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Assessing the impacts of water on industry–the case of Asia Pacific Breweries (APB)

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Abstract

Industrial water use accounts for 22% of global water consumption and for as much as 60% in high-income countries. Thus, it is a considerable factor when dealing with global water issues. This paper assesses the impacts of water on the brewing industry using the case example Asia Pacific Breweries (APB). By deriving a true cost of water (TCW) via a three-step approach involving (1) desalination, (2) secondary treatment and (3) reclamation, it reflects all water-related risks from physical availability to reputational, environmental, and legal risks in the context of a value-based management framework that embeds sustainability in the company's operational conduct. An excel-based sensitivity model takes into consideration all relevant drivers allowing for a detailed cost breakdown for each of the three steps. The resulting transparency of what drives the TCW yielded that the main components are energy, carbon, and capital costs. As derivatives on all three components exist, they were used to feign a hypothetical water option by replicating its pay-off as a weighted average of energy and carbon derivatives. At APB water accounted for merely 0.4% of total operating costs, but drove 335 times its cost in revenues. The resulting biomass from secondary treatment was also considered for energy recovery through anaerobic digestion and thermal hydrolysis. As a result, secondary treatment energy costs were subsequently reduced by 56%. As a result, the application of discriminatory pricing for industrial end-users of water was, therefore, concluded a viable option. Still, the demand-price elasticity of the company's products needs to be considered when applying said option to avoid the passing on of costs to consumers. A practical approach is considered involving a revenues per m^3 of water over the TCW metric to determine the extent of discriminatory pricing and temporary tax breaks to avoid the passing on of costs to consumers.

Key words: business, carbon, desalination, discriminatory pricing, drivers, energy, impacts, industry, True Cost of Water, value-based management, water, water treatment

INTRODUCTION

Freshwater and energy constitute two of the most essential and inseparable commodities for human life on earth. Industry, as the spine of modern society, however, places significant demands on both resources (Gude *et al.* 2010) as it constitutes 22% of global water demand (UNESCO 2003b) and even up to 59% of demand in 'high-income countries' (WBCSD 2009). The idiom 'no water, no business' illustrates all too clearly the inherent interdependence between industry and water (WBCSD 2009). As water is often underpriced, it is seldom adequately accounted for in production processes and product pricing. This leads to unsustainable water usage which takes its tolls on the environment and eventually societal wellbeing. In light of population, economic and industrial growth, water, which was long treated a ubiquitous and abundant resource, is now increasingly becoming the constraining factor to the very growth it is fueling (Veolia Water 2010). This ouroboros relationship necessitates a thorough understanding of what creates value and what drives costs. Therefore, this paper aims to provide a framework to assess the role of water for the brewing industry from a management and sustainability perspective. A value-based management (VBM) evaluation framework is applied using the

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case example of Asia Pacific Breweries (APB). By linking all the relevant drivers with the three pillars of sustainable development (SD) in an excel-based sensitivity analysis, the True Cost of Water (TCW) is calculated to reflect all water related risks whilst considering relationships such as the energy water nexus and energy recovery from anaerobic digestion.

The TCW is a figure that takes into consideration all risk factors associated with water and converts them into a cost figure. Conceptually, it assumes that the cost of absolute water scarcity is the cost obtained by sourcing and treating water via a three-step approach: (1) sourcing water through reverse osmosis (RO) desalination, (2) treating the process effluent via conventional secondary treatment (e.g. activated sludge (AS)) and (3) keeping it in the loop using micro/ultra filtration (MF/UF), RO and finally ultraviolet disinfection (UV). This approach stipulates that a physical lack of water can be replaced by the cost of desalination and a closed-loop water treatment system. This makes the TCW a function of various cost inputs ranging from energy to capital costs. This paper focuses on the most volatile and significant cost components of the TCW, reflecting the impacts of changes in its drivers so as to give directional implications as to its development. It also shows that water risks can be quantified purely in financial terms. This illustrates that the TCW is a good proxy to estimate the impacts of water on industry and water risks in general. Overall, the costs considered are energy costs, capital costs, the costs of carbon as an environmental externality, labour, maintenance, and chemical costs. An analysis of global data on desalination plants (Desaldata.com 2011) yielded that there is already a tendency to replace water scarcity with the cost of desalination. Interviews with industry representatives at the TUAS Desalination plant in Singapore revealed that desalination is in fact APB's main source of water (Siew 2011), thus, validating the viability and real world applicability of the three-step approach used to calculate the TCW.

WATER-RISKS AND INDUSTRY

Temporary water shortages, even mere constraints, can already disrupt production and cause substantial losses, growth constraints, and eventually bankruptcy. Should the availability of water or its sourcing and treatment costs be subject to sudden supply and price shocks not accounting for such developments may pose a significant business risk. Thus, said thorough understanding of what drives water costs under all conditions is not only required to effectively manage water risks on an industrial level, but also to manage water efficiently and effectively as an industrial resource and commodity. Industry should also be concerned with the underlying carbon footprint of its water consumption, as well as the socio-economic impacts thereof. Water consumption and subsequent wastewater treatment volumes, therefore, need to be understood in terms of their true costs which should also include a monetized greenhouse gas carbon equivalent value per m^3 of water used. Then it is only a matter of time before a closed loop water management system becomes the next inevitable step water-dependent industries will have to take. The sludge from secondary treatment not only poses a considerable cost factor but is also significant in terms of its carbon footprint and environmental impacts. Hence, anaerobic digestion and thermal hydrolysis-based energy recovery was also considered when assessing the overall energy and carbon footprints and the TCW.

VBM

APB Singapore was analysed in terms of its theoretical water consumption, wastewater strength, and carbon emissions. The resulting costs were then imposed onto its cost and profit structures to reveal the impacts of water on its operations. The date used was taken directly from (APB's 2010) Annual Report (APB 2010). The analysis was conducted using VBM. VBM provides a management approach that takes into account all relevant factors industries need to consider when looking to embed

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sustainability within their business. The three pillars of sustainability, i.e. economics, environment, and society, are broken down into six factors: efficiency, costs, carbon, environmental health, and socio-political/socio economic factors. At the same time these factors are directly interlinked with the industry-specific technology the respective company employs. Additionally, a fully functioning and sustainable industry needs to operate within the constraints of the underlying legal framework whilst adhering to the requirements imposed by it. VBM lets companies monitor, measure, and manage all relevant factors accordingly, whilst also providing a comprehensive driver model for sensitivity analyses and impact assessment. See Figure 1 for a generalized conceptual depiction of a VBM framework. In this paper, the VBM framework was used to calculate the TCW based on the six factors mentioned above.

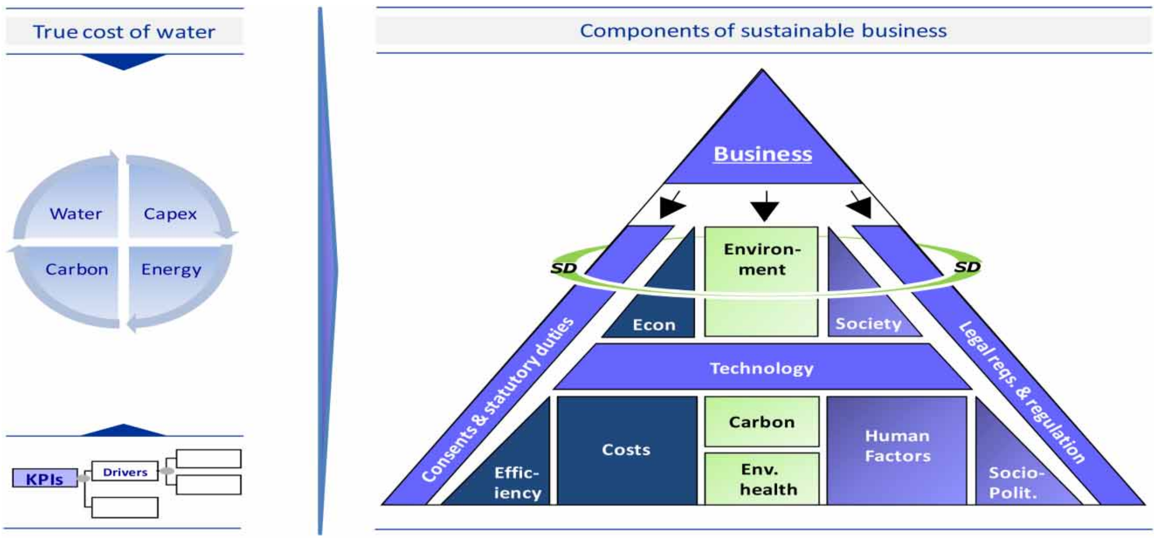

Figure 1 | SD pyramid.

Value-driver tree approach

In order to account for the various drivers and parameters involved in the derivation of the TCW, a driver-tree model interlinking all drivers with each other as well as with the defined key performance indicators (KPIs) was used. The KPIs defined were kWh/m^3 and S$/$m^3$ as well as the cost of carbon per m^3 in S$. The driver tree enabled the modelling of KPI sensitivities with respect to the driving factors behind them. The processes of RO-treatment, for example, were modelled in trees to reflect energy consumption per m^3 as a function of salinity measured in total dissolved solids (TDS) as well as capacity in m^3 to reflect the scale economies of higher capacities. (See Figure 2.) The approach was used for each of the three steps to derive the TCW.

The framework is applicable to both municipalities and industrial end users. Municipalities can estimate their KPIs based on assumptions pertaining to biochemical oxygen demand (BOD), chemical oxygen demand (COD), Nitrogen (N), Phosphorous (P) and suspended solids (SS) per population equivalent (PE) per day. This may vary according to location, e.g. while BOD is estimated to be 60 grams in most parts of Europe it is about 70 grams per PE in the USA (Tölgyessy 1993). Thus, the accuracy of the estimation of the TCW is highly dependent on the accuracy of the respective input parameters and limited to the explanatory power of average values.

As higher levels of rain and infiltration, together with a weaker BOD loading, influence the cost per unit of BOD by increasing non-BOD related removal costs such as pumping and aeration, equalization

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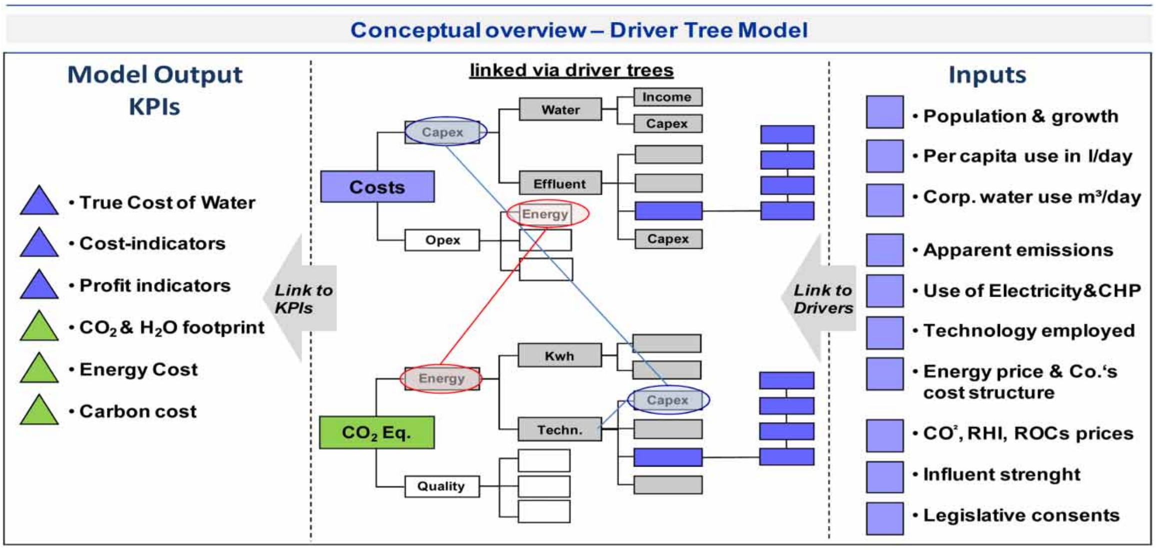

Figure 2 | High level illustration of driver-tree approach.

chambers are assumed as per the Singaporean example so as to have stable loading rate with a constant flow. This is to avoid diversion of costs to units not related to BOD removal (EPA Water Quality Office 1971). The parameters hydraulic retention time (HRT), SRT, and MLSS, were held constant and assumed to be functioning at steady state equilibrium with optimum efficiency and for a conventional AS plant. Dissolved oxygen (DO), and the food to microorganisms ratio (F:M ratio) were assumed sufficiently high at levels above their critical values to allow for efficiently working processes that are cost effective; i.e. a DO concentration of at least 2.0 mg/l (Davies 2005) with an F:M ratio of 0.2–0.4 kg BOD/kg MLVSS (Bitton 1998). Potential model extensions could allow for the reflection of a variable HRT, SRT, and MLSS, and how they would change in accordance with variations in the variable parameters flow (Q), organic loading (BOD, COD) as well as nutrients (N and P) so as to uphold the ideal equilibrium. This would allow for a higher complexity and accuracy of the model.

The pumping costs of water represent a significant, non-negligible, fraction of the costs which, however, vary significantly depending on type of pumps and the volume pumped (U.S. department of energy 1999), as well as pipe material (Rawlings & Sykulski 1999) and diameter (Doyle & Parsons 2002), and the distances and slopes required to pump. Moreover, the infrastructure costs (i.e. building a distribution network) also constitute a considerable fraction of costs which, however, were not modelled in this approach. The aim at this stage was simply to identify the major cost constituents driving the cost of water and model these based on steady state equilibrium of optimally functioning processes. However, every single step required in the process was modelled using data from literature and industry to derive a unique function yielding a cost figure per m^3 of water treated. (I.e. oxygen and energy requirements for BOD and nitrogen removal, return activated sludge (RAS) pumping, phosphorous and COD removal, UASB pre-treatment for high strength industrial effluent, capital costs, sludge treatment and energy recovery, and process emissions and carbon equivalents.) For a detailed derivation of the functions and models please refer to Leusder (2011).

Modeling energy, carbon and capital costs

Desalination

An equation that best reflects potential energy consumption per m^3 of RO permeate depends on the factors capacity and feed water salinity TDS in parts per million (ppm). It was estimated using an ordinary least squares (OLS) regression. Prior to performing this, the data were first analysed by

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creating scatter plots and determining the best-fit equation of salinity and capacity to kwh/m³ separately. Figure 3: linear, exponential and power uni-variate regression below shows potential relationships between higher salinity and power consumption as well as capacity in kwh/m³.

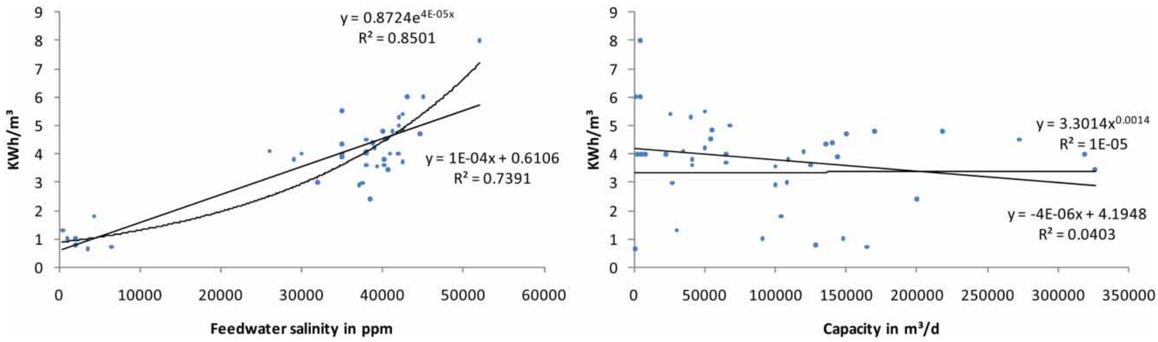

Figure 3 | Linear, exponential and power uni-variate regression.

These plots were used to assess what functional form may provide a sensible fit for the independent variables to best reflect the data, in order to then apply it in a multivariate regression analysis.

Based on the output of the regressions above, two multivariate regressions were compared. In the first model, the natural logarithm of power consumption (y), which reflects the exponential relationship found above, was used along with a linear form for salinity (x_1) and capacity (x_2). In the second model, a linear form for power consumption (y) salinity (x_1) and capacity (x_2) was used. The regressions were modelled as follows to establish as to whether the linear log-model or the plain linear model yield the better fit between power consumption, salinity and capacity:

1. Log linear model:

$$\text{Power consumption in } KWh/m^3 \ \ln(y) = \text{ß}_0 + \text{ß}_1(x_1) - \text{ß}_2(x_2)$$

2. Linear model:

$$\text{Power consumption in } KWh/m^3(y) = \text{ß}_0 + \text{ß}_1(x_1) - \text{ß}_2(x_2)$$

where x_1 is salinity in ppm and x_2 is capacity in m³ per day.

The *linear model* was found to exhibit a meaningful correlation between capacity and energy consumption, yielding that both salinity and plant capacity affect power consumption. An increase in salinity is reflected in an increase in power consumption ($t = 12.32$, d. f. $= 48$, $p < 0.01$) as is also confirmed by the findings of the Committee on Advancing Desalination Technology (2008). Plant capacity, however, was found to have the opposite effect with large plants showing slight economies of scale ($t = 2.28$, d. f. $= 48$, $p < 0.05$). The *log linear model* also yielded that there is a strong relationship between salinity and power consumption ($t = 16.80$, d. f. $= 48$, $p < 0.05$) and confirms the exponential relationship of salinity and energy as a result of increasing osmotic pressure throughout the desalination process.

As the removal of fresh water increases, feed water concentration goes up resulting in more energy required to remove the salt as a function of the recovery ratio, as indicated above in Figure 4 (ADU-RES 2006). Experience from Singapore (Siew 2011) and literature reviews have shown temperatures around 25 °C to be ideal for high recovery ratios with good permeate quality (Nisan *et al.* 2005). See Figure 5 below for a depiction of functions pertaining to temperature and permeate recovery.

Moreover, while in this model the relationship between capacity and energy consumption is still measurable, it is not significant ($t = -0.97$, d. f. $= 48$, $p > 0.10$), therein, reflecting the findings of

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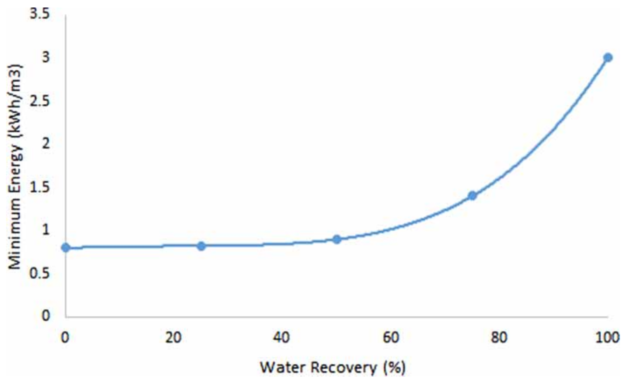

Figure 4 | Theoretical minimum energy required to desalinate seawater at 25 °C.

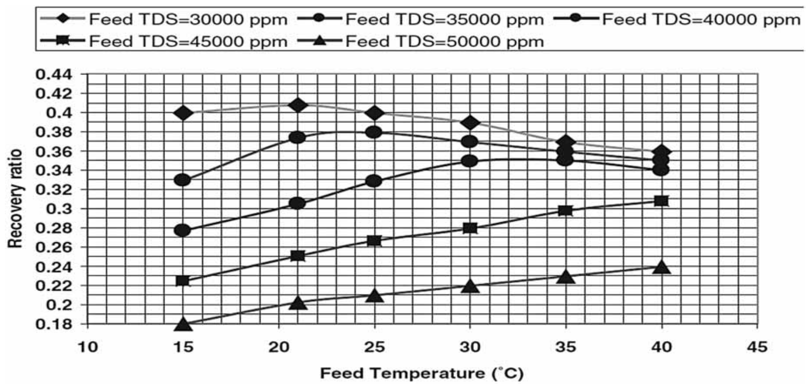

Figure 5 | Permeate recovery as a function of temperature (Nisan *et al.* 2005).

the capacity sensitivity analysis as conducted by the Committee on Advancing Desalination Technology (2008). Given that the underlying analysis is based on a limited set of 50 data points, caution must still be exercised as to the extent of the validity of an exponential over a linear relationship. Yet, as the model fits the data best, it can used to provide exemplary outputs for where there is a lack of data. See Figure 6 below for a statistical summary.

VARIABLES	Model
	Coefficients
Salinity	0.0000398***
	(2.37e-06)
Capacity	-4.09e-07
	(4.21e-07)
Intercept	-0.0704053
	(0.0948916)
Observations	50
R-squared	0.8601
Adj. R-squared	0.8542
D.f.	48
T-Stat salinity	16.80
T-Stat capacity	-0.97
T-Stat intercept	-0.74

Standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.10
$\ln(y) = ß_0 + ß_1(x_1) - ß_2(x_2)$

VARIABLES	Model
Salinity	0.0000951***
	(7.72e-06)
Capacity	-3.13e-06**
	(1.37e-06)
Intercept	0.985148***
	(0.3094126)
Observations	50
R-squared	0.7771
Adj. R-squared	0.7677
D.f.	48
T-Stat salinity	12.32
T-Stat capacity	2.28
T-Stat constant	3.18

Standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.10
$(y) = ß_0 + ß_1(x_1) - ß_2(x_2)$

Figure 6 | Log linear (left) and linear (right) multivariate regression model output.

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Based on the data from industry and the literature cited above, the formula to estimate energy consumption for desalination and reclamation in kwh/m³ was derived as:

$$\ln(y) = -0.0704053 + 0.0000398 * x_1 - 0.000000409 * x_2$$

This implies that an increase in x_1 by one unit means kwh/m³(y) goes up by 0.0000398. Analogously, an increase in capacity (x_2) by one unit implies a decrease in kwh/m³(y) by 0.000000409 with an R^2 of 0.86. Given that there are differing views in literature and practice as to the effects of capacity on energy consumption, with higher capacities said to either decrease energy consumption (Hitachi, 2010) or have virtually no measurable financial effect at all on an m³ basis (Committee on Advancing Desalination Technology 2008), the approach offers a choice between the multivariate log linear model where capacity is non-significant, and a multivariate linear model where it is significant.

Eventually, a lot depends on plant configuration, i.e. a lot can be modelled depending on the type of technology employed. Industry will be reliant on what is actually available and, thus, on estimates of what it currently costs to produce freshwater from wastewater (TDS 1,000), brackish water (TDS 10,000), average seawater (35,000), and Gulf-seawater (480,000). The modelled equation also can be used to estimate energy requirements for RO-based desalination where data is confidential and not available.

The point of the OLS regression is to illustrate what the current desalination power consumption is, based on real data. While there are other tools developed by various companies and agencies in the industry to estimate desalination costs, there are limitations to each of them (Committee on Advancing Desalination Technology 2008). As many of them are mostly for design purposes they may not reflect the actual power consumption as found in practice which may in fact vary significantly (Siew 2011). This is because desalination cost data, as with all water costing data, are a function of numerous variables with components not easy to ascertain. In line with the approach taken by the Committee on Advancing Desalination Technology (2008), which deemed the transparency of actual operating data from existing desalination plants a better indication as to potential costs, this analysis is further enhanced by data from Global Water Intelligence's Desalination Database (Desaldata.com 2011) as was modelled into the above equation.

For desalination the model yielded the following cost break-down based on the above parameter settings. Energy and capital costs are the main constituents of the per m³ cost of desalinated water, together accounting for 75% (Energy 41%, capital 34%). Capital costs were modelled using a standard annuity formula and a 4% WACC with a duration of 35 years (Leusder 2011). This was analogously done for secondary treatment and reclamation. With a price of S$ 20.6 per tonne of carbon, carbon costs do not represent a significant cost factor at S$ 0.04 per m³ of water produced (5% of total costs). The remaining 20% are made up of chemical, labour, and maintenance costs and are based on literature estimates as per the Committee on Advancing Desalination Technology (2008).

Secondary treatment

As the constituents of AS were modelled in a complex approach considering oxygen and energy requirements for BOD and nitrogen removal, RAS pumping, phosphorous and COD removal, UASB pre-treatment for high strength industrial effluent, capital costs, sludge treatment and energy recovery, and process emissions and carbon equivalents. The costs are broken down into 39% sludge costs, 12%, 5% carbon costs, 13% COD removal costs, 14% labour costs, 10% phosphorous removal costs, and 3% capital costs, (Leusder 2011).

Closing the loop

Water reclamation, which merely requires a one-stage RO train with two passes, is a cheaper alternative to desalination, which is a more complicated process (Jensen 2004) requiring higher feed-pressures at

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around 60 bar, a two-stage RO process, and thus higher energy requirements at around 4 KWh/m^3 (Siew 2011). As reclamation requires only about 1 KWh/m^3 and can be used where access to seawater is limited, it is generally the preferred alternative, given that it subsequently emits less carbon (Jensen 2004). Yet, given the public's acceptance for treated effluent, its applicability for households and beverage and food production is highly limited. As with desalination the main cost constituents of reclamation are energy, capital, and carbon costs which mere modelled using the same approach as for desalination above. For regions where no data could be obtained, the aforementioned linear log model can also be applied to estimate energy requirements. The required constituents used to derive an energy price using the above equation were feed water TDS and the capacity of the plant. In the two cases analysed here, the equation was not used as all data was readily available for modelling purposes. The two cases compared were the Singaporean water reclamation approach (NEWater) involving UV disinfection and the Californian approach involving advanced oxidation processes (AOP) using hydrogen peroxide. The latter allows for the removal of non-easily removable organic compounds such as highly toxic and badly biodegradable chlorophenols (CPs) (Pera-Titus *et al.* 2004), endocrine disrupting compounds (Rosenfeldt & Linden 2004), and even pesticides (Badawy *et al.* 2006).

The modelled approach allows for the selection of UV treatment for disinfection purposes at 70 mj/cm^2 at an average energy consumption of 0.035 kwh/m^3 (PUB 2002, PUB 2011) versus AOP at 300 mj/cm^2 at 0.08 kwh/m^3 (OCWD 2011). Moreover, it breaks down the total energy costs into micro- and ultra-filtration (MF/UF) at 0.2 KWh/m^3, RO at 0.6 KWh/m^3 and a remainder of 0.165 KWh/m^3 for the air instrumentation system and the chemical dosing pumps. Whilst labour costs were derived from literature, maintenance costs were estimated as second pass SWRO permeate costs at 0.01 (Committee on Advancing Desalination Technology 2008) and 30% thereof for second pass permeate, as per PUB experience (PUB 2002, PUB 2011). Chemical costs were derived as per data by the Orange County Water District (OCWD 2011). Overall, total chemical costs for water reclamation were thus estimated at 0.0016 S$ per m^3, whilst permeate conditioning costs were assumed to be equal to desalinated water conditioning costs as per the Committee on Advancing Desalination Technology (2008). In summary, the cost-constituents of reclaimed water are 75% capital costs, 10.5% energy costs, 1.5% carbon costs and 14% other costs, incl. chemical costs, (Leusder 2011).

Wastewater biomass and energy recovery

Conventional aerobic wastewater treatment is associated with the production of sludge (UNEP, 2000). Whilst some aerobic treatment systems are now said to be sludge-free due to an aerobic-aeration process with 'supercharged' air sustaining a highly effective flock of enzymes devouring the sludge material in the system (Global Water Group 2011), these are not widely distributed globally. Therefore, when assessing the impacts of wastewater treatment, sludge, in terms of treatment costs and disposal emissions but also in terms of energy recovery from anaerobic digestion and pre-treatment, must be taken into consideration.

Sludge treatment is an expensive and carbon intensive process. For a granular analysis of the process emissions and carbon equivalents associated with sludge please also refer to Leusder (2011). While sludge treatment costs vary between different regions, studies show that as a rule of thumb the overall costs (manpower, equipment and energy) can account for about 50% of the total cost of wastewater treatment (Kemira 2008). Studies by the EC confirm this. The most significant cost constituent is sludge treatment, which is expected to increase further as more stringent hygiene standards are introduced. Sludge dewatering and drying is also very costly, yet savings from the transport of wet sludge off-set these to a certain extent (European Commission Environment 1999).

In the following section, costs, emissions, and energy recovery are discussed exemplarily based on a set of underlying assumptions. The employed pre-treatment assumed is the Norwegian thermal hydrolysis system CAMBI®. As per the below process data by CAMBI and Black & Veatch, an

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electrical output of approximately 1MWh per tonne of dry solids (TDS) after an electrical input of 310 kwh can be achieved (Jolly & Gillard 2009) for thermal hydrolysis pre-treatment plants sized as small as 3,600 metric tonnes per year (CAMBI 2007). This is based on 400 m^3 biogas per TDS with a methane yield of 68% at 61% overall volatile solids (VS) destruction and 48% TS removal (McCausland & McGrath 2010). The mixing ratio of primary vs. waste activated sludge (WAS) is around 50/50 on a dry solids basis (Jolly & Gillard 2009) with a tendency towards a higher WAS content (Rognlien 2011). The sludge is fed at about 10–12% DS assuming 90% effective digester volume (EDV) and average VS of 75% with the VS loading being about 6.5 kg VS per m^3 digester capacity per day with a HRT of just over 12 days (Lowe *et al.* 2007).

In order to approximate the amounts of sludge contributed to municipal sewage treatment works by APB in Singapore, a simplified regression model for primary and secondary sludge based on municipal AS data from Ontario and Kuwait was used. The quantity and characteristics of sludge produced depend on the wastewater characteristics and the degree of treatment. The two main sources of sludge generated in AS plants are SS in the influent wastewater (raw primary sludge) and BOD removed during the AS process (oxidized secondary sludge). The capacities of the plants analysed ranged from 500 to 200,000 m^3/day. The wastewater treated is municipal and primarily domestic with treatment encompassing primary and secondary (conventional AS) treatment stages (Hamoda 1988). Using the following equation with an R^2 of 0.88, an estimate for annual sludge production in tonnes based on total SS and BOD removed yielded:

$$\log Y = 0.970\ \log X - 0.005$$

Based on the average brewery wastewater characteristics of 1,500 mg/l of BOD and SS of 60 mg/l (Worldbank, 1998) the total primary and secondary sludge volumes estimated for APB totalled 176.1 and 368.8 tonnes per annum, respectively, yielding a total annual sludge tonnage of approximately 544.9 tonnes. This is based on 70% of SS and 30% of BOD being removed during primary sedimentation (FAO 1987) at an overall treatment efficiency of 95%. As the ratio of primary vs. secondary brewery sludge is 33/67 on a dry solids basis, the accuracy of the potential electrical energy output as per Cambi's thermal hydrolysis pre-treatment is uncertain. Given that experimental results showed that municipal and brewery sludge mixtures at a ratio of 25:75% by weight (sewage:brewery) yielded higher biogas production (Babel *et al.* 2009) comparable outcomes in terms of net energy production may still be possible, though, this requires further investigation. For simplicity sake it is assumed that the energy yield is the same so as to give a rough ballpark figure by which to illustrate potential carbon and energy savings. The total energy yield that could be achieved from such an amount of sludge was then calculated at 490.4 MWh per annum with a carbon equivalent of approximately 267 tonnes of carbon. This was assuming that 1 KWh of grid electricity has 545 grams of carbon equivalent (Carbon Trust 2011) while energy generated via combined heat and power (CHP) has no carbon footprint. Alternatively, when accounting for carbon, by assuming that energy generated via (CHP) has a carbon equivalent of 0.295 kg per KWh (WRc 2007) this still saves 123 tonnes of carbon emissions. Overall, up to 89% of the energy used to aerate APB's wastewater and 56% of total AS energy cost can be saved using thermal hydrolysis, as per Cambi, if all of APB's sludge is pre-treated as per the above mentioned methodology. This reduces the aeration footprint to 0.147 KWh/m^3 from 1.315 KWh/m^3 previously and cuts total energy cost for AS from S$ 76,584 to S$ 33,464. In other words, sludge can save 1.168 KWh/m^3.

Water derivatives

The results of the three-step approach yielded that the two main components of the TCW are energy and capital costs. Given that the TCW was used to assess the impacts of water on APB, it makes financial

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sense to address potential fluctuations in the price of water by locking it in at a certain point in time. By replicating the pay-off of an underlying without having to physically hold it, derivatives provide such a lock-in mechanism. As water derivatives do not exist at present, the approach employed in this paper replicates their hypothetical pay-off by pricing an option based on the main constituents of the TCW; i.e. energy, carbon and capital costs. As energy, carbon, and interest rate derivatives (as a capital cost proxy) do exist, the volatilities of their underlyings can now be used to replicate the pay-offs of a hypothetical water option. The example uses a European option as a fixed maturity instrument. An option is used as it is the most costly yet also most flexible means of managing risk in the short run. It also allows for the evaluation of delta, gamma and vega risks, which can reflect the risk companies face with respect to water as they show how the price of water and water risk management changes relative to movements in the price of the underlying. In other words, they provide somewhat of an embedded market sensitivity analysis of water's price based on its main pricing constituents. The option price itself is calculated using the Black-Scholes option pricing model (Black & Scholes 1973).

With energy accounting for ~50% of the TCW, the volatility thereof can be assumed to be a good proxy for the volatility of the water price itself. Carbon is yet another source of volatility in the water price and can still be subject to massive legislature induced price hikes. Thus, the volatility of natural gas at 0.375 (as an exemplary proxy for electricity) and the volatility of carbon at 0.355 (Bloomberg 2011) were used as a weighted average to estimate a proxy for the volatility of water. The weighting applied was 97/3 energy/carbon given the relative contributions of the respective components to the TCW. Increasing carbon prices will, eventually, skew this weighting towards carbon. By using the weighted average cost of capital (WACC) of the company as the risk-free rate for the determination of the option price, the volatility of the cost of capital is also reflected in the option price. An overview of the used input parameters to determine an option price is depicted in Table 1.

Table 1 | Option input parameters

Option pricing parameters		
Underlying	S0	1722376
Strike Price/Exercise Price	X	1722376
Risk Free Rate	r	0.04
Time to Expiry in Years	T	1
Volatility	σ	0.374418

The option is priced with a duration of 1 year and a WACC of 4% to represent the risk free rate. The volatility used is the implied volatility of natural gas as a source of energy (Bloomberg 2011). The underlying price and the strike price were assumed equal. Other costs were assumed constant over time. Based on this input, the price of a call option is S$ 285,944 and the corresponding put option is S$ 218,409.

The *delta* of an option is the first partial derivative of the call option price with respect to the price of the underlying, i.e. water or in this case its proxies, and measures the rate of change of the option price with respect to the underlying while holding all else constant (Hull 2008). So if the price of water or the components constituting its price change by one dollar, the price of the option changes by the value of delta. Along these lines, the gamma of an option is defined as the rate of change of the option's delta with respect to the price of water or the components constituting its price. Mathematically speaking, it is the second partial derivative of the option value with respect to the underling (Hull 2008). Here, a percentage change in the price of water changes the option delta by the value of gamma. The vega of an option is defined as the rate of change of the option price with respect to the volatility of the underlying and is calculated as the first partial derivative of the option price

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with respect to the volatility of the underlying asset (Hull 2008). In this case, vega has been adjusted to reflect a percentage change. That is, a 1% increase in volatility increases the option price by S$ 5,389. As such, the 'Greeks' can be used to reflect the changing costs of managing water risks through derivatives, thereby, offering a different perspective on water risks and their management. While there is a vast array of 'Greeks' for the risk management of derivatives, this paper merely illustrates the three mentioned above. For an overview of the put and call 'Greeks' see Table 2 below.

Table 2 | Option Greeks

Call Option Risks		Put Option Risks	
Delta	−0.615637	Delta	0.384363
Gamma	−0.000001	Gamma	−0.00001
Vega	−6,580.6	Vega	−6,580.6

RESULTS AND DISCUSSION

This paper illustrated the dependence of the price of water on the parameters: energy, capital, and carbon. The underlying analysis was based on an energy price of S$ 0.1 and a carbon price of S$ 20.6 (base case). Energy prices are often subsidized and generally subject to high fluctuation given the finite nature of oil, coal, and gas. Moreover, with carbon prices required to be at least € 35 per tonne (Crooks *et al.* 2010), water is subject to yet another volatile cost component. Therefore, in order to make the model more robust, a sensitivity analysis to reflect these changes was conducted holding constant labour, maintenance, and chemical costs. These costs are hence referred to as other costs. The below figure illustrates the impacts of a 100% increase of the energy costs holding constant carbon and capital costs vis-á-vis the base case. The impacts of energy recovery from sludge were not taken into consideration in this analysis.

Figure 7 shows an increase in relative energy costs from 14% to 25% for water reclamation, 13% to 22% for AS, and finally an increase from 41 to 58% for desalination, the most energy intensive of the

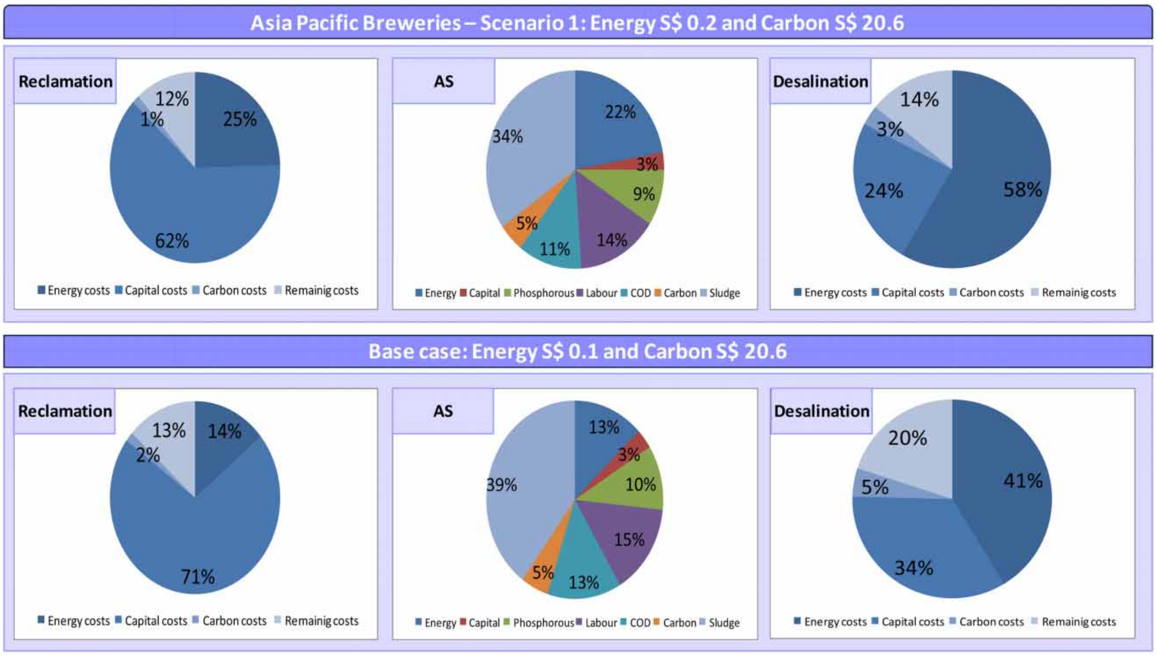

Figure 7 | Energy sensitivity analysis Scenario 1 vis-á-vis the base case.

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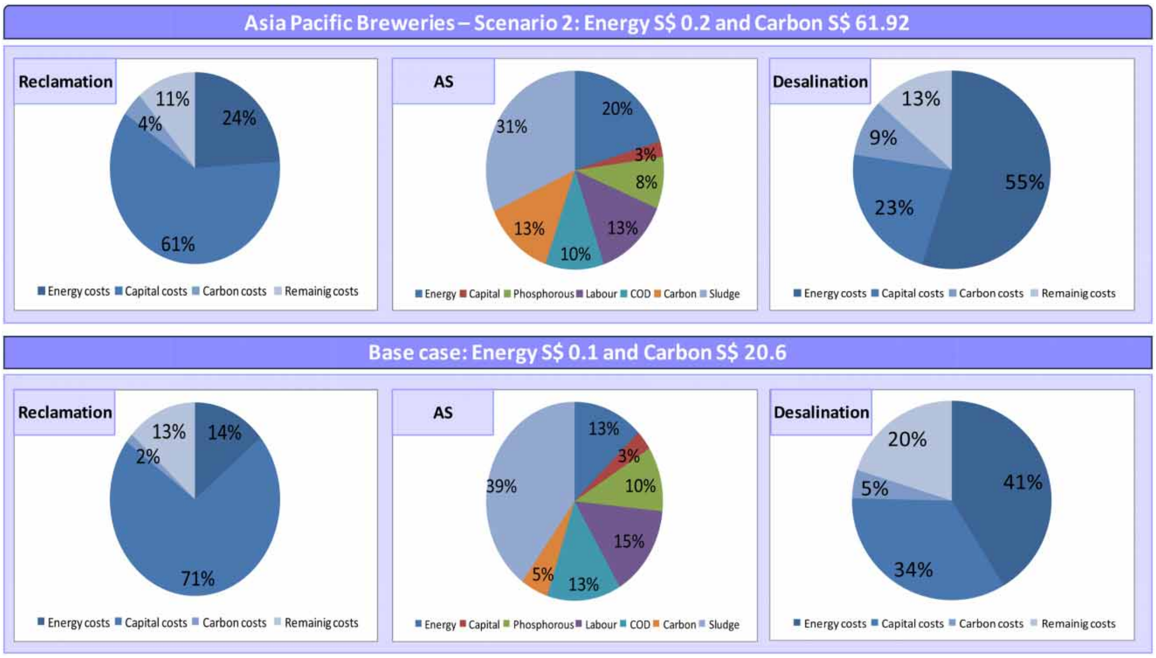
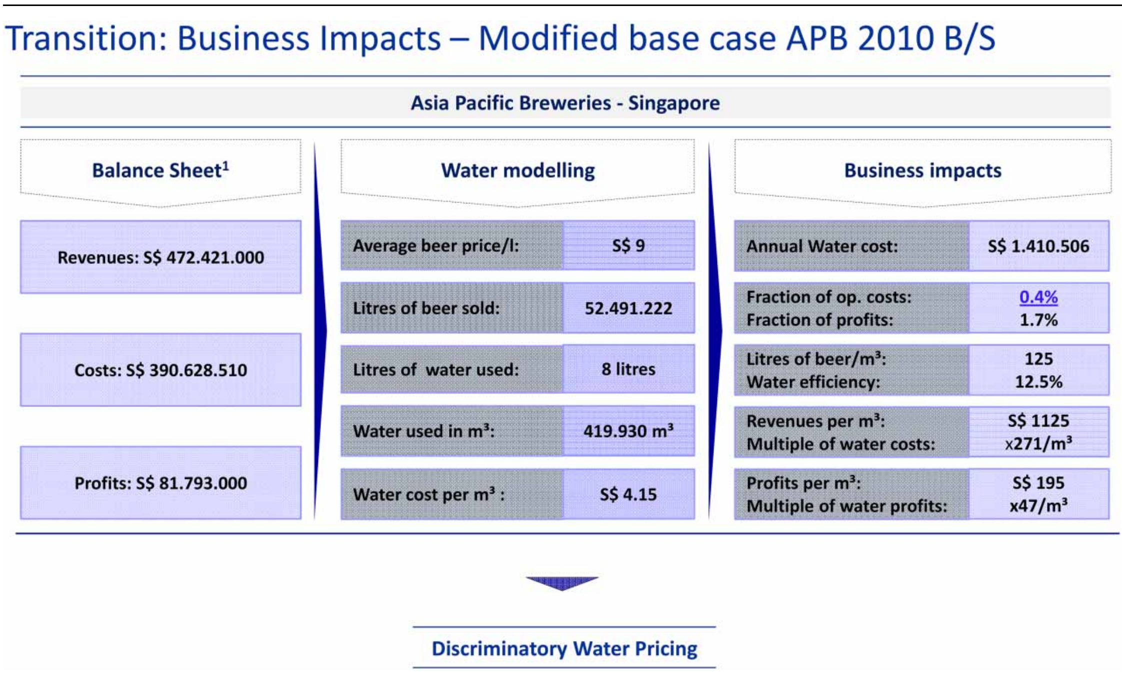


Figure 8 | Carbon sensitivity analysis Scenario 2 vis-á-vis the base case.

three options. Conducting the same analysis for a carbon price of S$ 61.92 at an energy price of S$ 0.1 yielded that the relative costs of carbon for water reclamation increased to 4% from formerly 2%. The relative cost of carbon for AS, the most carbon-intense process, more than doubled to 13% from a mere 5%, while desalination increased to 9%. (See Figure 8.)

Table 3 | Impacts of the TCW on APB



[1]Publicly available data from annual report.

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The conducted analysis yielded that even with substantial increases in the relevant parameters of the TCW, the resulting costs remained negligibly low for breweries. This implies that there is room for discriminatory pricing.

Based on a carbon price of S$ 20.64 and energy selling at S$ 0.1 per KWh, the results yielded a TCW of S$ 4.15 per m^3. In terms of APB's total operating costs this merely accounts for 0.4%. However, $1m^3$ of water drives S$ 1,125 in revenues, i.e. it yields 125 litres of beer at S$9 per litre of beer with profits of S$ 195 per m^3. (See Table 3).

Scenario 2 increased both energy and carbon prices by 100% and 300% respectively. The changes were then reflected using the previous driver-tree approach. The results increased the TCW to S$ 5.13. This increased water's fraction of total costs to 0.5% from previously 0.4% whilst augmenting the fraction of profits to 2.6% from previously 2.1%. The profit and revenue multiples (how much profit and revenue is driven by $1m^3$ relative to its price) fell to 37 and 219 from 46 and 273, respectively.

Water derivatives offer a means of using the price of water as a parameter to determine the cost of managing water risks. Moreover, they also offer a set of additional risk metrics; the Greeks. These underline how managing water risk essentially boils down to managing energy, carbon and interest risk as well, as they reflect how the price of the option changes as a result of small changes in the price of the underlying which again is approximated by its major constituents and changes thereof, i.e. carbon, energy and interest rates.

CONCLUSION

This paper illustrated that water risks can be quantified purely in financial terms, thus, illustrating that the TCW is a good proxy to estimate industrial water risks and their impacts. It was derived using a complex VBM model that accounts for all drivers of the water price and is a function driven mainly by energy, carbon, and capital costs. As such, water risk management is merely a question of adequately accounting for these constituents. Derivatives offer an effective means of incorporating all three constituents in one financial product. The TCW was, thus, also used to derive a water option price as well as its delta, gamma, and vega. It was based on the weighted average implied volatilities of the carbon price and the Indonesian gas price as a proxy for electrical energy volatility in Singapore. The cost of an option at par was S$ 285.944 for calls and S$ 218.409 for puts given a maturity of 1 year and a risk free rate of 4%.Water derivatives facilitate the replication of the pay-offs of purchasing these underlying constituents. However, as water merely makes up 0.4% of total operating costs for APB shows that a derivative protecting against water-price fluctuations without actual physical delivery will only address 0.4% of the companies costs, leaving its profits to stand and fall with water's physical availability. This does not protect APB from forfeiting the entirety of its revenues, as without water, it simply cannot produce anything. As such, a consistent supply of a standardized quality effluent traded via forward contracts would offer protection against such a shortfall.

The thermal hydrolysis of sludge even cut secondary treatment energy costs by 56%. As the TCW constitutes such a low fraction of total operating costs, yet, water drives a significant amount of revenues and profits, the application of higher water tariffs to APB based on the respective value of water to it (discriminatory pricing) may lead to an efficient outcome, if demand-price elasticities of the company's products are accounted for accordingly. Based on the above analysis in Table 3 we can see that the higher the value that can be derived from one m^3 of water relative to the fraction of the TCW over the total operating costs of the firm is, the more the firm should be charged per incremental m^3. The higher this value is, the more efficient the outcome of discriminatory pricing will be. However, whilst this approach yields a financially accurate basis for water pricing, it would still skew the desired results unfavourably, if the corresponding demand price elasticity is not taken into consideration accordingly. As such, a discriminatory increment in the water price would need to go hand in

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hand with a price fixing or rather be administered through a temporary tax break for the company until it manages to implement measures to incur the same total cost of water by reducing the overall amount of water or by simply closing the loop and recycling its own effluent.

The key difference of the TCW and VBM framework approach versus conventional assessment and management approaches is that it illustrates how each individual driver affects the KPIs. A further feature is the ability to feed balance sheet data into the model yielding KPIs reflecting a company's susceptibility to water and wastewater treatment costs. Applying this approach to a number of companies in the same sector can be used to derive an index by which to assess entire industries and their relationship to water. The developed approach is not a design software but an impact assessment tool considering environmental (carbon and pollution), economic (energy and carbon costs) and societal (e.g. acceptability of reclaimed and desalinated water) impacts on companies. As all the required data is publicly available, the model can be used to provide a directional indication of potential impacts for companies without access to confidential information. However, in its present configuration it can only accommodate municipalities and food/beverage companies. Further research is required to adapt it to, e.g. semiconductor companies and other industries.

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