A Discussion Paper on:

Cumulative effects from alteration of headwater drainage features and the loss of ecosystem integrity of river networks

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1.0 Introduction

How water security and biodiversity will be affected by climate change and urban sprawl/agricultural intensification is of paramount importance. Society and scientists are both struggling to understand and manage the uncertainties and risks associated with the cumulative effects of changes to stream systems upon which society is dependent for water supplies, security, recreation and wellbeing. However, the science that guides decisions about effective land and water use is incomplete, largely because minimal effort has been directed at understanding the role of headwater drainage features (HDFs) in watersheds. Uncertainty regarding the effects of development on HDFs and potential thresholds for response make it very difficult to set guidelines on development in headwater areas while maintaining ecosystem services. At the moment we can only determine the consequences of development after a new altered ecosystem state is attained, and this presents major challenges to management. In fact, legislation often overlooks HDFs as so much of the legislative authority for water protection is focused on exploitable fish (which often do not occur in headwaters) or navigability (e.g. Nadeau & Rains 2007; Leibowitz et al., 2008). Once degradation occurs, it is either expensive or impossible to restore degraded systems; therefore, it is necessary to protect sufficient source areas and their functions prior to irreversible damage occurring.

1.1 Background on headwater drainage features

HDFs represent the start of any flowing water ecosystem, and are also potentially the most sensitive to alteration of any part of the fluvial network (e.g. Fagan, 2002; Gomi et al., 2002; Richardson & Danehy, 2007). For this document we use “HDFs” to mean those drainage features that show evidence of fluvial processes (Feminella, 1996; Gomi et al., 2002). They include: both permanent and intermittent streams, wetlands, swales, and altered features. In terms of water security, ecosystem services and biodiversity conservation, these areas have the greatest response to local land use alterations. HDFs support downstream food chains (e.g. Wipfli et al., 2007; Richardson et al., 2010; Hennigar 2012), and provide important ecosystem services such as nutrient cycling, flood attenuation and sediment regulation (TRCA, 2007; Sadler-Richards, 2004).

HDFs are often overlooked and undervalued even though they can constitute 70-90% of a drainage network (Meyer and Wallace, 2001; Nadeau and Rains, 2007; Finn et al., 2011). There are many reasons for this including the small size of individual HDFs, their large number within a watershed, and their appearance as little more than ditches at certain times of year. However, these features work cumulatively to provide a suite of ecosystem services to the watershed. Impairments to the watershed occur when landowners alter landscape features to increase land yields or manage hydrologic nuisances through channelization and enclosures. Each headwater stream that is altered represents an incremental change in the water supplies, habitat, and ecosystem services in the associated catchment. At some point the impairment of a HDF can be sufficient that its contribution of ecological services to downstream reaches is negligible, or becomes a negative effect. In effect the tributary becomes trimmed from the network. In the absence of scientific information on how the number of streams “lost” reduces ecosystem values, we cannot wisely predict how much of the fluvial network requires protection to ensure that services such as flood mitigation, water quantity and quality, habitat conservation,
and downstream recreational opportunities continue to be provided. As a result, accurate prediction or measurement of cumulative effects in downstream riverine systems is difficult (e.g., Seitz et al., 2011). This difficulty suggests we start at a more tractable scale of HDFs and study how impacts at that scale evolve down the channel network.

Two main challenges complicate quantification of the relative contribution of each HDF to downstream waters. These are:

1) Traditionally there has been an absence of quantitative datasets that represent a broad area and cross-section of bio-physical conditions; and

2) No clear path to analyze the data in a way that provides predictive models for use in scenario testing by decision-makers.

Overcoming these challenges is essential to making informed decisions on how much and what types of alterations can occur without significant alterations to ecological integrity. An additional set of challenges focuses not on the effects downstream, but on how to ensure that the distribution and condition of HDFs sustain their integrity in a landscape context.

Box 1: Cumulative effects are the only effects that matter

Government agencies are often required to consider cumulative effects in their decisions, but there is very little guidance on how to do so (e.g., Canadian Environmental Assessment Act). A large amount of this uncertainty results from our lack of understanding of the complex and specific ways that these effects accrue. From a drainage network perspective there are at least three types of cumulative effects that require understanding.

1. How alterations in HDFs influence downstream reaches: The effects might be non-linear; synergistic, or antagonistic (e.g., Wagenhoff et al., 2011), or be confounded by dilution from any discharges between the headwaters and a downstream measurement site (Harpole et al., 2011; Seitz et al., 2011). In addition, responses may lag impacts by years to decades, for instance, by sediment storage along the network (e.g., Gomi et al., 2002).

2. How different independent actions in the watershed or within a particular HDF interact to alter a dependent variable used to measure stream conditions (e.g., pollution, habitat alteration, harvesting on an important fish or animal).

3. How conditions within individual HDFs contribute to the overall state of a watershed. As HDFs are channelized or diverted underground, the habitat available for headwater organisms disappears. HDFs are responsible for a large proportion of the biodiversity and ecosystem functions in fluvial networks, and have a high variation in species diversity within and across systems (Clarke et al., 2010; Finn et al. 2011) showing that they are not fully substitutable for each other.

Fortunately, nested fluvial networks are an ideal setting to study these various pathways of cumulative effects on stream systems (Grant et al., 2007).
context. In Ontario, a group of collaborators began to address these gaps by first developing a standardized data collection protocol (Stanfield, 2013) that can be applied to different sizes of streams to quantify upstream impacts on fluvial and sediment pathways, fish, benthos, and other components. Then through a collaborative approach, the modules in the protocol have been applied to create a spatially extensive and comprehensive dataset (Figure 1) that could be used to evaluate cumulative effects throughout the river continuum.

With these extensive data, the group organized a workshop to explore approaches for answering the question, “What headwater drainage features could be altered before downstream sections of streams would be negatively impacted?” The workshop used the analogy of an arborist who decides which limbs to trim from a tree to help visualize the consequences of “losing” tributaries within a catchment. In this context a tributary is “trimmed” from the network when the pathways and processes it contributes to ecological integrity are significantly altered. This document summarizes the findings of this session that was held at the University of Ottawa, Ontario, January 24-25, 2013.

Our objective in publishing this discussion paper is three-fold. First, we would like to disseminate the results of the January workshop to interested partners. Second, we hope that others will become engaged in the challenge of developing science-based decision-making tools for addressing cumulative effects from headwater alteration. Third, we hope that researchers and managers will consider using our collaboratively collected and accessible datasets to help answer questions.
The objectives for the workshop were to develop:

1. A prioritized list of challenges and solutions to HDFs being better managed within a developing landscape.

2. Study designs that, when implemented, could support a pragmatic decision-making process to enable a manager to identify “How many and which headwater drainage features can be impacted before mainstem reaches of rivers also become impacted?”

3. A process to address major barriers to building tool(s) for use in applying science-based approaches to decisions about HDFs.

The following sections outline the results.

2.0 Priority challenges and solutions to better HDF management/protection

The following list of challenges and their solutions are ordered to reflect the workshop participants’ prioritization for implementation. This list identifies new approaches to better manage HDFs using adaptive management and science based decisions.

2.1 Public and decision maker indifference to HDFs

Communicating to the public and decision makers that small HDFs are worthy of policy protection is of paramount importance. Clear and accessible demonstrations of direct linkages between cumulative HDF development and human health or ecosystem services are needed to engage the public and decision makers. A part of the strategy could be to communicate the financial benefits and reduced risk, both short and long term, to land owners of better managing HDFs on the landscape as a powerful motivator of behavioural change. For more information on this see implementation strategies below.

2.2 Model predictions of effects from alterations to HDFs are not available to decision makers and land owners.

Decision makers and the public require means to readily evaluate the effects of various actions on the state of their watershed and community. The intent is that a communication tool, or tools, referred to here as a Decision Support System (DSS), is necessary to assist with public buy-in and appreciation of the dynamic relationships between HDFs and measures of human/ecosystem health described above. The challenge is that developing a DSS that could evaluate the effect on a variety of ecological, biological, human health, economic, and social indicators requires tremendous collaboration and resourcing so that the tool remains both functional and up-to-date. Such an ambitious need will require a phased-in approach that builds functionality in an iterative way. Development of a graphical interface that enables basic scenario testing utilizing land cover information is readily achievable once predictive models are
developed although, as described below, the process of DSS and model development should be linked.

Another challenge in developing the DSS is to match the appropriate level of complexity and detail with the user’s needs and skills in a way that does not simplify the science to the point where the predictive power to quantify effects of altering complex ecosystem linkages is lost. Implementation must therefore consider the needs of users and scientists and might require the use of summary variables (e.g., percent impervious cover) or surrogate response variables (e.g., a total dissolved solids rather than specific water chemistry parameters) for implementation.

Finally, we wish to emphasize that the intent of the DSS and the predictive models is to foster collective learning and an understanding of the iterative nature of managing cumulative effects among decision makers. In this way, we hope that a more collaborative governance model for watersheds will emerge that can address the “wicked problems” of sustainability (Paquet, 2013).

Box 2: The scientific approach to decision-making requires measurable indicators of success

Indicators provide a measure of a certain aspect of the state of a system, in this instance an ecologically and socially sustainable watershed. By definition they can be compared to a benchmark to determine a condition that is valued by humans (e.g., water quality or quantity, biodiversity, human health, etc.). The indicators provide a measure of the state of some valued component (VC) of a watershed. For example, the indicator dissolved oxygen concentration provides a means of measuring and communicating the state of water quality to the public. The VC in this instance is water quality. The VCs can reflect any number of important or valued characteristics of a watershed (e.g., VEcologicalCs, VSocialCs, VHealthCs). In this instance, indicators provide a measure of the VC that reflects the state of the watershed community.

2.3 Quantifying effects of actions on indicators is complicated by scale

Management decisions are based on a variety of criteria, and are likely to have differing effects on aquatic systems depending on where in the watershed an activity of impact may occur. Therefore, it is important to assess the state of HDFs based on temporal and spatial changes in aquatic response indicators relative to background or reference conditions (Dube et al., 2013). Quantifying the state of each feature will aid in assessing how development within a watershed, or loss of certain HDFs, will result in downstream changes and there are many approaches that can be used to measure this state (see Stoddard et al. 2006 for a review). However, the degree to which downstream effects are associated with specific disturbances to an upstream HDF will be dependent on factors related to spatial (e.g., smoothing effects) and temporal scales (Dube et al., 2013) that may not be independent, as described in the examples below.
Temperature, a widely used as a predictor of stream biota (see Schlosser, 1991), offers a good example of how spatial and temporal scale can influence observed conditions. Stream temperature changes diurnally in response to air temperature fluctuation. It also tends to increase downstream as volume/depth increases, reducing surface area interactions with air. Similarly, modifiers such as riparian vegetation can differentially influence stream temperature depending on the size of the stream it is shading (i.e., its location in the network) (Richardson and Danhe, 2007). Finally the magnitude of change in temperature and overall patterns at a given location may change in response to climate. With these dualities of scale effects, the complexity of factors that should be considered to make quantitative models sufficiently predictive may counteract the benefits of this simple and sensitive metric.

There can also be lag times between an activity and subsequent environmental response. Geomorphic processes that form channels work at much longer time spans than is typically considered in land use planning. For example, Trimble (1983) has shown that in high energy systems, for every ton of sediment supplied to a stream, over time (decades), stream widening occurs that can generate as much as seven tons of sediment in the downstream reaches. However, the response time is influenced by the available energy in the system such that, if insufficient, the initial response of the system will be infilling. Such has been the case in many of the streams that flow from the top of the Oak Ridge Moraine in Southern Ontario. In the 1800’s the moraine was deforested, releasing large volumes of sediment into the streams. In the steep headwaters, streams responded by down-cutting and creating massive gullies. Further down the system, where gradients were lower, sediment filled the valley-lands. Many of these systems are still responding to this major disturbance. Predictive models should consider how time lags are influencing the stream condition.

The same types of dichotomies, where one stressor could have two divergent effects (e.g., erosion or infilling), are true for other factors, such as those that influence the frequency, magnitude and duration of discharge (hydraulics) (Stanfield and Jackson, 2011), channel roughness from vegetation and large wood (Leopold and Maddock, 1953), and chemical inputs to the system (Meyer et al., 2005). Such variance in datasets can also reflect the nature of actions or stressors acting on an indicator. For example, “press” actions such as those that relate to long term land use change may cause a permanent change to a system (loss of biodiversity etc.). Whereas shorter term “pulse” actions such as extreme weather events or a chemical spill might result in a shorter term impact and will require special approaches to be accommodated in predictive models.

Characterization of the influence of each HDF, or combination of HDFs, on downstream VCs requires an understanding of each feature’s role relative to its location in the watershed. Ideally, landscape models will incorporate spatial considerations in the model predictions. For example, as a result of dilution or smoothing factors, actions that are closer to a sample area are more likely to have a measurable affect than those that are further away (Wang et al 2001, 2003; Stanfield and Kilgour, 2012). It is also true that predictor variables will differ in their importance depending on the proximity of a land
parcel and the feature position within the watershed. Quantifying this variability will require approaches that measure the distance-effect relationship and the degree to which configuration of land cover affects the response indicators. Fortunately, many of these approaches are already available for terrestrial landscapes (Fortin and Dale, 2005). The degree to which these second-order effects, for example local land use (a single lot/field), are incorporated into predictive models, and future DSS, will depend on how much unique variance these factors explain relative to first-order effects, such as catchment land use. To date many of the studies have found that minimal predictor power is added by local predictors of land use (Wang et al 2001, 2003; Stanfield and Kilgour, 2012). This mismatch between the scale at which predictive models are effective and the scale at which decision-makers require direction, (e.g., individual lots) will be an on-going challenge.

2.4 It is challenging to link measures of ecological condition and human health in ways to direct public action

Most people would not understand or support a public policy mandate to manage stream temperatures or substrate size distributions as a means of reducing heart disease or increasing economic stability in a community. Support for such an initiative requires evidence of strong linkages between the ecological VC and other VCs and the clear communication of the perceived pathways of the effect. Integration of the VC models would benefit from the use of similar predictor variables, especially those predictor variables that are generally not modified by humans and are commonly used to describe the “state of a place” (e.g., geology, soils, dendritic network). Drawing such linkages will require approaches to the interpretation of both human and ecological indicators that are place-based in order to be relevant at the most important scale that matters to the public: the individual property owner.

The impact of management decisions on the state of a larger river will vary greatly depending on the spatial and temporal time frame around which the effect is being evaluated (see above). Considered individually, the effect of removing a barrier from a headwater stream may not be directly measurable 20 km downstream. This is not to say that the decisions are any less important, it is just that the parameters used to evaluate risk will be quite different at the two scales and, as such, application of a single scale of inventory and interpretation is not desirable. Which indicator, or suite of indicators, is chosen to represent the VCs (e.g., flow, sediment, invertebrates, heart disease, etc.) will depend on what aspects of the ecosystem society wants to manage, regardless of the relationship of the VC with predictor variables. Whatever indicators are chosen for the VCs, the metrics should be measurable regardless of the condition of the VC and across the range of disturbances. For example, fish assemblage is often used as a metric of stream health (Stanfield and Kilgour, 2012), but is unusable in fishless streams. Having a variety of correlated indicators, for example water quality in fishless streams, increases the depth of interpretation of causal pathways, but adds complexity and cost to monitoring.
3. Model explorations

There are strategies that could be applied to address at least some of the challenges outlined above. Our intent is not to offer a cookbook approach; rather we provide some approaches to identify priority HDFs in a watershed. We propose that an eco-informatics approach be employed because of the need to utilize fairly large datasets and the intensive analytical approaches that, when successful, would attempt to characterize the state of the system at any point in the network. Eco-informatics is an emerging science that attempts to integrate data, knowledge and methods necessary to provide ecological data to a scientific or policy-making process (Michener and Jones, 2012). We build the conceptual model in stages, offering one example of a VC to demonstrate how it could be applied to identify critical HDFs. We also explore and describe how a differential equation might offer insight into the processes that underlie the relationships identified using the eco-informatics approach. We believe the two pronged approach, of using eco-informatics and a differential equation, offer the best coordinated solution to the problem.

Figure 2: Schematic of a stream network that is used in each example below to quantify conditions at point x,y. Numbers indicate individual stream or HDF segments.

3.1 A conceptual framework to quantify tributary influences on stream condition

Relationships between indicators of VCs (e.g., hydrology, temperature, biota, water chemistry) and landscape predictor variables at a downstream location of the river network (shown as point x, y in Fig 2), are influenced by the combined conditions of the upstream HDF segments (numbers 1, 3, 4, 5 and 9 in Figure 3), as well as the contributions from connecting segments (2, 7 and 8 in Figure 2). There will also be local influences
from factors such as groundwater upwelling or stream restoration etc. that should be considered as a modifier of the stream at the point of measurement. Therefore, the state of a VC at point x, y (VC_{xy}) is a function (f) of various upstream predictor variables that are spatially explicit measures that capture the effects of all upstream tributary-specific inputs (s_i) and local influences that are present at the site (L):

\[
\text{Eq. 1: } VC_{xy} = f(s_1, s_2, s_3, \ldots L)
\]

With this approach the contributions of each segment are characterized using its unique drainage area (subcatchment) and the features known to be correlated with the VC. For example, many studies have demonstrated that area, geology and land use, as well as other factors such as major ground water upwellings are primary predictors of stream conditions of many VCs (e.g., Dunne and Leopold, 1978; Wiley et al., 1997; Wang et al., 2003; Stanfield and Kilgour, 2006, 2012). So each s_i where, i represents each segment, could be described as Eq. 2. However, it is also likely that each characteristic would contribute differently to each VC, so that the characterization of each segment (s_i) would include a contribution coefficient (c) for each predictor variable.

\[
\text{Eq.2: } s_i = (c(\text{area}) + c(\text{geology}) + c(\text{land use}) + c(\text{other factors}))
\]

This simplified approach of characterizing individual segments is readily adapted to accommodate a variety of spatial configuration effects, where they are important. For example, proximity of features (Wu, 2004), position and the “branchiness” (Fausch, 2002) and fragmentation of the dendritic network (see Cote et al., 2009), are all known to influence stream conditions. To accommodate these relationships, the model could include a suite of spatially explicit predictor variables that could also include regionally important characteristics of an ecozone (see Frissell et al., 1986). To accommodate these effects Eq. 1 would be modified as follows:

The VC at position (x,y) could be predicted by:

\[
\text{Eq. 3: } VC_{xy} = a \times b \times f[s_1, s_2, s_3, \ldots s_i]
\]

Where a is a coefficient that represents an upstream spatial geometric index of the stream, b represents one or more network-wide variables that are specific to an ecozone (e.g., climate) and f is as defined above. To demonstrate that the process is achievable, we provide one example of how a simple model might work.

GIS tools can provide segment-specific watershed polygons for attributing each s_i with existing geospatial information, including local conditions. In southern Ontario for example, these datasets are available from Land Information Ontario. Whether the model is fitted using additive, multiplicative or bootstrapping approaches will depend on which approach best fits the data. The intent is to quantify the individual characteristics of each segment to the VC in ways that enable ‘what if’ scenarios to be applied to evaluate the effect of removing/trimming an s value from the system. For example, modellers could exclude the entire attributes of a segment (e.g., S_i) or just one component of the segment
(e.g., land use in $S_3$). Differences in predictive power of each model would identify the importance of the attributes of each segment. Researchers could then explore the degree to which the other predictor functions improve the model fit. For example, the dendritic connectivity index of Cote et al (2009) could readily be applied to a study area to provide an index of fragmentation for each stream segment using the FiPex tool (Oldford and Sam, 2010). Regionally specific climate data is available to be applied within a predictive model to accommodate $b$. The model with the highest predictive power is selected, as are the segment attributes that most greatly influence the model’s predictive power. Non-linear responses (see Stanfield and Kilgour, 2006, 2012 for examples), or interaction terms can be incorporated within the model where needed. Such characterizations and calculations are easily automated within existing GIS applications.

Such an approach could be employed to model any VC, including extreme values such as minima or maxima. Finally, model development could be done iteratively as datasets become available and our understanding increases. For example, initially, this approach assumes that space can be substituted for time (i.e. the range of possible VC states for a site can be estimated by measuring indicators across a number of sites with a gradient of land use conditions), and that parameters of the segment functions do not vary in time. These assumptions could be later relaxed by incorporating a time lag effect, either through one of the system-level coefficients ($f$, or $a$) or as part of each $s$ value when better time series predictors become available.

This approach is data intensive and relies on gradients in stream conditions to develop response relationships. However, the approach is still largely a black box approach and the accuracy of the predictions will be limited by a number of confounded sources of error (see inset box 1) that effect the scale at which VCs can be reliability predicted relative to the scale at which land use decisions are made. To explore and better understand the causal pathways will require more detailed studies, likely in selected watersheds that complement this broad scale approach. Insights could also emerge from the differential equation approach described below.

### 3.2 An approach to understanding causal pathways

Patch models, or metapopulation models, are widely used in terrestrial spatial ecology to describe and understand spatially distributed systems (e.g., Wu and Levin, 1994). These process-based models can help identify the theoretical measures for the (relative) importance of a given patch for a species (Ovaskainen and Hanski, 2003b). In this way, the relative importance of even currently unoccupied patches can be quantified and potentially found to be crucial for a species’ survival. Such models commonly take the form of differential or difference equations and the specific form of the equation will vary, depending on whether they attempt to quantify for species abundance, community structure or species patch occupancy probabilities. Provided the VC of choice is a fish or benthic community, such models could help evaluate the significance of individual segments ($s_i$) to each node/location in the stream network.

Such analyses could help elucidate a long standing debate about the various factors (e.g., population growth/decline verses colonization and extinction dynamics),
that influence fish and other populations and the scale at which they manifest (Fagan, 2002). We suggest that, for now, complete parameterization of the population dynamic approach is not feasible; a strong understanding of the quantitative influence of environmental factors would be required, as described for population growth and movement rates in previous sections above. Rather, we suggest that this approach be considered as a parallel means to explore hypotheses generated with the approach described in section 3.1 in ways that help understand correlations. An example of how this approach could be applied is provided in Appendix II.

4.0 Implementation

Three considerations are identified as being critical to development of a cumulative effects process that must be implemented simultaneously. First there must be continuity of information flow between the researchers’ needs, the developers of the decision support system and the decision-makers’ needs. Second, a clear, stepped – adaptive process is needed to deliver initial predictive models for use in a DSS that will improve over time. The final consideration is that implementation must engage the real decision-makers, the property owners and water users in ways that impart strong motivation for behavioral change. These three things are all essential to address the perceived core issue of the tragedy of the commons (Hardin, 1968) that is at the root of the challenges that communities face. Key needs to address these three challenges are summarized below.

4.1 Clear communication during development

The key to success of the predictive models is that they clearly meet the needs of the stakeholders that have a role in the activities carried out on the land and in the water (i.e., land owners, businesses, planners, etc.). As part of this process, models need to be accessible to the decision makers through an easy-to-use DSS. The process will require new approaches by scientists, managers, modelers and DSS developers, at all stages, to ensure the appropriateness of indicator selection. Also, a clear understanding of the implications of VC choice on the precision of model predictions across spatial and temporal scales must emerge and be understood by all process participants. Integral to the success of the working group(s) will be the development of a clear conceptual framework around which the models are developed and presented in a DSS.

It is as yet unknown whether the creation of predictive models that are readily accessible in a DSS is sufficient to help stakeholders with decisions. Nor is it clear that the issue of understanding the scale of impacts is important to managers. That is, an altered feature might be predicted to have an effect on the immediate downstream segment, but its effect might be insignificant, when considered in isolation, by the time the stream intersects with a larger segment downstream. To ensure that the DSS and the predictive models meet management needs, it is proposed that a process be established that brings together decision makers and researchers on a regular basis to communicate needs and model development progress/limitations. This iterative process will ensure that the models are user-friendly, jargon neutral, and meet both scientific rigor and user needs.
4.2 An adaptive approach is essential to success

Development of the models and incorporation into a DSS will require transdisciplinary collaboration among communities of practice to select an appropriate suite of indicators and to monitor their condition across different study areas as part of an adaptive management process (Figure 3). This process offers a feedback system between targets, monitoring, analysis and adaptation or adjustments (adapted from Holling, 1978). As researchers develop the models they will have the opportunity to evaluate various predictor variables. A key to success would be to do this in a collaborative way so that, if possible, a core set of predictor variables is used to develop initial models of VCs. A diagnostic model that attempts to explain causal pathways that emerge from the black box VC models could utilize more specific predictor variables and diagnostic indicators. This dual approach will facilitate communication with decision makers and support the need by researchers to explore new patterns.

Characterization of land use that is used in the predictive models must be capable of differentiating activities between properties at the scale at which most decisions are made: subdivisions, farms, individual properties. To that end, the models will require one or more measures of land use that effectively quantify the contribution of each land parcel to the condition of the watershed. Impervious cover is a widely recognized measure of urbanization (Leopold, 1968; Shaver and Maxted, 1995; Boyle et al., 1997; Brydon et al., 2006,) and, more recently, researchers have begun to use more comprehensive measures of land and water use that can be applied to all land parcels. For example, Stanfield and Kilgour (2012) applied a land disturbance index to all land parcels in a watershed. Such approaches lend themselves to the attribution, and therefore scenario testing, of both positive (e.g., low impact development best management practices) and negative (e.g., dredging, tiling, and cutting of riparian vegetation) land parcel management.
Two challenges that will require give and take between the scientists developing models and the rest of the implementation team will be time delays in the development of the various models and in the differences in predictive power for different types of models. Base data layers will continue to improve and managers will inevitably wish to utilize the newest information in decision-making. However, model development will always be “behind the times” and therefore potentially deemed no longer relevant. Project teams are encouraged to build predictive models to facilitate an iterative process through which new scenarios could be “tested” in a Delphi process. For example, models can only confidently predict scenarios based on the input categories of, for example, land use. However, the DSS could offer options to modify contributions from specific land use categories to reflect new, as yet, un-modelled land uses, such as low impact development (LID). Being able to incorporate local differences in land use will improve the utility of the tool and offer an opportunity to improve models iteratively. Ideally, and over time, the boundaries of effects from each category of disturbance will emerge through the modeling efforts to define the confidence limits on each model.

4.3 User pays and incentives linked to the decision support system is a motivator of change

Managers and society in general, must be engaged in the process of model development and implementation from the onset to insure that the tools meet their needs and expectations, and that they understand the inherent limitations of the tools. But adoption of the process by land and water users must also have a means to invoke positive change. In many municipalities a new ‘user pays’ approach is being applied to address issues of sustainability, such as storm water that affects entire communities (see: http://www.kitchener.ca/en/livinginkitchener/Stormwater_Utility.asp). Linking property levies and incentive programs to outputs from a property level evaluation, not only of storm water but potentially other attributes shown to be important characteristics of a healthy landscape, provides a direct means of engaging citizens to improve properties for the common good. With effective landscape models that correlate property attributes with VCs, such a process is attainable. Property owners would be able to run scenarios on land use options and evaluate how the changes are predicted to affect their watershed community. Scenarios could also be run for storm water levies. Ultimately, models could also predict changes to property values, insurance rates etc., once linkages to flood risk or human health metrics are considered. Such a tool would be a powerful motivator of change.

5.0 Next Steps

In some ways, the development of this paper is but one more step in the implementation process. Considerable resources have been invested in Ontario to attribute stream conditions at thousands of sites that range in size from the smallest HDFs up to the outlets of rivers (Fig. 1). These datasets have been collected using standardized methods and with the main purpose of better understanding cumulative effects to streams and, in recent years, to address some of the questions posed in this paper. The data are freely available to researchers to help with this task (http://www.comap.ca/fwis/).
However, the questions posed in this paper go well beyond developing an understanding of ecological conditions, to how to incorporate sustainable practices into land and water management decisions. Fortunately there is a growing interest in a more integrated approach that merges ecological, social and health analysis as a means of understanding and managing communities and landscapes (see for example: Palmer 2012).

The technological tools to enable the communication that will be essential to success are available. There is clearly a need for innovative approaches to utilize these technologies that will engage society at all levels. Improvements will come from all disciplines and from the ongoing support of hundreds of people engaged in all levels of monitoring and who will willingly give of their time and expertise because they believe in the adaptive approach to managing ecological and social problems (Holling, 1978). Keeping the process on track will take leadership and so we are calling on each of you to help make this happen as we believe that scenario testing tools that enable land and water use decisions to be evaluated in the context of cumulative effects to societal values are an integral solution to making a sustainable world.

References


Shaver, E. J., and J. R. Maxted. 1995. The use of impervious cover to predict ecological condition of wadeable nontidal streams in Delaware. Delaware County Planning Department, Ellicott City, Maryland.


Appendix I

Model Predictor Variable requirements

Depending on the indicators chosen, and the area under consideration, information might be attributed from stream segments, watersheds or landscapes. As an example of how geospatial information would contribute to model development and the DSS we analyze the needs for a VC that has as a basis indicators of stream condition (e.g., hydrology). Terrestrial, social, economic and many human health metrics might require many of the same base datasets, but for simplicity they are not reported here.

For hydrologic indicators it is certain that watershed polygons of each segment that can be hierarchically linked to define their cumulative position in the watershed such as is generated using platforms such as ArcHydro (Maidment, 2002) will be a key requirement. Attributes of each segment (see Table 1) located upstream of a measurement point (e.g., xy in Figure 1) will form the foundation of subsequent analysis. Factors to be considered should facilitate the prediction of extreme events that are likely to influence the indicators (e.g., storm flows, drought, etc). The following list represents the likely spectrum of attributes to be considered within a modeling exercise and it is recognized that redundancy will be present. Most existing models available today utilize a much smaller subset of predictor variables that provide a useful and readily achievable tool that could be improved as additional datasets emerge and the need for more precise models evolves.

Table 1: Potential GIS based predictor variables to be considered in models to quantify stream conditions.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Drainage Area</td>
<td>All upstream drainage area (contributing area or catchment)</td>
</tr>
<tr>
<td>Geology</td>
<td>Summary of the most appropriate geology layer that summarizes contributions</td>
</tr>
<tr>
<td></td>
<td>of surface and ground water. In many areas a Quaternary Soils layer is</td>
</tr>
<tr>
<td></td>
<td>appropriate.</td>
</tr>
<tr>
<td>Soils</td>
<td>Summary of the most appropriate soils layer that summarizes shallow</td>
</tr>
<tr>
<td></td>
<td>contributions of surface water. In many areas a soils layer is appropriate.</td>
</tr>
<tr>
<td>Slope/gradient</td>
<td>The difference in elevation over a length of the segment. For standardization</td>
</tr>
<tr>
<td></td>
<td>and ease of calculation this is typically based on the total length of the</td>
</tr>
<tr>
<td></td>
<td>segment</td>
</tr>
<tr>
<td>Land Use/Land cover,</td>
<td>A summary classification of the land use and land cover</td>
</tr>
<tr>
<td>vegetation</td>
<td>(vegetation) on the landscape that includes the entire area and enables proportions and contrasts to be quantified</td>
</tr>
<tr>
<td>------------</td>
<td>---------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Riparian canopy</td>
<td>A subset of the land cover data that characterizes only the proximal conditions from polygons that are generated at specific distances from the water edge. Width of the polygon measured/attributed is likely to vary with the size of the feature as their influence on stream processes is scale dependent.</td>
</tr>
<tr>
<td>Reach contributing area</td>
<td>Proximity effects can influence stream conditions (e.g., temperature via groundwater inputs). Where appropriate and measureable these proximal areas should be measured to improve predictability</td>
</tr>
<tr>
<td>Local features of importance</td>
<td>Important land, water or biotic attributes applicable to the scale of inquiry, such as significant ground water upwelling areas, or key hydrologic features should be incorporated into the dynamic models</td>
</tr>
<tr>
<td>Spatial configuration of land cover</td>
<td>Including connectivity of both the river and terrestrial corridors and the spatial structure of the land cover</td>
</tr>
<tr>
<td>Climate/weather</td>
<td>Measures and timing of both temperature and precipitation, its variability, seasonality, and how it is predicted to change over time for scenario testing</td>
</tr>
<tr>
<td>Structures</td>
<td>As many major infrastructure attributes as can be modeled should be included, but specifically: dams; channelized and enclosed streams; and on-line ponds (including storm water ponds); and point source discharges and withdrawals of water</td>
</tr>
<tr>
<td>Regulated Flow</td>
<td>Flow regimes where they are regulated</td>
</tr>
</tbody>
</table>
Appendix II

Example use of a discrete-time model to elucidate causal pathway changes to fish

To illustrate how a discrete-time modelling approach could support the cumulative effects modeling, we will describe a model for a population of a single fish species. Their population numbers, or density, in a segment (patch) \( i \) at the beginning of year \( t \), before the young hatch, is denoted by \( U_i(t) \). During a year, there is reproduction, mortality and movement. The population densities at any particular time are predicted by equation 4:

\[
\text{Eq. 4: } U_i(t+1) = (1 - p_i) f_i(U_i(t)) + \sum_{j \neq i} c_{ij} p_j f_j(U_j(t))
\]

where \( f_i(U_i) \) is the population after reproduction (i.e. the surviving individuals from the previous year plus the young of the year) in segment \( i \); \( p_i \) is the probability of moving out of from segment \( i \), and \( c_{ij} \) is the probability of moving to segment \( i \) when leaving segment \( j \). Growth, mortality and movement probabilities will in general depend on local conditions. They could also depend on upstream conditions. This latter aspect typically does not arise in terrestrial systems and is therefore not present in classical patch models, but is currently under investigation (F. Lutscher, Univ. of Ottawa, pers. Comm.). If local conditions are suitable and upstream conditions are favorable, we expect that the target fish species growth will be high, mortality will be low, the probability of leaving a segment will be small, and the probability of moving to that segment will be large. The above mentioned approaches to measure suitability could therefore inform parameter choices for such a patch model.

One measure estimated from the model is the long-term population growth rate, which predicts population persistence or extinction, and the corresponding reproductive value, on a segment (patch) basis (Caswell, 2001). The calculation of the sensitivity of this quantity with respect to each model parameter is well understood and fairly easy to implement (Caswell, 2001). But the long-term population growth rate is not the only information that can be gained from a patch model. Other, more sophisticated quantities that can be derived from the same modelling approach include the: (i) contribution of a patch to overall population capacity; (ii) contribution to the expected fraction of occupied reaches, (iii) contribution to the expected time to extinction, (iv) contribution to colonization events in the network (Ovaskainen 2001, Ovaskainen and Hanski 2003a, 2003b). There are a few applications of such models to fish populations (e.g. Charles et al. 1998, 2000, Chaumot et al., 2003, 2006, Goldberg et al., 2010), but to our knowledge, none of them explored the importance of HDF on downstream populations. We are not aware of any community models in watersheds, but metapopulation or patch-occupancy models have been used occasionally (Fagan, 2002, Perkin et al., 2013). The seemingly most promising approach to link patch models to existing data is to take a metapopulation viewpoint and compute the expected fraction of occupied reaches from the model. Assuming that a quasi-steady state has been reached in the fish assemblages in this region, the extensive data (> 1200 sample sites) could be compared to these predicted states as a means of both validating and improving the models in an iterative process.