves, poplar groves, dry cereal croplands, and irrigated croplands, provided various levels of ecosystem service provisioning. Grossman (2015) examined the impacts of agricultural practices in eastern Paraguayan forests on ecosystem services, and found that services such as net primary productivity and soil organic carbon (carbon sequestration) were reduced by almost 50% on largely cultivated lands. For this reason, I have weighted Bartholomé and Belward (2005)'s land cover type "*cultivated and* *managed areas*" at 50%, i.e. we assume that half of the original ecosystem services persist. Grossman (2015) also included forest bird biodiversity as one of his supporting ecosystem services.

The description of a "*mosaic cropland*" landscape provided by Fritz (2003), indicates that the majority of the area is cropland and the remaining forest or vegetation cover is of degraded quality. As a result, I have weighted mosaic cropland/tree cover and cropland/shrub at 75% and 60% for shrub cover. I presume that the tree cover will provide more ecosystem services, but not full ecosystem services because of its degraded state. Thus, the 75% accounts for the loss of ecosystem services as a result of forest degradation, but remains higher than the 50% decrease of agricultural land occupation. The cropland/shrub land cover type is lower than the mosaic cropland/forest because based on the presumed natural state of all rainforest, it is estimated that these areas of shrubs were once forests, but got deforested and became shrub lands. It is presumed that these shrub lands will provide a higher level of ecosystem services than agriculture, but not as high as the mosaic cropland/tree cover; hence, I have weighted this land cover category at 60% (i.e. 60% of services remaining). Herbaceous cover and regularly flooded shrub/herbaceous cover is given a weighting less than 100%, because they are presumed to have less ability to fully provide ecosystem services based on the aforementioned logic of the presumed natural state.

Herbaceous cover (GIS ID 13), is weighted at 30%, because Bartholomé and Belward (2005) describe this land cover as "[…] *plants without persistent stem or shoots above the ground*". Thus, the ability for CO_2 sequestration, nutrient retention, erosion control, and other regulating and provisioning ecosystem services will decrease. Likely, the shoots in the soil are contributing to some erosion control, but are most likely not strong enough to prevent heavy erosion and/or landslides. The 30% is an arbitrary value derived from this pragmatic approach towards ecosystem service provisioning. The final land cover type to be addressed is the regularly flooded shrub/herbaceous cover (GIS ID 15), weighted at 75%. This is because the pixel with this GIS ID is adjacent to a 100% ecosystem service providing land cover type: tree cover that is flooded regularly with salt water (GIS ID 8). Thus, I assume the area represented in the pixel was once cleared, but now remains mostly untouched by human activity. Since this pixel is representative of a wetland and potentially has a "*sparse tree layer*" (Bartholomé and Belward 2005), it is likely to have higher ecosystem service provisioning than just herbaceous or mosaic herbaceous land cover (Williams 2006). Thus, regularly flooded shrub/herbaceous cover is weighted at 75% due to its location and presumed potential to provide ecosystem services.

Rodrigues et al. (2013) address the changes in Amazonian soil quality and functions after the changes from forest to agriculture. They noted a significant decrease in soil quality and an increase in biotic harmonization among microbes, which could have an impact further along the ecosystem web (Rodrigues et al. 2013). Tockner and Stanford (2002) indicated that the conversion of floodplains to agricultural land cover can alter the ecosystems so significantly, it renders them functionally extinct (Tockner and Stanford 2002). Hence, floodplains are allocated a full value in the PLES system, because if they are impacted, the resulting damage is likely to be very severe.

Land cover type 14 is allocated 0.0 ability to hold ecosystem services, because the geographic data matched the mining area for Trombetas; therefore, I assumed land cover type 14 to be open mine and have no ecosystem service provisioning. Land cover type 20 is also listed as a 0.0 because I am only focusing on terrestrial ecosystem services in this study.

S6. FRESHWATER PROVISIONING

For the most holistic overview of freshwater consumption, I used the average values from 2010-2014 because of a dip in production from the years 2013-2014, when Alunorte was not operating at full capacity. This ensures that the results reflect the average overall impact that Hydro has had, and is having, on ecosystem services in the watersheds. I used natural flow data from Verones et al (2013); natural flow, as defined by the Food and Agriculture Organization (FAO), is "*the average annual amount of water that would flow into the country in natural conditions...without human influence.*" (FAO 2003). The calculation of the natural flow data from Verones et al (2013) fit the FAO definition with the inclusion of precipitation and river and stream flow (FAO 2003). Based on the information provided by Hydro, an average of 18,138,996 m^3/yr of water is taken from surface water (SW) and 3,270,925 m^3/yr year from rainwater (RW); 9,006,017 m^3/yr are returned to the watershed from the tailings dam, totaling 21,409,921 m^3/yr in natural flow. The total water consumed, 12,608,728 m^3/yr, is subtracted from the natural flow (Equation 1). This is then divided by the incoming river flow in order to find the impact in relative terms (percentage) in the watershed (Equation 2). These steps are repeated for every next, largest pixel along the river network to find the overall impact.

$$river_{out} = river_{in} - \left(\sum consumed - \sum returned\right) \qquad (1)$$

$$\frac{river_{out}}{river_{in}} \qquad (2)$$

Based on the natural flow data from (Verones et al. 2013), the starting point has an annual flow of 13,167,846 m^3/yr. After I removed the net consumed water, I found that the return to a 95% or greater, natural flow rate happens just 4.6 kilometers northwest of the suggested point of extraction, well within the ADI. At that point, the river from which Hydro extracts joins a larger river. In fact, roughly around the edge of the ADA is where the increase from 4% to 95% occurs. At the northern most point of the ADI, the river flow reaches 99% of its original value- approximately 20 kilometers downstream from the extraction point (Lehner et al. 2006). Again, the models on which the calculations are based will have inherent uncertainties and assumptions which can propagate through this study.

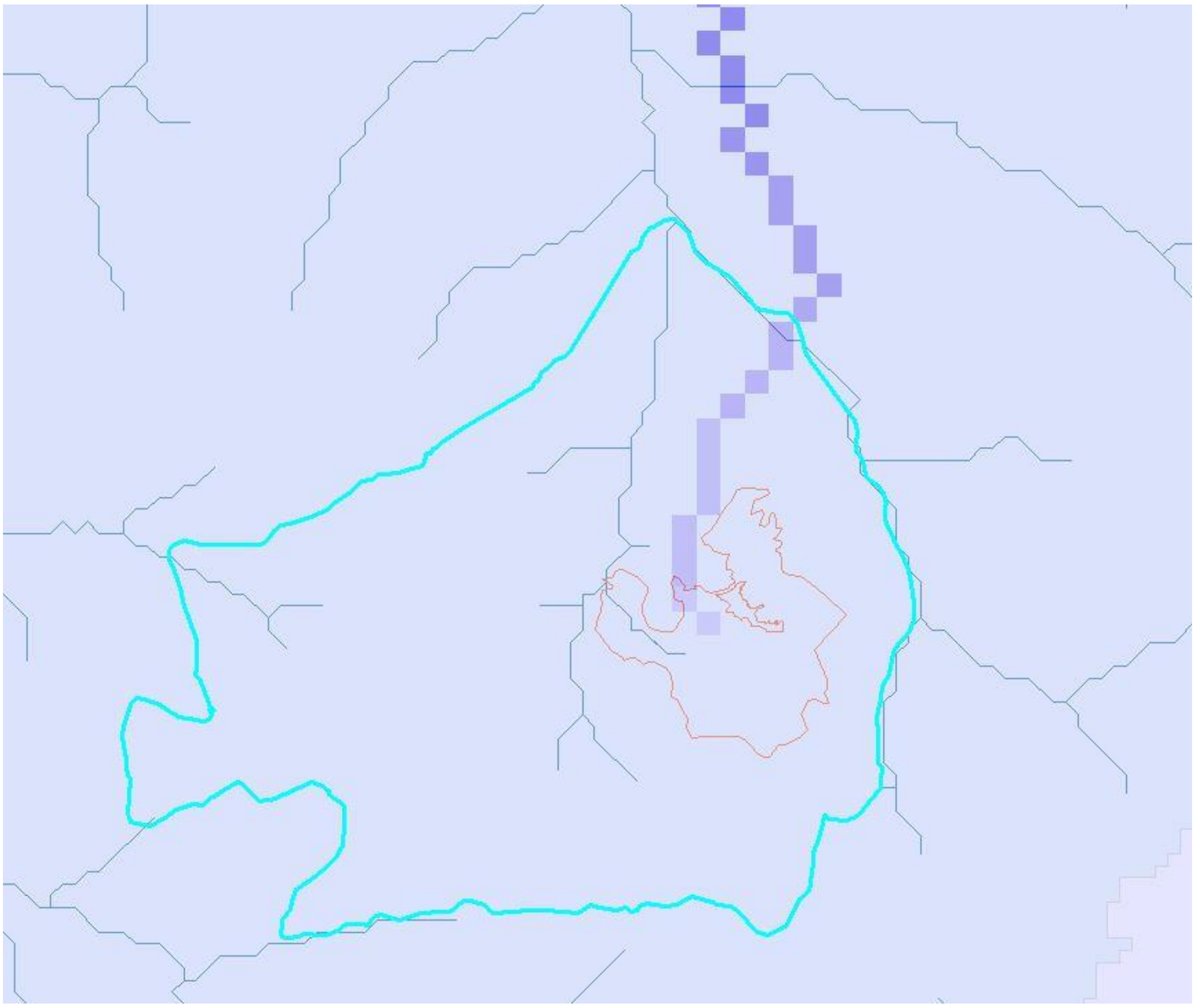

Figure S1. Freshwater provisioning. The pixel wholly inside of the mine represents 4.2% of the original flow. However, the next pixels increase in darkness, with a starting value 95%. The darker, purple pixels represent higher percentages. As mentioned in the text, the flow data is visually not in line with the stream network (Lehner et al. 2006) because of offsets in the natural flow model.