Water Quality in the Des Plaines River
Near the Pheasant Run Landfill in Bristol, Wisconsin

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**Abstract**

Water quality monitoring studies are necessary in order to manage sources of pollution and maintain water that is safe for humans and the environment. The Pheasant Run Landfill, located in Bristol, Wisconsin, is positioned near the Des Plaines River and its tributary Brighton Creek. The landfill is located on soil that is not suitable for landfills and poses a risk for contamination of nearby bodies of water. Measurements of dissolved oxygen, pH, conductivity and chloride ion concentrations were taken in the Des Plaines River upstream from the landfill, downstream from the landfill and upstream from the landfill in Brighton Creek to determine if the landfill is contaminating the Des Plaines River. Dissolved oxygen percent saturation was significantly lower downstream than upstream, pH and chloride concentrations were significantly higher downstream than upstream and conductivity was not significantly different. Further research could be done to establish more certainty that the differences in chemical properties are the result of the landfill polluting the river and no other sources, but this research suggests that the landfill is likely contributing to the contamination of the river.

**Introduction**

Landfills are valuable in keeping living conditions sanitary by collecting waste in a single location instead of spreading it throughout a community. However, with a high concentration of contaminants in one area, this increases the risk of polluting the region surrounding the landfill. Leachate from landfills contains pollutants that are dangerous to the health of humans as well as the environment. When landfills are near bodies of water, this increases the risk of spreading contaminants if they do escape from the landfill. It is necessary to ensure that landfills are not polluting neighboring bodies of water in order to prevent the spread of pollutants from the landfill.

The Pheasant Run Landfill is built close to the Des Plaines River on soil that poses a high risk for contamination of surrounding water. It is possible that the landfill may be polluting the neighboring river because of its proximity, particularly during rain events in which runoff from the landfill may enter the river. Dissolved oxygen, conductivity, chloride levels and pH can be used to determine the presence of leachate in the water. With contamination, it would be expected that downstream from the landfill would have higher conductivity and chloride concentrations.
concentrations due to an increase in chloride ions frequently found in leachate entering the river. It would also be expected that pH levels and dissolved oxygen would be lower downstream due to decomposition of waste reducing the pH and additional nutrients entering the river leading to eutrophication. Water quality monitoring is necessary to protect the health of the ecosystem and maintain water quality standards.

**Literature Review**

**Landfill Risks**

Landfills offer a place to dispose of waste in order to create a more sanitary environment to live in, keeping waste away from where people live. However, as human population increases, more waste is generated and with this, the risk of pollution increases. It is important to maintain close monitorization of landfills to catch problems that may arise before they cause major issues.

Air pollution, impacts on wildlife and water pollution are some of the risks that are associated with landfills. Air pollution is a major concern as gases from the decomposing waste escape into the atmosphere. Landfills can produce between 70 and 300 m$^3$ of biogas for every 1000 kilograms of municipal waste, with the type of gas varying between mostly methane or carbon dioxide based on the state of decomposition of waste in the landfill (Bialowiec 2011). Occasionally landfill operators will collect gases and use them to produce energy, which can be a benefit. In 2004, Pheasant Run Landfill collected the most gas out of all privately-owned landfills in Wisconsin with 48.1 million m$^3$ of gas (Michels and Hamblin 2006). In 2016, the Pheasant Run Landfill extracted around 42.5 million m$^3$ of gas, 53.5% of which is methane, using gas collection wells (DNR 2018, Landfill LMOP Data 2018). Methane is a strong greenhouse gas
which can contribute to climate change, but with collection systems in place, this gas can be harvested and used as a source of energy.

Another concern that arises when considering pollution from landfills is the large number of birds, such as seagulls, that live near the landfill. These birds are attracted to landfills as they offer a large source of food. This can be a concern as some of these birds carry diseases which can be passed to humans. These diseases can include *Salmonella*, *Campylobacter*, *P. shigelloides* and others, which have been found in 30% of seagull feces samples collected on beaches in Racine and Milwaukee (Kinzelman et al. 2008). This large population of seagulls could also potentially cause a problem as they may displace species native to the area.

Water pollution is another hazard that is posed by landfills due to the possibility of leachate from the landfill coming in contact with surface and groundwater. Leachate is water that comes in contact with solid waste in landfills and can be contaminated with pollutants. It is important to prevent leachate from entering groundwater and neighboring surface water to avoid posing a hazard to human health. It can spread quickly if it escapes from the landfill and the pollutants it contains are often difficult to remove completely even when treated (Masoner and Cozzarelli 2015; Noerfitriyani et al. 2018; Eggen et al. 2010). These risks can potentially cause problems if the landfill is not monitored properly.

**Water Pollution: Leachate**

Leachate can contain many dangerous pollutants that persist over time including chemicals such as PCBs, ammonia, high concentrations of chloride ions, as well as altered oxygen and pH levels. PCBs, or polychlorinated biphenyls, are a dangerous chemical that can be found in solid waste. PCBs were originally used for industrial purposes because they act as
insulation and they are not flammable. They were banned from production in 1979 due to health problems that they cause and their slow decomposition rate (EPA 2018). Building materials, such as caulk, can break-down over time and release the PCBs they contain into the environment and when landfills accept these materials, the leachate is more likely to have a higher concentration of the PCBs (Herrick et al. 2007). PCBs are prone to bioaccumulation and can pose a problem in the fish that are eaten in contaminated water bodies. With the reduction of PCB production, levels of PCBs decrease in species of fish that humans eat, but there are still conditions that can lead to PCB levels to increase from year to year in particular fish species (Stow et al. 1995). This shows that even after PCBs are no longer being produced, they can still pose a threat to the environment and human health because of their longevity. Because the Pheasant Run landfill is approved to accept PCB contaminated solid wastes that are not considered bulk wastes by the Toxic Control Act, these chemicals are an example of a threat to the ecosystem that could persist for a long time, even after the closure of the landfill (DNR 2019).

Leachate can have a negative effect on water quality in rivers. In some cases, leachate is treated and released into rivers and if it is not treated well, this can lead to the contamination of the river. Treated leachate was released from the Cipayung Landfill in Depok City, Indonesia in the neighboring Pesanggrahan River (Noerfitriyani et al. 2002). This led to an increase in biochemical oxygen demand, total nitrogen and chemical oxygen demand, all higher than the standard limit and dissolved oxygen levels were lower than normal. This is a problem because this can lead to the formation of dead-zones, where many fish die, as there would not be enough oxygen in the water for them to survive. Similarly, it has been found that ammonia
levels increase and oxygen levels decrease in surface water that is contaminated, which can have a damaging effect on plants and animals in the body of water (Kjeldsen et al. 2002). These chemical impacts can have devastating effects on ecosystems near a landfill.

Leachate has been found to have high concentrations of chloride and high electrical conductivity. Electrical conductivity levels in leachate from landfills ranged from 4817 μS/cm to 6770 μS/cm and chloride levels ranged from 892 to 2090 mg/L in the Dal Skog landfill and the Esval landfill, respectively (Haarstad and Mæhlum 2007). The chloride levels and electrical conductivity were linearly correlated when measured in leachate. On average, leachate is found to have a chloride concentration of 2120 mg/L and has a conductivity range of 2500 to 35000 μS/cm (Kjeldsen et al. 2002). When monitoring the leachate plume from the Norman landfill in groundwater, leachate concentrations varied from over 600 mg/L to less than 100 mg/L with greater concentrations closer to the landfill (Masoner and Cozzarelli 2015). Leachate that had leaked from the Norman Landfill was found to have Cl- concentrations vary with precipitation. Electrical conductivity increases as there are more ions in a solution. As a result, as chloride ions are added to a solution, the electrical conductivity will often increase. Chloride can damage organisms by increasing their mortality rate and slowing down their reproduction if levels are too high (Fontenot and Lee 2013). Over time, if concentrations of chloride are very high, this can also be damaging to human health.

Leachate can affect the pH in water based on the contents of the landfill. The pH of leachate normally ranges from 4.5 to 9 depending on the composition (Kjeldsen et al. 2002). In some landfills, the contents lead the pH to be more basic, for example, in the Cipayung Landfill in Indonesia, the pH in the leachate was around 8 (Noerfitriyani et al. 2002). In the Dal Skog
landfill, the pH was found to be 6.2 for the leachate, while in the Esval landfill, the pH was found to be 7.1 (Haarstad and Maehlum 2007). Leachate can vary in pH based on the contents of the landfill and the stage of decomposition. Ash in landfills is often found to lead to an increase in pH in leachate (Water Encyclopedia 2019). In 2017, the Pheasant Run Landfill did not collect any ash (Wisconsin Municipal and Industrial Waste Landfill Tonnage Report 2017).

The age of the leachate affects the pH as leachate older than 5 years tends to have a pH of greater than 7.5 as the pH stabilizes and methane production increases (Abbas et al. 2009; Kjeldsen et al. 2010). The Pheasant Run Landfill was established in 1963 but continues to accept solid waste which would likely lead to newer leachate production and pH values in the acidic range.

Monitoring groundwater for leachate contamination is important because leachate plumes can spread quickly in ground water. This is a problem because it can lead to the contamination of drinking water such as private wells as well as surface water. Near the Norman landfill in central Oklahoma, the leachate plume from the landfill, which was unlined, increased in size in 24 years by 878% in soil deposited by the Canadian River which was an expansion of 182,600 m² (Masoner and Cozzarelli 2015). On October 9th, 1994, the U.S. Government established regulations that must be followed in the management of landfills such as the groundwater and private well monitorization to prevent leachate plumes by catching leaks early near the landfill (Code of Federal Regulation 2019).

Depending on different conditions, leachate concentration varies with distance from the landfill. In the case that the leachate comes in contact with neighboring bodies of water, precipitation can affect the rate at which the leachate spreads. During dry periods, there tends
to be more pollutants near the landfill and less downgradient while during wet periods it is the opposite (Masoner and Cozzarelli 2015). This is important when monitoring the spread of leachate as it changes depending on dilution and ability to expand to locations further from the landfill.

**Leachate Treatment**

There are many different methods to treat leachate. In the past, the “dilute and disperse” method was used in landfills in order to treat leachate. This consisted of unlined landfills which allowed the leachate to escape the landfill and have a lower concentration near the landfill as it would disperse into the environment (Bialowiec 2011; Masoner and Cozzarelli 2015). This method allowed for the leachate to be diluted, however, the contaminants in the leachate would enter the groundwater and could cause health problems. In addition, as time goes on, the waste in the landfills continues to break down and more pollutants are released into the environment so these landfills pose a long-term problem.

Newer methods used at different landfills to partially treat leachate include using plants like cattails and reeds to bioremediate the leachate. These plants are able to absorb heavy metals from the leachate into their root system without significantly damaging their ability to grow, as was supported in a study in which the cattails and reeds were grown in a pond where leachate was stored (Grisey et al. 2011). These plants can be used to gather pollutants which could be damaging to human health and prevent them from spreading throughout the area where the pollutants are found. There are also wetlands in the proximity of the Pheasant Run Landfill so the plants growing in these areas may be able to protect the Des Plaines river from carrying pollutants downstream in the event that there is contamination.
Other methods of treating leachate include aerobic treatment in which oxygen is added to the leachate on site to increase decomposition rates. It has been found that aerobic treatment of leachate reduces the electrical conductivity and chloride levels in the leachate (Haarstad and Mæhlum 2007). These methods of treatment offer a few ways of reducing the impact of the landfill on the environment.

Soil Contamination

Pollutant retention in soil near and in landfills is essential to consider when looking at water quality because this can determine how quickly leachate can spread. Leachate in soil varies in concentration with distance from the landfill. It has been determined that contaminants from leachate are highly concentrated in soils near the landfill and with distance, the concentration gradually decreases (Liu et al. 2013). This trend is similar to the trend of decreasing concentration of leachate found in groundwater further from a landfill (Masoner and Cozzarelli 2015). Landfills cannot indefinitely contain contaminants and depending on the construction of landfills, this can affect the rate at which contaminants escape. It has been determined that thicker soil and less heavy metals in landfills have the greatest amount of retention of the heavy metals (Yobouet et al. 2016). Soil type also has an effect on retention. In areas with sandy or gravelly soil, contaminants can wash out of the soil quickly and contaminate the surrounding areas while silty or clay soils do not allow the contaminants to escape as quickly, but they can also store larger quantities for a longer period of time (Yobouet et al. 2016).
Case Study: Pheasant Run Landfill

The Pheasant Run Landfill is a municipal waste facility owned by the company Waste Management of Wisconsin, Inc. This landfill was opened in 1967 and is expected to close in 2094. It is designed to hold 35,503,636,416 kg of solid waste and by 2016, 19,848,474,549 kg of waste (55.9% of capacity) had been disposed of in the landfill (EPA 2018). The landfill accepts different types of non-hazardous solid waste including friable and non-friable asbestos, biosolids, construction and demolition debris, solids from drum management, industrial and special wastes, municipal waste, naturally occurring radioactive material, waste that requires bioremediation and yard waste (WM Solutions 2019). This landfill is located 85 meters from the Des Plaines River and 33 meters from Brighton Creek, the river’s tributary.

Near the Pheasant Run Landfill, the majority of the soil is silty loam or silty clay loam (Web Soil Survey 2019). This soil type does not drain as fast as sand and it drains more quickly than clay. The soil in the area is mostly the result of glacial deposit so there are a variety of mixtures of soil types such as clay, silt, sand and some gravel (U.S. Army Corps of Engineers 2013). The soil may allow for filtration of leachate if it enters the river as the soil drains but may also allow for contaminants to bond with clay particles.

According to the Web Soil Survey, the area around the landfill is not ideal for the location of a sanitary landfill due to the tendency to leach. Much of the soil is rated as very limited or somewhat limited when considering its applicability towards use for the location of a landfill based on risk of pollution (Figure 1). The poor rating is due mostly to the risk of flooding, ponding of water, a shallow depth to the saturated zone, and high amounts of seepage (Web Soil Survey 2019). The depth of the water table is found to range from 0 cm deep from the
surface near the base of the landfill and over 200 cm at the top of the landfill. Pooling also occurs frequently around the base of the two parts of the landfill and pools rarely on parts of the north east part of the landfill (Web Soil Survey 2019). Along the Des Plaines River, there is a relatively high risk of groundwater contamination due to medium permeability of soil, low depth to water table and there are sand and gravel deposits (Schmidt 2007). When the landfill had been first been built, restrictions on locations of where landfills could be built were less strict and it was not until 1984 in which stronger federal regulations were put in place.

Much of the area surrounding the landfill is in the flood plain and there are also wetlands surrounding the area (Figure 2). In Title 40, Subpart B section 258 of the Code of Federal Regulations (2019), location restrictions for landfills are established for floodplains and wetlands. These include not restricting the flow of the river during the 100-year flood, not reducing water storage capacity and not being at risk for the river causing solid waste to escape from the landfill by erosion. The landfill must also not severely damage wetlands. The soil in this area is often disrupted by water and this makes the area less ideal for the placement of a landfill.
Figure 1: Soil Suitability for a Sanitary Landfill (WSS 2019)
Wetlands and Floodplain near the Pheasant Run Landfill

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Figure 2: Wetlands and Floodplain near the Pheasant Run Landfill
In 2008, Waste Management and the Town of Paris came to an agreement that the Pheasant Run landfill could be expanded. Initially Waste Management sought to expand the landfill 17 million cubic meters but then reduced the request to 14.1 million cubic meters (Pheasant Run Landfill Western Expansion Negotiated Agreement 2008). Even with this reduction, this would have put the landfill closer to private wells than was allowed by regulation. In the end, the State of Wisconsin only allowed the landfill to expand 6.9 million cubic meters. In the agreement, the landfill was required to follow the regulations established in order to safely run a sanitary landfill.

**Des Plaines River**

In the past, the Pheasant Run Landfill has had some instances in which waste spilled or leaked. On July 19th, 1991, the landfill had a leaking underground storage tank which contained gasoline, diesel and waste oil, which was resolved by May 13th, 1992 (DNR). On December 22nd, 2006, the landfill reported a spill of 3000 liters of leachate into the compost retention basin due to soil erosion. The spill event was closed on April 24th, 2007 and the wall that had eroded was built back up (DNR). There have been a few events in which the landfill has had problems which has led to the contamination. With these risks, it is important to monitor the landfill’s effect on the Des Plaines River.

Recently, there have been problems in the Des Plaines River in Illinois with pollution. High levels of PCBs, mercury, phosphorus and arsenic have been reported in news articles as well as low dissolved oxygen levels in areas with a large number of factories and production of industrial waste (Mallory 2017; Abderholden 2016). Parts of the river are considered impaired in Illinois and are being monitored to determine chemical contamination in the water,
sediment, fish and macroinvertebrates (DRWW 2017). There are also fish found in the Des Plaines River in Cook, Lake and Will Counties in Illinois which should not be consumed at all or should be consumed once a month or less including the common carp, channel catfish and freshwater drum due to high concentrations of PCBs (IDPH 2019). This area is a considerable distance from the Pheasant Run Landfill and is near areas where there is a high amount of industrial waste so the high concentrations in this area are not likely caused directly by pollution from the landfill, but it is possible that the landfill could contribute to some of the contamination in the river. The river has also been found to have high nutrient loads in Wisconsin, most likely due to 63.3% of its watershed being agricultural (Water Quality Management Plan 2011). The river has had a history of pollution throughout much of the watershed.

**Objectives and Hypotheses**

Because of the proximity of the landfill to the river and the poor quality of the soil it is located on, it is possible that the landfill is contaminating the neighboring body of water. This study will determine the likelihood that the landfill is contaminating the river by measuring various chemical properties in the river. Specifically, if the landfill were polluting the river, it would be expected that dissolved oxygen would be lower downstream than upstream due to additional nutrients entering the river from the landfill, leading to an algal bloom and die-off and a decrease in dissolved oxygen due to the consumption of the dead algae and oxygen by bacteria. The pH is also expected to be lower downstream because younger leachate has an acidic pH and the landfill is still accepting waste so the leachate may have a lower pH. The chloride concentration would be expected to be higher downstream because leachate contains
a large amount of chloride ions and this would additionally lead to a higher conductivity downstream due to an increase in ions.

**Methods**

**Study Area**

This study takes place in the Des Plaines River near the Pheasant Run Landfill in Bristol, Wisconsin. The headwaters of the Des Plaines River is in Racine and it flows south and merges with the Kankakee River to form the Illinois River near Joliet, Illinois (Des Plaines River Water Trails 2015). The entire watershed is 3768 km² in Illinois and Wisconsin (Stormwater Management Commission). This river has a watershed that covers 344 km² in Wisconsin with around 16.75% of the watershed being urban, 63.25% being various types of agriculture and 20% being surface water, wetlands, woodlands and open rural lands (Figure 3) (2011 Water Quality Management Plan Update 2011). There are more than 582 kilometers of streams and rivers in the Des Plaines River watershed and around 3.39 km² of lakes, ponds and open water (2011 Water Quality Management Plan Update 2011). The river flows to the east of the Pheasant Run landfill. Much of the area surrounding the landfill is wetlands and farm fields and drains into the river. Brighton Creek enters the Des Plaines River just east of the landfill and was sampled as well.
Figure 3: Land use in the Des Plaines River watershed in Wisconsin.
In order to determine if the Des Plaines River is being polluted by the Pheasant Run Landfill, three locations in Kenosha County were sampled and compared. One upstream location on the north side of Highway N was sampled in the Des Plaines River, upstream from the landfill. This location is labeled as Upstream Des Plaines in the study area map (Figure 4). This location indicated the river’s water quality without the influence of the landfill. The second sampling location was on the north side of Highway K, southeast of the landfill. This is labeled as Downstream Des Plaines on the study area map (Figure 4). The final location was in Brighton Creek, just south of Highway K and south of the landfill (Figure 4). This sampling site is upstream of the landfill and indicated water quality in the creek before flowing past the landfill and joining with the Des Plaines River. The proximity of this sampling point to the landfill raises the question of possible contamination in this sampling site, however, the sampling site will likely not receive a large amount of pollution because of the downstream flow away from the sampling site. Even with contamination of this site, if measurements Downstream are different than the combined upstream measurements, this can suggest pollution downstream of this sampling site. In addition, as the landfill is located all along the bank of Brighton Creek and the Des Plaines River, contamination downstream of the Brighton Creek sampling point, as well as the upstream Des Plaines point, would still be possible.
Des Plaines River Water Sampling Locations Near the Pheasant Run Landfill

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Figure 4: Sampling Locations
Field Methods

Sampling took place over the summer of 2019 on August 8th, 13th-14th, 20th-24th, 28th-29th and September 8th. On these 11 days, three samples were collected at each location for a total of 33 samples per sampling site. In each sampling site, the samples were collected near the stream banks at sites along roadways that were reached on foot. The samples were collected in various sampling bottles which were rinsed three times with the water from the sample sites before being filled. They were then taken back to the lab to be tested for pH, conductivity and dissolved oxygen shortly after collecting the sample and samples were refrigerated and tested for chloride within a month (lab methods described below). Temperature was also measured at each site every day that was sampled.

Discharge was measured at the three sites on September 8th, 2019. Five depth measurements were taken at equal intervals across the width of the bodies of water at all sampling sites. The widths of the streams were also measured. The velocity was measured by using a timer to find how long it would take for an orange to float a measured distance. An orange was used because the river and creek had low velocity so a high velocity flowmeter that would have been available for use would not have been effective. Oranges act as effective measurements of flow as they float partially submerged in the water (EPA 2012). This timing was repeated three times at each location. Discharge was measured on only one sampling day, but the discharge visually appeared to be consistent throughout the entire study.

The following equation was used to find the discharge,

\[ Q = \bar{d} \times w \times \bar{v} \times x \]  

(1)
where $Q$ is the discharge, $\bar{d}$ is the average depth of the stream in m, $w$ is the width of the stream in m, and $\bar{t}$ is the average time for the orange to travel in seconds the distance $x$ in meters in the stream after three trials.

**Lab Analyses**

Chemical analysis was conducted on each of the samples assessing pH, dissolved oxygen, conductivity and chloride concentrations. Three dissolved oxygen readings were taken at each location on each day, except for the last day in which only two dissolved oxygen measurements were taken from each site due to time constraints. Dissolved oxygen concentrations were measured using a La Motte Dissolved Oxygen test kit which implements the Winkler Method (Bruckner 2018). A 60 mL glass sampling bottle was filled with water from the sampling sites and then 8 drops of manganese sulfate and 8 drops of alkali-iodide-azide reagent were added to the bottle. The sample was capped and inverted 10 times then the precipitate was left to settle. Next, 8 drops of sulfuric acid were added to the solution. The bottle was capped and inverted several times until the precipitate dissolved. Next, 20 mL of the solution was transferred into another container. Sodium thiosulfate was titrated into the solution until it turned a pale yellow and then 8 drops of starch indicator solution were added, turning the solution blue. The titration was continued until the solution turned clear and the measurement of sodium thiosulfate remaining in the titrator was recorded.

Dissolved oxygen measurements were converted from parts per million to percent saturation. This was done by using a series of equations that consider air pressure, dissolved oxygen concentration and temperature to find the percent saturation (Water on the Web 2017).
Nonstandard air pressure was calculated using the equation:

\[
\ln P = 5.25 \times \ln \left(1 - \frac{h}{44.3}\right) \tag{2}
\]

where \(P\) is nonstandard air pressure and \(h\) is altitude at sampling location.

Next, equilibrium oxygen concentration at nonstandard pressure was calculated using the equation:

\[
C_p = C^* \times P \left[\frac{(1 - P_{wv}/P)(1 - \theta P)}{(1 - P_{wv})(1 - \theta)}\right] \tag{3}
\]

where \(C_p\)=equilibrium oxygen concentration at nonstandard pressure, mg/L, \(C^*\)= equilibrium oxygen concentration at standard pressure of 1 atm, mg/L, \(P\)= nonstandard pressure, atm \(T\)= temperature, K, \(\Theta\)=0.000975-(1.426 x 10^{-5}t)+(6.436 x 10^{-8}t^2), \(t\)= temperature, °C, and \(P_{wv}\)= partial pressure of water vapor, atm, computed from

\[
\ln P_{wv} = 11.8571 - \left(\frac{3840.70}{T}\right) - \left(\frac{216961}{T^2}\right). \tag{4}
\]

Finally, percent saturation was calculated using the equation:

\[
% Saturation = \left(\frac{100 \times DO_{mg/L}}{C_p}\right). \tag{5}
\]

Conductivity and pH were measured using an Apera Instruments PC60 Multi-Parameter Pocket Tester kit. The probe was calibrated, if necessary, for each day. The samples were collected and tested for pH and conductivity later in the day by placing the probe in the water and waiting for the measurement to stabilize. Three measurements were recorded at each of the sampling locations for all 11 days. The samples used for measuring pH and conductivity were then refrigerated and brought back to the lab within a month to be tested for chloride concentrations.
To find the chloride concentration of the samples, a Thermoscientific 9617 BNWP ion-plus sureflow chloride probe was used. Using 100.00 mL deionized water and 0.1658 g NaCl, a 1000 ppm chloride concentration standard was made in a volumetric flask. Various other standards were made from this standard by diluting the solution with deionized water to make 10 ppm, 25 ppm, 50 ppm, 75 ppm, 100 ppm and 250 ppm chloride concentrations. 500 μL of 1.0 M KNO₃ was added to each standard. The probe was placed in the standards, starting with the lowest concentration and working up to the highest concentration and the electrode potential in mV was recorded. Three trials were completed for each standard before testing the water samples. The water samples were then prepared by putting 10.0 mL of the water sample in a small cup with 500 μL of 1.0 M KNO₃. The probe was placed in each of the samples three times, recording the electrode potential in mV each time. After the water samples’ electrode potentials were measured, the standards were again tested for another three trials.

The chloride concentrations were then calculated from the data. A line graph was generated by finding the average voltage of the standards and plotting the voltage against the log of the concentrations. The equation for the line was then used to calculate the chloride concentration in the water samples based on the voltage that was measured with the probe.

The chloride measurements for two of the samples for each of the sample sites had to be thrown out due to outliers within the samples by day, likely due to contamination of the two samples. All corresponding samples at all sites were thrown out for consistency.

**Statistical Analysis**

In order to compare between sampling sites due to unequal discharge at the three locations, the discharge was used to find a ratio between the two upstream locations to
calculate the expected downstream measurement for all of the chemical properties measured in this study to compare to the measured downstream chemical properties. The expected downstream measurement was calculated by using the equation:

$$\text{Expected Downstream} = \frac{(U \times Q_U) + (BC \times Q_{BC})}{(Q_U + Q_{BC})}$$  \hspace{0.5cm} (6)

where $U$ is the chemical property measurement found upstream in the Des Plaines River, $Q_U$ is the discharge upstream in the Des Plaines River, BC is the chemical property measurement found in Brighton Creek and $Q_{BC}$ is the discharge in Brighton Creek.

An example of this, if the discharge was equal in both Brighton Creek and upstream in the Des Plaines River, then the chemical measurement downstream would be expected to be equal to the average of the measurements in the upstream locations. If, for example, chloride concentration is 100 ppm in Brighton Creek and 50 ppm upstream in the Des Plaines River, then it would be expected that downstream would have a chloride concentration of 75 ppm if there was no pollution.

This equation was used to find the expected downstream measurement using every corresponding measurement from upstream in both the Des Plaines River and Brighton Creek and was then compared to every actual downstream measurement.

When comparing the calculated expected downstream measurements ($n=33$ for pH and conductivity, $n=32$ for dissolved oxygen and $n=31$ for chloride) and the actual downstream measurements ($n=33$ for pH and conductivity, $n=32$ for dissolved oxygen and $n=31$ for chloride), a two-tailed t-test with a p-value criterion of $\alpha = 0.05$ was used. This made it possible to determine if there was a significant difference between the upstream and downstream measurements.
Results

Overall, three out of the four measured characteristics of the Des Plaines River were found to be significantly different upstream versus downstream. Brighton Creek and the downstream measurements for all chemical qualities followed a similar pattern over time of increases and decreases, with the exception of chloride which downstream measurements appeared to lag after the Brighton Creek measurements (Figures 6, 8, 10 & 12).

Dissolved oxygen was found to be significantly lower in the measured downstream than the calculated downstream (p=0.003512) (Figure 5). The calculated expected downstream (Upstream and Brighton Creek values combined) had an average of 91.17% saturation (Figure 4). Downstream, the average percent saturation was 78.33% (Figure 5). On average the percent saturation upstream in the Des Plaines River was 91.25%. Upstream in Brighton Creek the average percent saturation was 90.57%. Three of the days in the middle of sampling, Brighton Creek and Upstream measurements were very high with percent saturations over 100% (supersaturation may be due to rapid temperature fluctuations or photosynthesis) (Figure 6). Downstream and Brighton Creek also had higher measurements on these days, but not as high as Upstream.
Figure 5: Comparison of average percent saturation of dissolved oxygen expected downstream based on measurements upstream in the Des Plaines and Brighton Creek versus the actual measurement found downstream in the Des Plaines River. Dissolved oxygen was found to be significantly lower for the actual downstream measurements than expected downstream measurements (p=0.003512).

Figure 6: Average percent saturation of dissolved oxygen for three trials at each sampling location found on different days through the month of August and the beginning of September of 2019. This graph shows the change in percent saturation over time with a period of high saturation on August 23rd and 24th in all locations, but was most dramatic Upstream.
The pH was found to be significantly higher in the measured downstream versus the expected downstream (p=0.02091) (Figure 7). The average pH that was measured downstream was 7.99 and the average pH that was expected downstream was 7.88 (Figure 7). This trend was opposite of what was expected. Upstream in the Des Plaines, the average pH was 7.82 and in Brighton Creek it was 8.29. The pH measurements were relatively consistent over time with an increase in pH on August 23rd through the 24th, with the most dramatic increase seen upstream in the Des Plaines River (Figure 8). In addition, when comparing pH measurements over time with dissolved oxygen percent saturation upstream in the Des Plaines, these measurements follow a very similar pattern of increases and decreases over time (Figures 6 & 8).

![Expected vs Actual pH measurements downstream from the landfill](image)

Figure 7: Comparison of average pH expected downstream based on measurements upstream in the Des Plaines and Brighton Creek versus the actual measurement found downstream in the Des Plaines River. The pH was found to be significantly higher for the actual downstream measurements than expected downstream measurements (p=0.02091).
Figure 8: Average pH for three trials at each sampling location found on different days through the month of August and the beginning of September of 2019. This graph shows the change in pH over time with relatively consistent measurements over time, with an increase in pH between August 22\textsuperscript{nd} and August 24\textsuperscript{th}, mostly dramatically seen upstream in the Des Plaines River.

The conductivity was not found to be significantly different in the measured downstream versus the expected downstream (p=0.79862) (Figure 9). The measured downstream value was 799.3 μS/cm on average while the expected downstream value was 793.0 μS/cm (Figure 9). Upstream in the Des Plaines River, it was 785.2 μS/cm on average and in Brighton Creek it was 841.4 μS/cm on average. Conductivity measurements were similar to one another for most of the sampling in the three locations and measurements did not vary much over time (Figure 10).
Figure 9: Comparison of average conductivity expected downstream based on measurements upstream in the Des Plaines and Brighton Creek versus the actual measurement found downstream in the Des Plaines River. Conductivity was found not to be significantly higher for the actual downstream measurements than expected downstream measurements (p=0.79862).

Figure 10: Average conductivity for three trials at each sampling location found on different days through the month of August and the beginning of September of 2019. This graph shows the change in conductivity over time with relatively consistent measurements.
The measured downstream chloride concentrations were found to be significantly higher than the expected downstream chloride levels \( (p=0.00204) \) (Figure 11). The average chloride concentration for the expected downstream measurement was 50.49 ppm while the average for the measured downstream was 60.31 ppm (Figure 11). Upstream in the Des Plaines River, the average chloride concentration was 46.68 ppm. In Brighton Creek, the chloride concentration was 75.94 ppm on average. When calculating chloride concentrations, two trials were thrown out because they were outliers. The measurements for these two samples were far from the measurements of the other samples collected at the same location and date and it is likely that the sample was contaminated. Chloride concentrations in Brighton Creek and downstream fluctuated over time but concentrations upstream in the Des Plaines river were relatively consistent (Figure 12).

Figure 11: Comparison of average chloride expected downstream based on measurements upstream in the Des Plaines and Brighton Creek versus the actual measurement found downstream in the Des Plaines River. The chloride was found to be significantly higher for the actual downstream measurements than expected downstream measurements \( (p=0.00204) \).
Discussion

Overall, because of the significant differences in pH, chloride concentration and dissolved oxygen found between the upstream and the downstream locations, the results support the idea that there was a source of pollution between these sites that was contributing to the contamination of the river. This may have been from the landfill or it could have come from another source, such as agriculture.

Significant differences between the dissolved oxygen percent saturation upstream and downstream demonstrates that there is a difference between the locations so there is likely something that is occurring in between the sites that is causing this difference. Eutrophication occurs when high nutrient loads lead to algal blooms, which eventually die off. Bacteria populations then increase because of the abundance of dead algae. They consume the algae
and along with it, oxygen, reducing dissolved oxygen in the body of water (Yang et al. 2008).

Sources of possible pollution in the area include the landfill and the agricultural land nearby which both could contribute nutrients to the river due to waste or fertilizers.

When calculating the percent saturation for the upstream, downstream and Brighton Creek measurements, some of the measurements were greater than 100% saturated, so the water samples were found to be super-saturated. There are different explanations for why this may be. One possibility is that there was a rapid change in temperature in the river or stream and since the water is very slow moving, the water did not equilibrate fast enough to accommodate for the higher temperature, resulting in a percent saturation over 100% (Xylem 2019). Another possibility is that the photosynthesis of plants in these bodies of water increased the amount of oxygen in the water to a percent saturation of over 100% (Xylem 2019). It is possible that either or both occurred, increasing the dissolved oxygen measurements.

In addition, upstream in the Des Plaines River the pH also had a major increase at the same time that the dissolved oxygen increased. During this time period, the water temperatures were much warmer that were measured on other days and it had rained previous to these measurements. A possible explanation for the increase in pH and dissolved oxygen may have stemmed from agriculture. Fertilizers may have been applied during this time as many can increase pH. Additionally, as the rain washed the fertilizer into the river prior to the increase in pH and dissolved oxygen, this would have led to excess nutrients available in the water. Then on the day that it was sunny, August 22\textsuperscript{nd}, there may have been an algal bloom, increasing the dissolved oxygen in the water.
The calculated expected pH values were found to be significantly lower than the measured pH values. This indicates that the downstream measurements were significantly higher than the mixture of the Upstream and Brighton Creek measurements. This result is opposite of what was hypothesized; however, this does not necessarily rule out contamination from the landfill. Measurements of pH in leachate have resulted in higher pH values in the past and is often found in leachate older than 5 years old (Abbas et al. 2009; Kjeldsen et al. 2010). The increase in pH could also be due to runoff from farm fields as some fertilizers can increase the alkalinity of soil and water.

Conductivity expected values were found to be very similar to the measured values. Upstream in the Des Plaines, Brighton Creek and Downstream had very similar values throughout and did not follow a noticeable pattern of one location continuously having higher measurements than another. However, there was found to be a significant difference between the calculated expected downstream and the measured downstream values for chloride concentration which would not be expected as an increase in chloride ions would lead to an expected increase in conductivity.

Sediment types in the beds of the streams in this study varied and could offer a possible explanation for the increase in chloride but no increase in conductivity. In the Des Plaines River, the sediment consisted of a large amount of organic material, while in Brighton Creek, the sediment was primarily sandy. This could have led to the consistency of conductivity downstream versus upstream. Clay and organic material have a higher cation exchange capacity than sand so it is possible that the ions bonded to the sediment in the river, but not in the creek. Because Brighton Creek was sandy, the ions would be less likely to bond to the sediment
in the creek, but as the creek joined with the river, the sediment was mostly organic material so
the ions were able to bond. Chloride is an anion so it does not bond with soil particles (Smart
Fertilizer 2020). Even though the chloride ion concentration was found to be higher
downstream than expected, conductivity would not necessarily increase because it is possible
that other ions had bonded with sediment particles. More ions would bond with this sediment
and would reduce the number of ions that were found upstream, offsetting the increase in
chloride ions downstream.

In the future, different approaches could be taken to strengthen the research.
Monitoring more locations would be beneficial because it would create a more complete
picture of where the contaminants are coming from. Some sites that would be useful to
monitor include small streams that enter the river near the landfill and a few sites further
upstream in the river and Brighton Creek. It is possible that if the landfill were contaminating
the water system that it could potentially be contaminating the Brighton Creek site as the
landfill is close to this sampling point. The land that the river and creek flow through is
relatively flat, and although the Brighton Creek sampling point is actually upstream of the
landfill, it is possible that contaminants from the landfill could still reach this location as it is
downhill from the landfill, as can be seen in a topographic map of the study area (Figure 13).
Brighton Creek and Downstream also followed similar patterns over time for measurements of
chemical qualities, which could indicate that Downstream is changing because of Brighton
Creek. However, because Upstream Des Plaines River had a greater discharge than Brighton
Creek, it would be expected that Downstream would follow the pattern of Upstream more
closely, but this was not the case. This additionally leads to the possibility that Brighton Creek
could also be contaminated and an outside source is contributing to the consistent fluctuation in both locations.

Figure 13: Topographic map of study area
Sampling further upstream would help to balance the possibility of the Brighton Creek sampling site being contaminated. Measuring further upstream would offer a more accurate representation of Brighton Creek without the influence of the landfill to determine if the fluctuations in chemical measurements over time are due to contamination from the landfill or other sources further upstream. Samples were collected during this study further upstream on one day and interesting patterns were noticed but not enough data was collected to draw conclusions. It would be interesting to collect more samples from upstream in the river and the creek to see if these patterns are consistent over time. This study was able to identify that there is something influencing the water in the river between upstream and downstream of the landfill, but with more sampling sites it would be easier to identify the actual source of these differences.

Sampling over longer periods of time could also hold benefits to the future of this study, and ideally on a daily basis. This would ensure that the patterns that were observed are consistent throughout the year, such as times when agriculture is not in season, when temperatures are different and when plants, animals and bacteria are less active. It would also make it possible to compare measurements over time more accurately and it would be possible to compare trends with precipitation.

Another improvement that could be made to this study would be to measure discharge at each location each time so the ratios would be more accurate. In this study, discharge was measured at each site only once because of time constraints and the visual consistency of
discharge over time at each location. If there are inconsistencies in discharge, this could affect the results.

Using better measurement tools could be beneficial such as a dissolved oxygen probe instead of using the test kit for more precise measurements. This would reduce the possibility for human error in titrations. In addition, it could also be interesting to look at other pollutants such as heavy metals to determine if there is a hazard that is posed to human and environmental health. Heavy metals are often present in leachate and if the landfill is polluting the river, heavy metals could potentially be entering the water system.

The results for three of the four measurements were found to be significant, but this does not mean that the water quality is unsafe at this point. Dissolved oxygen levels were good both upstream and downstream of the landfill, maintaining a concentration above 5 mg/L for all samples collected, which is the minimum dissolved oxygen set for the state of Wisconsin in non-trout water systems (EPA 1988). All chloride measurements were also below maximum concentrations of 230 ppm (EPA 1988). The pH of the water system must be maintained between 6 and 9 and this was the case for all samples (DNR 2010). It may be possible that the landfill is contributing to the contamination of the river, but the water quality of the river according to the measured chemical qualities is not dangerous to the health of humans or aquatic life. In addition, the differences, even if significant statistically, were not dramatically different upstream versus downstream. Even as the quality of the water is still at safe levels, it is still important to monitor the water quality to ensure that it stays that way.
Landfills pose risks to bodies of water, even if maintained properly and are still in operation. This study indicates that it is likely that the landfill has impacted the water quality in the Des Plaines River, and even though the difference is small upstream versus downstream of the landfill, it is still concerning that the difference is present. Over time, landfills may contribute more pollution as linings deteriorate and the risk of pollution is high when landfill are located close to bodies of water because leachate spreads quickly in water. If water is polluted, this leads to contaminated drinking water. Landfills are often located in rural areas and the people that live near landfills often use ground water instead of treated city water. Contamination from landfills increases the risk of damaging recreational benefits of bodies of water, even far down stream with enough contamination. This can harm fish as well as make the water unsafe to swim in. The risks are high with landfills located near bodies of water.

Waste is being created in amounts like never before and it is important to carefully choose the sites of new landfills as old landfills reach full capacity. Landfills should not be built near bodies of water in order to protect from the rapid spread of pollution that will likely occur over time. In addition, reducing waste by improving recycling as well as reducing the amount of waste in general are ways to keep from filling landfills quickly and requiring more space for landfills. With a reduction of waste and careful planning of the location of landfills, this can help to reduce the damage created by landfills.

Although it is difficult to directly determine the source of pollution, the data from this study supports the conclusion that there is something polluting the river in between the sampling points, possibly from the landfill. It is important to monitor water quality to maintain the health of ecosystems and safe conditions for humans. This study helps to better understand
the possible effects of landfills on neighboring water systems and emphasizes the importance of monitoring surface water for contamination.
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